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Enterococci Concentrations in Diverse Coastal Environments Exhibit Extreme Variability

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Fecal indicator bacteria (FIB) concentrations in a single grab sample of water are used to notify the public about the safety of swimming in coastal waters. If concentrations are over a single-sample standard, waters are closed or placed under an advisory. Previous work has shown that notification errors occur often because FIB vary more quickly than monitoring results can be obtained (typically 24 h). Rapid detection technologies (such as quantitative polymerase chain reaction) that allow FIB quantification in hours have been suggested as a solution to notification errors. In the present study, I explore variability of enterococci (ENT) over time scales less than a day that might affect interpretation of FIB concentrations from a single grab sample, even if obtained rapidly. Five new data sets of ENT collected at 10 and 1 min periodicities for 24 and 1 h, respectively, are presented. Data sets are collected in diverse marine environments from a turbulent surf zone to a quiescent bay. ENT vary with solar and tidal cycles, as has been observed in previous studies. Over short time scales, ENT are extremely variable in each environment even the quiescent bay. Changes in ENT concentrations between consecutive samples (1 or 10 min apart) greater than the single-sample standard (104 most probable number per 100 mL) are not unusual. Variability, defined as the change in concentration between consecutive samples, is not distinct between environments. ENT change by 60% on average between consecutive samples, and by as much as 700%. Spectral analyses reveal no spectral peaks, but power-law decline of spectral density with frequency. Power-law exponents are close to 1 suggesting ENT time series share properties with 1/*f* noise and are fractal in nature. Since fractal time series have no characteristic time scale associated with them, it is not obvious how the fractal nature of ENT can be exploited for adaptive sampling or management. Policy makers, as well as scientists designing field campaigns for microbial source tracking and epidemiology studies, are cautioned that a single sample of water reveals little about the true water quality at a beach. Multiple samples must be taken to gain a snapshot into the patchy structure of microbial water quality and associated human health risk.

Introduction

The United States Clean Water Act and BEACH Act require coastal states to monitor recreational waters for fecal indicator bacteria (FIB) to assess water quality. Exposure to FIB from municipal wastewater and urban runoff in marine waters correlates to adverse health outcomes in swimmers according to formal epidemiology studies (*1–3*). Monitoring results are used for public notification of water quality via beach advisories and closures. In the United States, 98% of agencies conducting monitoring use a single-sample exceedance criteria for issuing advisories and closures (*4*). If FIB concentrations in a single grab sample of water exceed the criteria, public notification of poor water quality is required. For enterococci (ENT), the preferred FIB for monitoring marine waters (*5*), the recommended single sample standard for beaches is 104 most probable number (MPN) of colony forming units (CFU)/100 mL (*6*).

United States Environmental Protection Agency approved methods to measure FIB require an 18–96 h incubation period as they are culture-based. Several studies have shown that temporal changes in FIB concentrations in beach water occur at shorter time scales (*7, 8*). Thus, out-of-compliance beaches remain open during the laboratory incubation period and may be in compliance by the time warnings are posted (*8, 9*). Rapid detection technologies are culture independent, allowing FIB quantification in under 4 h (*10, 11*). Transitioning to rapid methods has been proposed as a means for addressing management errors resulting from the delay associated with culture-based assays.

However, there is strong evidence that no matter how rapidly a test result can be obtained, a single sample of water will not adequately describe water quality for an entire day. It is now known that FIB vary at time scales less than a day. In particular, FIB vary with tidal and solar cycles (*12, 13*) which modulate their transport and inactivation in coastal waters, respectively. Fortunately, the manner in which FIB vary with tides and sunlight is predictable, so health-protective monitoring can be conducted (for example, periods with highest FIB can be sampled). A single study has documented FIB variability at time scales less than an hour in a turbulent surf zone and attributed this to rip cell mixing (*14*). In this case, variation did not appear to be predictable. More work is needed to examine FIB variability at short time scales (less than an hour) at diverse beach environments to determine if short-period variability is present along all coastlines or only present in turbulent surf zones. Such extreme variability could have profound influence on the policy outcomes (i.e., beach advisories and closures), monitoring plans, and usefulness of rapid detection technologies.

There is reason to believe that FIB variability at time scales less than an hour will be common based on work with other physical, chemical, and biological parameters in the coastal environment (*15–18*). For example, temporal variability in temperature, nitrite, and fluorescence has been documented at scales of seconds to hours in coastal waters (*15, 16, 19*). These studies found that parameter variability, or "patchiness", is not confined to a set of frequencies, nor did they find that the variability is random (i.e., white noise). Rather, they found that extreme variability of many coastal parameters is fractal in nature. That is, variability is observed at all time scales and there is no characteristic time scale associated with the signal.

Fractal time series are identified from a power-law decay in spectral density (E) with frequency (f) (*16*). The power

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TABLE 1. Descriptions of Experiments Included in the Study (Freq. is the Frequency at Which Samples Were Collected during the Experiments; "Building" Indicates that the Waves Increased from 0 to 1 m over the Course of the Experiment)

site	location	start	end	freq. (1/min)	tide range (m)	breaker height (m)
HSB02	Huntington Beach	4/12/02 16:00	4/13/02 10:00	0.1	1.4	1
LPS05	Lovers Point, South	10/22/05 11:00	10/23/05 9:00	0.1	1.5	0–1 (building)
LPS07	Lovers Point, South	2/3/07 11:00	2/4/07 11:00	0.1	1.7	0
LPN07	Lovers Point, North	2/3/07 11:00	2/4/07 11:00	0.1	1.7	0
LPmin	Lovers Point, South	10/23/05 2:00	10/23/05 3:00	1	0.2	1

law-exponent β in $E(f)\sim f^{-\beta}$ can be related to the fractal dimension D as follows: $D = 2 - 0.5(\beta - 1)$ where D varies between 1 and 2 (*16*). D and β are useful for describing how energy in a time series varies from one time scale to the next. Their magnitudes are controlled by physical (e.g., turbulent velocities and dispersion) and biological (e.g., variation in growth and grazing rates) processes (*15, 18*). If $\beta = 0$, the signal in the time domain is referred to as white noise because $E(f)$ is constant. In this case, the signal is not fractal, but is considered random because variability at every frequency contributes equally to the time series. If $\beta = 1$, the signal is fractal and classified as $1/f$ noise which is ubiquitous in nature (for example, flow in streams (*20*) and DNA sequences (*21*)). In this case, the energy associated with each frequency falls off as frequency increases. Because $E(f)$ and f are related, the signal in the time domain is considered structured. When turbulent velocities are responsible for advecting a passive scalar, $\beta = 5/3$ as described by Kolmogorov (*15*).

In the present study, I examine extreme temporal variations (periods between 1 min and 24 h) in FIB concentrations in diverse marine coastal environments ranging from wave-sheltered to wave-exposed open ocean beaches. I report five new ENT data sets, collected at 10 and 1 min periodicities. A goal of this paper is to determine if ENT variation at short time scales is dictated by the physical environment in which they were measured (i.e., a quiescent, wave-sheltered cove or a turbulent surfzone). In addition, I examine how variation at different time scales or frequencies contributes to the overall ENT signal using Fourier analysis. In particular, I examine if high frequency variability is random or fractal in nature. The implications of the results for monitoring beaches for ENT and human health risk are discussed.

Materials and Methods

Enterococci (ENT) are the focus of this study because they correlate best to human health outcomes in marine waters (*5*). ENT concentrations were measured every 10 min for 18 h at Huntington State Beach (HSB, 33°38′ N, 117°58′ W) in 2002, and every 10 min for 22 and 24 h in 2005 and 2007, respectively, at Lovers Point, CA (LP, 36°37′ N, 121°55′ W). In 2005, ENT concentrations at LP were measured every 1 min for approximately an hour during the longer duration 10-min study (Table 1). During each experiment, samples were taken at a fixed location, and thus sampling was Eulerian in nature.

Tides and waves are major factors affecting mixing and transport in the very nearshore and might explain heterogeneity in ENT variability between experiments. To characterize the tides and waves during each experiment, tide level and range were obtained from XTide (http://www.flaterco.com/xtide/files.html) and breaker heights were recorded visually by the author (Table 1). In 2002, water samples were collected from HSB at station 6N (*22*) (hereafter referred to as experiment HSB02). HSB is characterized by a well-developed surfzone, and during HSB02 breakers were 1 m high. During 2005 and 2007, samples were collected at LP, which is sheltered from waves under the majority of conditions except during long-period northwest (NW) swell. During 2005, I sampled LP at a single location on the beach once every 10 and 1 min, as described above (hereafter referred to as LPS05 and LPmin for 10 and 1 min period experiments, respectively, Table 1). The experiments began under quiescent conditions with no waves, and over the course of the study a NW swell built until 1 m waves were breaking on the beach. During 2007 at LP, I collected samples at two locations on the beach, approximately 50 m apart (sites N and S) (hereafter referred to as LPN07 and LPS07, respectively). Waves were absent during the entire study, and the water was extremely quiescent. The tide range during all studies was similar, with the exception of the study where samples were collected every minute for 1 h at LP (LPmin) during which the water level barely changed.

Fifty mL of water was collected in sterile containers and immediately stored on ice and analyzed within an hour of collection. Prior to analysis, containers were mixed by inverting three times. Ten mL subsamples were assayed for ENT using Enterolert defined chromogenic substrate assays implemented in a 97-well format (IDEXX, Westbrook, ME). An interlaboratory comparison study in southern California using waters adjacent to HSB found that Enterolert yielded results consistent with traditional methods of membrane filtration and multiple tube fermentation with low error rates (*23*). Therefore, Enterolert is expected to perform well in the present study. Ten mL of well-mixed sample water and reagent were dispensed into 90 mL of Butterfields buffer. This allowed detection of ENT between 10 and 24192 most probable number (MPN)/100 mL. Concentrations and 95% confidence intervals were determined from MPN tables. The 95% confidence intervals represent a measure of the method uncertainty. For data analysis purposes, ENT concentrations below the lower limit of detection (10 MPN/100 mL) were assigned a value of 5 MPN/100 mL.

Data were analyzed using SPSS v.11 (SPSS) and Matlab v7.0.4 (Mathworks). Kruskal–Wallis tests were used to compare ENT concentrations measured between sites or conditions. Following Whitman and Nevers (*24*), the number of samples (n) required during the experiments to achieve a specific level of certainty, or coefficient of variation (CV), about the experiment average ($\bar{x}$) given the standard deviation (s) was calculated as $n = (s/CV\bar{x})^2$. CV values of 20% and 50% were chosen for simplicity, although any CV could have been used.

Fourier transforms were applied to detrended ENT data series. To determine whether spectral densities decayed as power laws with frequency and were thus fractal, spectral density estimates were averaged within equal logarithmically spaced intervals following Lovejoy et al. (*15*). Linear regressions were applied to determine power-law exponents β and their 95% confidence intervals. This approach assumes that a single fractal dimension can be used to describe data (*16*).

Results

Ten Minute Time Series. Time series of ENT measured once every 10 min are illustrated in Figure 1 along with tide level (HSB02, LPS05, LPS07, LPN07). High frequency variability is evident that cannot be explained by measurement uncer-

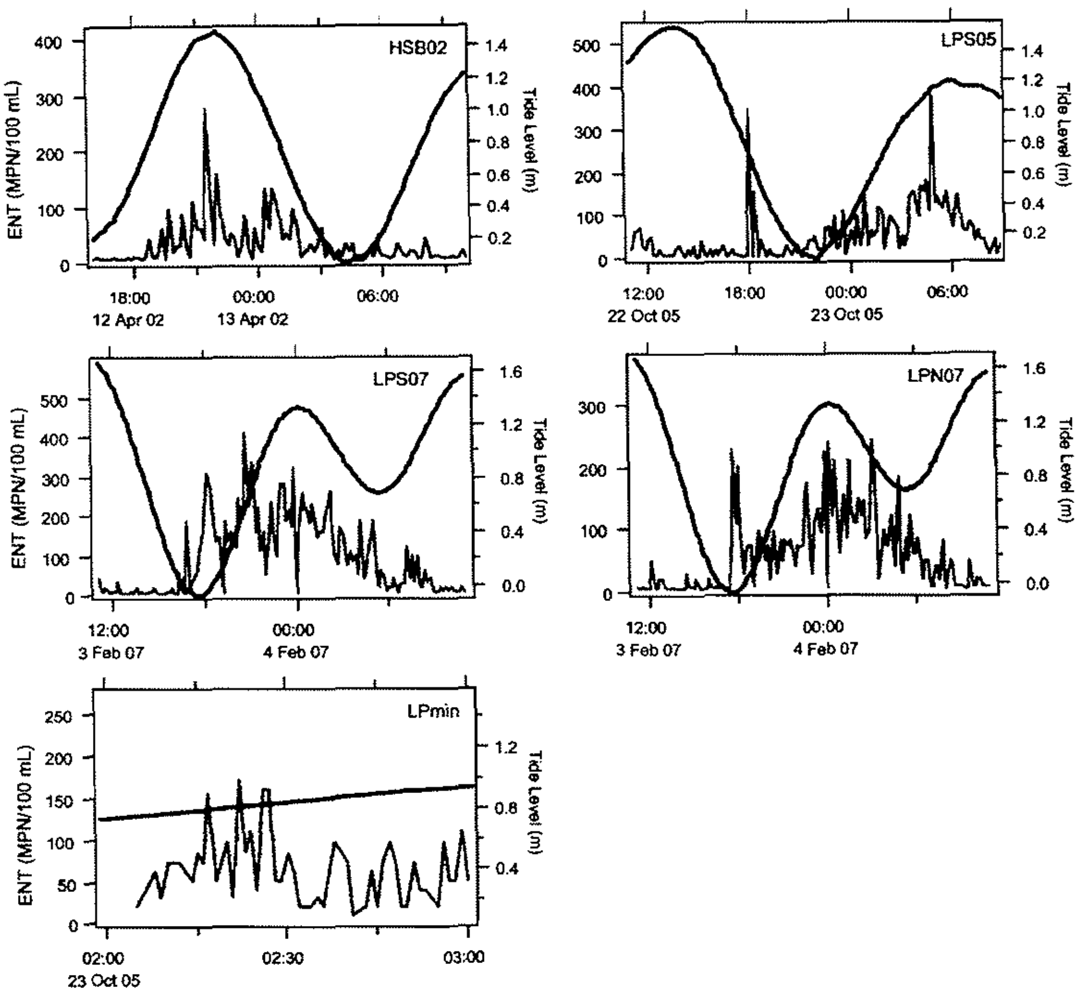

FIGURE 1. ENT time series analyzed in the present study. Shaded areas represent 95% confidence intervals about each measurement as determined from MPN tables, black line is measured ENT. Heavy black line is tide level (shown on right axes). The code in the upper right corner describes the location and time of experiments (see Table 1).

tainty. This is based on the fact that measurements do not fall within the 95% confidence bounds of one-another (gray shading in Figure 1). Confidence intervals varied according to concentration as determined by the MPN tables and ranged from 37 to 311 MPN/100 mL (Figures S1 and S2 in the Supporting Information). ENT distributions measured during the experiments are significantly different from each other ($p < 0.05$) with the exception of LPN07 and LPS05 which are similar ($p > 0.05$) (Figure 2). The highest ENT concentrations were measured at site LPS07, followed by LPN07 and LPS05, and HSB02 (Figure 2, Table 2). The number of samples with ENT below the lower detection limit of 10 MPN/100 mL is reported in Table 2. No measured concentration was over the upper detection limit.

All sites display significant diurnal patterns: ENT concentrations are significantly higher at night compared to the day ($p < 0.05$). This supports reports of the sunlight inactivation of indicator organisms in natural waters (*25*). All sites show significant variation with tide. ENT concentrations at LP sites (LPS05, LPS07, LPN07) are higher during flood compared to ebb tides ($p < 0.05$). In contrast, ENT concentrations at HSB02 are higher during ebb compared to flood tides ($p < 0.05$). These results are in agreement with previous reports of semidiurnal variation of ENT at these beaches and are likely due to tidal modulation of ENT sources (*26, 27*).

The average change in ENT concentration between consecutive samples during the experiments ranges from 26 (HSB02) to 45 (LPS07) MPN/100 mL per 10 min (Table 2). The maximum change in ENT concentration between samples is 345 MPN/100 mL per 10 min measured at LPS05. At all sites, the maximum change in ENT concentration between consecutive samples is greater than the California single-sample ENT standard of 104 MPN/100 mL. This indicates that changing the sampling time by as little as 10 min could result in a change in the posting or advisory status of the beach. There are instances when there is no change between ENT measurements between consecutive samples. Many of these (approximately 40%) occur when 5 MPN/100 mL is assigned as the lower limit of detection and thus may be an artifact of our detection limit.

The difference between ENT concentrations measured in consecutive samples relative to the experiment average (δ) was calculated (Table 2). The distributions of δ are not different between experiments ($p > 0.05$) and range from 0 to 7 (10 min)$^{-1}$ and average 0.6 (10 min)$^{-1}$. This means that overall, ENT concentrations typically vary by 60% every 10 min.

Using the standard deviations and means reported in Table 2, a beach manager would need to collect 39 (HSB02), 31 (LPS05), 25 (LPS07), and 25 (LPN07) samples to obtain an estimate of concentration within a coefficient of variation of

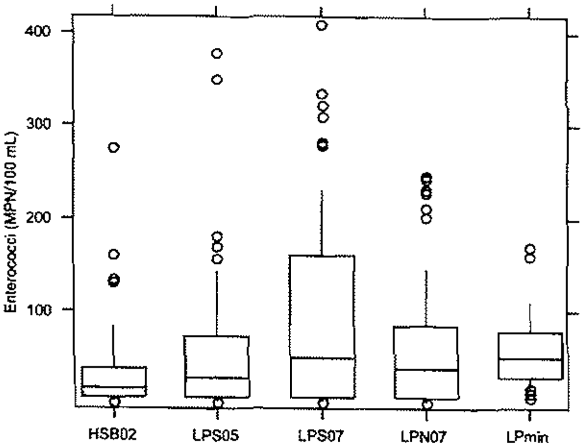

FIGURE 2. Box and whisker plots show the range of ENT concentrations measured during each study. The lower, middle, and top box edges correspond to the 25th, 50th, and 75th percentiles of the indicated set of measurements, the "whiskers" indicate the 10th and 90th percentiles, and the symbols show measurements lower and greater than the 5th and 95th percentiles, respectively.

TABLE 2. ENT Concentration Measurement Results[a]

experiment	*N*	UD	ave	SD	GM	ave change (min−max)	ave δ (min−max)
HSB02	102	24	33	41	19	26 (0–234)	0.8 (0–7.1)
LPS05	131	14	54	60	31	35 (0–345)	0.6 (0–6.4)
LPS07	144	22	96	95	44	45 (0–318)	0.5 (0–3.3)
LPN07	144	28	60	59	32	36 (0–238)	0.6 (0–4.0)
LPmin	49	0	62	39	51	34 (0–140)	0.5 (0–2.3)

[a] *N* is the number of samples collected and UD is the number of samples with ENT below the lower detection limit of 10 MPN/100 mL; ave is arithmetic average, SD is standard deviation, GM is geometric mean, all with units of MPN/100 mL; ave change is the average change between consecutive samples with minimum and maximum given in parentheses and units of MPN/100 mL per 10 min except for LPmin where units are MPN/100 mL per min; ave δ is the average change between samples relative to the experiment average with units (10 min)$^{-1}$ except for LPmin where units are (min)$^{-1}$.

20% about the experiment mean. If a coefficient of variation of only 50% were desired, 6 (HSB02), 5 (LPS05), 4 (LPS07), and 4 (LPN07) samples would be required.

There are no peaks in the spectral densities at specific frequencies (Figure 3). Rather, spectral densities decay as power-laws with frequency. Power-law exponents β for each spectra are within 95% confidence of 1 with the exception of LPS05. β for LPS05 ranges between 0.3 and 0.9 with 95% confidence. All linear regressions were statistically significant (r values reported in Figure 3, $p < 0.05$).

Spatial Variation between LPS07 and LPN07. During the LP experiment during 2007, samples were collected concurrently at two sites on the beach approximately 50 m apart. The measurements at these sites are well correlated to each other (Spearman's $\rho = 0.71$, $p < 0.05$); however the two data series are significantly different ($p < 0.05$) with LPS07 having higher ENT concentrations than LPN07. The same concentration was measured simultaneously at the two sites 18 out of 144 (12.5%) times. The mean difference between measurements at LPS07 and LPN07 collected at the same time is 56 MPN/100 mL and the maximum is 379 MPN/100 mL. Importantly, 59/144 (41%) measurements at LPS07 are over the California single-sample standard of 104 MPN/100 mL while only 27/144 (19%) are over the standard at LPN07.

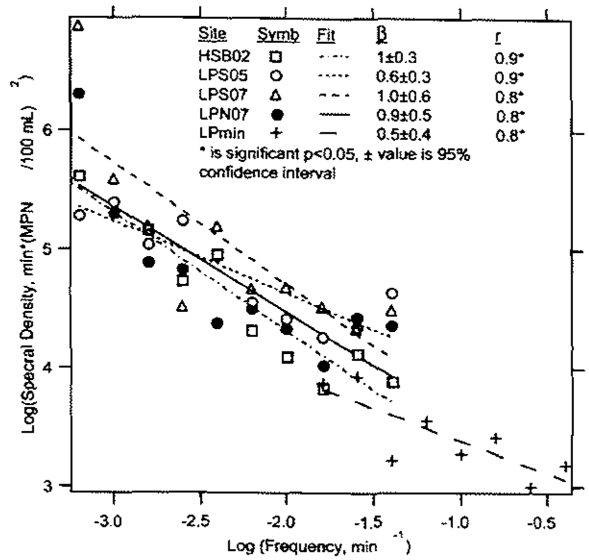

FIGURE 3. Log_{10}-transformed spectral densities plotted as a function of log_{10}-transformed frequency. Linear regressions are shown as lines, with slopes (equal to β in $E(f) \sim f^{-\beta}$) Pearson's r, and p value for regression. "Symb" in legend is the symbol used for each experiment.

Thus, the probability of measuring an exceedance of water quality standards changes depending on where one is sampling on the beach, within a short distance of 50 m. One explanation for the higher concentrations at LPS07 is that the site is located closer to a storm drain on the beach than LPN07. However, 28% of the simultaneous samples were actually higher at LPN07, the site further from the storm drain, indicating that proximity to the storm drain is not the only factor that impacts ENT concentrations.

One Minute Time Series. I measured ENT every minute for one hour during the LPS05 experiment (LPmin, Figure 1). ENT were extremely variable (average change of 34 MPN/100 mL per minute, maximum change of 140 MPN/100 mL per min). As with the 10 min time series, the variation between samples cannot be explained by measurement uncertainty based on the fact that measurements do not fall within the 95% confidence bounds of one-another (gray shading in Figure 1). The changes in ENT concentrations and δ between consecutive samples during this hour are not significantly different from those observed during the LPS05 10-min experiment ($p > 0.05$). If an estimate of ENT concentration with a coefficient of variation of 20% and 50% relative to the 1 h experiment mean were desired, then 10 and 2 samples would be required, respectively.

Spectral density decays as a power law with frequency with $\beta = 0.5 \pm 0.4$ (Figure 3, $r = 0.8$, $p < 0.05$). The exponent is not different ($p > 0.05$) from that measured for LPS05 where $\beta = 0.6 \pm 0.3$, suggesting that the scaling observed with the lower frequency LPS05 data set applies to a greater range of frequencies.

Discussion

ENT concentrations collected at 10 and 1 min intervals along the shoreline of marine beaches illustrate that temporal variability is extreme. Changes in ENT concentrations between consecutive samples greater than the California single-sample standard of 104 MPN/100 mL are not unusual. Extreme variability is present in experiments conducted in a turbulent, well-mixed surf zone (HSB02), in waters transitioning from quiescent/tide-dominated to wave-dominated (LPS05 and LPmin), and in a quiescent tide-dominated environment (LPN07 and LPS07). Variability, measured as the change in consecutive ENT measurements normalized by the experiment average ENT concentration, is not different between sites, thus does not appear to be a function of the degree of wave exposure.

It should be noted that the extreme variability documented here is not a result of the method used to enumerate ENT. In another study, we used membrane filtration in conjunction with mEI media to measure ENT concentrations at LP every 20 min (*26*). We saw similar ENT variability. It is likely that any ENT analysis method will give similar results regarding variability. However, experiments need to be conducted to document variability with methods that measure nucleic-acid targets for ENT quantification.

Although results are not reported here, *E. coli* were also measured using Colilert-24 and Colilert-18 (IDEXX) during the experiments described in Table 1. Colilert has been shown to perform well in California marine waters for *E. coli* enumeration (*23, 28*). Conclusions regarding variability in ENT apply to these bacteria as well. It is likely that variability over similar time scales will apply to other microbial targets including source-specific markers like those in *Bacteroidales* (*29*), but this should be confirmed.

Low frequency patterns associated with sunlight and tides are apparent in each time series that lasted for longer than 1 h. It is interesting that neither diurnal nor semidiurnal peaks are evident in the spectra (Figure 3). This is likely due to the relatively short duration of the time series relative to diurnal and semidiurnal periods.

Despite the lack of spectral peaks, coastal ENT concentrations are structured because time series can be described mathematically as decaying power-laws in the frequency domain. Even though the physical environments studied are different with regard to wave exposure, ENT concentrations are structured similarly with power-law exponents close to 1 (Figure 3). The fact that the power-law exponents are not equal to zero implies that the variability is not random, or white noise, as this would have produced a flat spectra. ENT time series share properties of $1/f$ noise (*30*) and have a fractal dimension $D \sim 2$. Seuront and Lagadeuc (*31*) report D between 1.367 and 1.626 for temperature, salinity, and fluorescence in tidally mixed waters in the English Channel. $E(f)$ of their data series declined more rapidly with increasing f, compared to those in Figure 3. Relative to my data series, low-frequency oscillations were more dominant than high-frequency oscillations in their data series.

The fact that the ENT data share characteristics with $1/f$ noise indicates ENT are "patchy" and that there were ENT patches or filaments of all durations or sizes transported by the fixed sampling site during the experiments. Patchiness in time and space is expected to develop in coastal environments where intermittent sources, nonuniform currents, turbulent diffusion, and changing chemical or biological characteristics influence persistence and transport of ENT (*15, 32*).

How knowledge of the fractal dimension of the ENT series might be harnessed to provide recommendations for sampling plans to protect human health is not clear. By definition, a fractal time series has no characteristic time scale associated with it, so sampling at a particular time interval cannot be recommended. An important point is that ENT concentrations are not random white noise even though there are no spectral peaks. More work on understanding fate and transport of ENT in coastal waters is needed so that researchers can fully understand how patchiness develops.

The result reported here regarding extreme variability presents a challenge to policy makers and the protection of human health. Assuming ENT are from an urban runoff or municipal wastewater source and the epidemiological models (*1–3*) are correct, ENT concentrations correlate to health risk. This suggests that not only are ENT patchy in time and space behaving as $1/f$ noise, but so are human pathogens and human health risks. An inability to estimate the true concentration of ENT in coastal waters limits our ability to protect human health. A way of sampling the coastal ocean for ENT to uncover a true estimate of human health risk is needed. If a health-protective estimate is desired, then sampling should be conducted at night during ebbing (at HSB) or flooding (at LP) tides. The high frequency variability indicates that regardless of sampling time, a single sample of water tells one little about the true water quality, so multiple samples need to be collected. If it is not feasible to collect multiple samples, then a spatially or temporally composited sample will improve the estimate of the true water quality. At minimum, consecutive samples collected at 1 min intervals could be composited to obtain a better estimate of water quality. Policy makers, as well as scientists designing field campaigns for microbial source tracking and epidemiology studies, are cautioned that a single sample of water reveals little about the true water quality at a beach.

Predictive models (*22, 33–35*) may help to estimate average water quality given high frequency variability of measurements. These models use physical, chemical, and biological factors to predict concentrations of ENT. If enough high quality data are used to train models, they may be able to provide better estimates of the central tendency of daily ENT concentrations than single grab-sample measurements. Future work should examine this possibility by comparing model predictions to high frequency data measurements.

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Supporting Information Available

Figures S1 and S2. This information is available free of charge via the Internet at http://pubs.acs.org.

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Public Mis-Notification of Coastal Water Quality: A Probabilistic Evaluation of Posting Errors at Huntington Beach, California

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Whenever measurements of fecal pollution in coastal bathing waters reach levels that might pose a significant health risk, warning signs are posted on public beaches in California. Analysis of historical shoreline monitoring data from Huntington Beach, southern California, reveals that protocols used to decide whether to post a sign are prone to error. Errors in public notification (referred to here as posting errors) originate from the variable character of pollutant concentrations in the ocean, the relatively infrequent sampling schedule adopted by most monitoring programs (daily to weekly), and the intrinsic error associated with binary advisories in which the public is either warned or not. In this paper, we derive a probabilistic framework for estimating posting error rates, which at Huntington Beach range from 0 to 41%, and show that relatively high sample-to-sample correlations (>0.4) are required to significantly reduce binary advisory posting errors. Public mis-notification of coastal water quality can be reduced by utilizing probabilistic approaches for predicting current coastal water quality, and adopting analog, instead of binary, warning systems.

Introduction

Many government-sponsored environmental monitoring programs issue health advisories whenever pollutant concentrations reach levels that might pose a threat to human health. The utility of health advisory programs logically depends on their ability to disseminate timely and accurate information, in a format that is useful and easy to understand. This study examines the health advisory component of a large (statewide) shoreline water quality monitoring program in California. Health advisories take the form of warning signs that are posted at public beaches whenever shoreline water quality (as measured by fecal indicator bacteria) fails to meet one or more of seven different state standards. The California health advisory program is one of a growing number of such programs nationwide, sponsored in part by the Federal Beaches Environmental and Coastal Health Act passed by the U.S. Congress in October 2000 (*1*–*4*). A noteworthy aspect of the California program is its binary nature, in which information about coastal water quality is conveyed to the public by the presence or absence of warning signs on the beach during the high-use period from April 1 through October 31 of every year. This binary approach stands in contrast to other long-standing reporting programs, for example, weather forecasts, in which the information provided to the public is probabilistic in nature (*5*).

In this paper, we set out to answer several questions: (1) What is the magnitude of error associated with binary health advisories? (2) How are these error rates affected by the degree to which the concentrations of bacteria in consecutive samples are correlated? (3) Can the accuracy and effectiveness of health advisories be improved by changing the way data are collected and analyzed and/or by changing the way water quality information is conveyed to the public? To answer these questions, we develop a probabilistic framework for analyzing posting errors and compare the theory to observations of posting errors at Huntington Beach in southern California. Huntington Beach is an ideal natural laboratory to examine shoreline water quality issues because of the magnitude of the historical water quality problem, the wealth of available shoreline monitoring data, and the fact that a series of special studies have been conducted with a wide range of sampling frequencies (*6*–*8*).

Public Notification of Shoreline Water Quality in California

Beginning July 1, 1999, the State of California mandated fecal indicator bacteria monitoring at all public beaches with more than 50 000 annual visitors and established seven statewide concentration standards for fecal indicator bacteria in the surf zone. When the concentration of indicator bacteria at a monitoring site exceeds any of the California standards, the local health official must post a sign warning beach goers of potential health risks associated with entering the water (*surf zone posting*). If a sewage spill is suspected, the local health official may close the surf to public access (*surf zone closure*). Four of the seven standards are single-sample standards, for which a monitoring site is considered to be out of compliance if the concentration of indicator bacteria in a single sample exceeds specified concentrations for total coliform (TC), fecal coliform (FC), and *Enterococcus* species (ENT). The California single-sample standards for TC, FC, and ENT are respectively 10 000, 400, and 104 most probable number (MPN) or colony forming units (cfu)/100 mL; a fourth single-sample standard for TC of 1000 MPN or cfu/100 mL applies when the TC/FC ratio falls below 10. The remaining standards are 30-day geometric mean standards, for which a monitoring site is considered to be out of compliance if the geometric means of TC, FC, and ENT in all samples collected within a 30-day period exceed 1000, 200, and 35 MPN or cfu/100 mL, respectively. These standards correspond, at least theoretically, to a threshold rate of bather illness of 19 cases of highly credible gastrointestinal disease for every 1000 bathers. (*3*, *9*–*11*) There are many historical reasons for choosing this particular threshold, including the fact that it represents the background rate of gastrointestinal illness among the general population (*12*).

Observations of Posting Errors at Huntington Beach

The surf zone posting protocols described above were adopted with the goal of conveying to the public up-to-date information about surf zone water quality. However, a post de facto comparison of posting records and water quality test results indicates that the public is often mis-notified about current water quality conditions. This point is illustrated in Figure 1A where we compare measurements of

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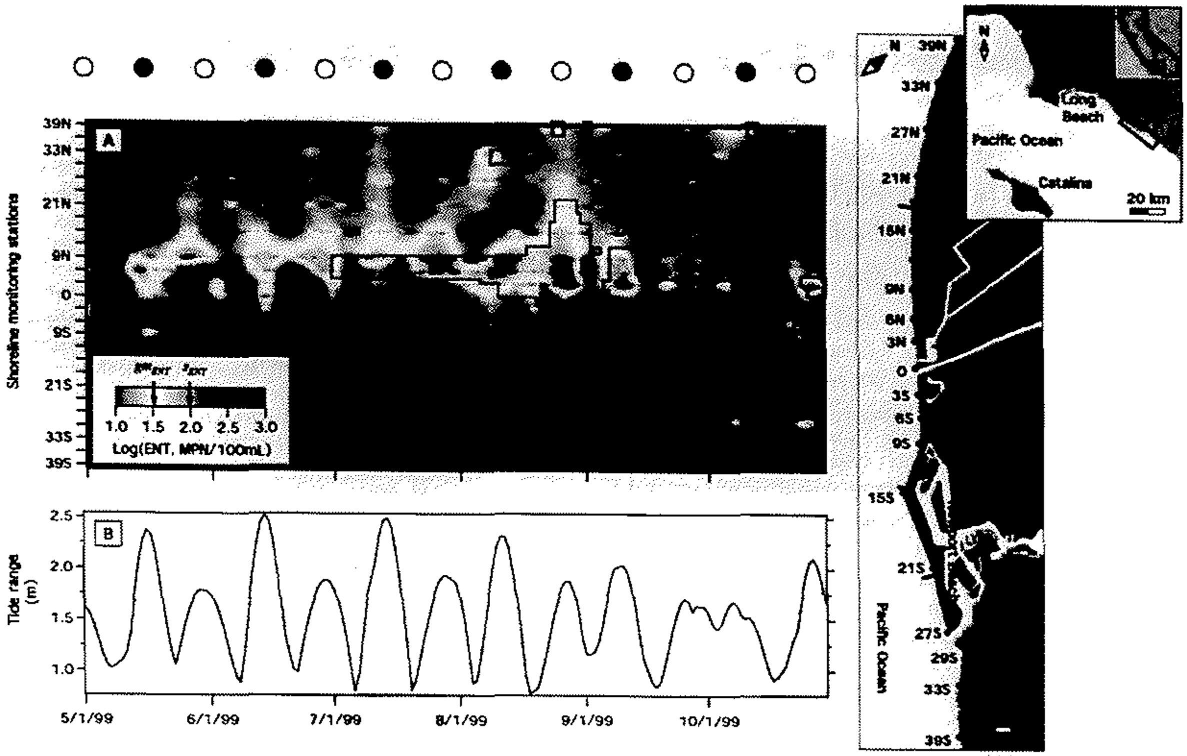

FIGURE 1. (A) Log-transformed concentration of ENT in the surf zone at Huntington Beach and Newport Beach plotted against surf zone stations (vertical axis, see also map) and time (horizontal axis). Each sample collected from the surf zone appears as a small dot in the plot. Open and closed circles at the top of the figure indicate the timing of full and new moons, respectively. Black and red polygons represent the posting and closure history, respectively. gm_{ENT} and s_{ENT} refer to the California geometric mean and single-sample standards for ENT in the surf zone. Several posting events (labeled 1–4) are discussed in the text. (B) Plot of the maximum daily tide range, defined as the difference between the daily higher-high and lower-low tides.

ENT in the surf zone at Huntington Beach (color ranging from blue to black) with the posting and closure history (black and red polygons, respectively) over the period May 1–October 31, 1999. This time period was selected because it includes the summer of 1999 when Huntington Beach experienced a record number of postings and closures, and it straddles the start of California's new water quality regulations that went into effect July 1, 1999. Most (95%) of the postings indicated in the figure were triggered by the single-sample and geometric mean standards for ENT. Yet there are many instances when the concentration of ENT exceeded the single-sample standard but signs were not posted (referred to here as underprotection errors) or where the concentration of ENT was below the single-sample standard but signs were posted (overprotection errors). Often, overprotection errors immediately follow underprotection errors (e.g., see events 1–3 in Figure 1A). Presumably, posting errors caused by the single-sample standards originate from the variable nature of water quality in the surf zone (see next section) and the inherent time delay (ca. 2–3 d) between when a sample is collected and when a sign is posted or taken down. The geometric mean standard also triggered overprotection errors involving multiple shoreline stations and lasting several weeks (see last half of event 4 in Figure 1A). The geometric mean is computed from test results collected at a particular site over the preceding 30 days (the so-called 30-day geometric mean standard); therefore, once a violation has occurred, a relatively large sequence of compliant samples are required before the geometric mean falls back below the standard. It should also be noted that the use of geometric means for evaluating human health risk has been challenged on theoretical grounds (*13*).

Importantly, we note that beach *closures* can be more accurate predictors of poor water quality than beach *postings*. Beginning on July 1, 1999, the local health officer closed sections of Huntington Beach out of concern that the surf zone contamination might be from a source of sewage (red polygon in Figure 1A). From personal observations, the health officer was aware that the concentration of fecal indicator bacteria in the surf zone at Huntington Beach was generally highest during full and new moons (when the daily tide range is maximal, compare Figure 1, panels A and B) (*14*). Awareness of these lunar cycles influenced the health officer's decisions about when to close the beach and was one of the factors that allowed him to correctly anticipate the large pollution event that occurred during the full moon in early September (see large closure event in Figure 1A). This anecdote suggests that posting error rates might be reduced if posting protocols were designed to take into account factors known, through past experience, to influence local water quality. Some of the factors affecting surf zone water quality are described next.

Patterns and Randomness in Shoreline Water Quality

Surf zone water quality has both periodic patterns and random fluctuations. The concentration of fecal indicator bacteria in the surf zone at Huntington Beach, for example, exhibits a cascade of periodic patterns including (*6–8*): (1) tidal cycling in which the concentration is higher during ebb tides and lower during flood tides (or vice versa); (2) diurnal cycling in which the concentration is higher at night and lower during the day; (3) spring–neap cycling in which the concentration is higher during spring tides and lower during neap tides (evident in Figure 1A); (4) seasonal cycling in which the concentration is higher during the winter storm season and lower during the summer dry season; (5) El Niño cycling in which the concentration is higher during stormy El Niño winters and lower during dry La Niña winters; (6) multidecadal patterns in which periodic large-scale investment in sewage and storm runoff infrastructure improves coastal water quality.

Monitoring programs can detect these periodic patterns only if the time interval between samples is smaller (by at least a factor of 2) as compared to the characteristic period of a particular pattern of interest (*15*). For example, samples must be collected at least every 3 h in order to detect tidal cycling because each ebb and flood tide lasts ca. 6 h. Routine monitoring programs in California, which typically sample each site once per day to once per week, can detect patterns 3–6 described above depending on the length of time over which data are available. Importantly, processes with characteristic periods less than the sampling interval cannot be detected because the water quality signal is aliased by the sampling program. The relative uncertainty associated with the water quality sampling and testing methods, which ranges up to 23% (*16*), is also a source of noise (*17*). In the next several sections, we develop and test a probabilistic model that can account for the repeating patterns and random noise inherent in water quality measurements. To make the results of the probabilistic analysis accessible to a broad audience, each section begins with the primary question to be addressed, immediately followed by the answer supported by the analysis.

Probability of Single-Sample Exceedences

Question: *Can the fraction of samples violating single-sample standards be predicted from statistical features of local water quality, such as measures of central tendency and spread?* Answer: *The fraction of samples violating single-sample standards can be predicted from the log-mean and standard deviation of fecal indicator monitoring data, provided that the data are well described by a log-normal distribution. Furthermore, the theory predicts and observations confirm that, under certain conditions, a marginal change in water quality can lead to a substantial change in the number of signs posted at the beach.*

The probability that the concentration of bacteria in a single sample will exceed a standard (s) can be represented mathematically as follows:

$$P_{ex} = P[C > s] = \int_s^{\infty} f_C(c)\, dc \qquad (1)$$

where C is a random concentration variable, c is a particular realization of the random variable, and $f_C(c)$ is the probability density function for the concentration of fecal indicator bacteria in the surf zone. The exceedence probability (P_{ex}) is a measure of water quality: $P_{ex} \to 1$ if water quality is very poor, and $P_{ex} \to 0$ if water quality is very good.

The monitoring data at Huntington Beach conform reasonably to a log-normal distribution (based on Kolmogorov–Smirnov normality tests (*18*); maximum difference $K - S = 0.08$ at the significant level $\alpha < 0.01$) (see Figure S1 in the Supporting Information) as do monitoring data at other coastal sites throughout the world (*19–21*). Accordingly, we replaced C with log C in eq 1 and substituted the Gaussian probability distribution function for $f_{\log C}(\log c)$. After simplification, the following relationship was obtained between the exceedence probability and a nondimensional variable referred to here as S^*:

$$P_{ex} = \frac{1}{2}\,\mathrm{erfc}\, S^* \qquad (2a)$$

$$S^* \equiv \frac{\log s - \mu_{\log C}}{\sqrt{2}\sigma_{\log C}} \qquad (2b)$$

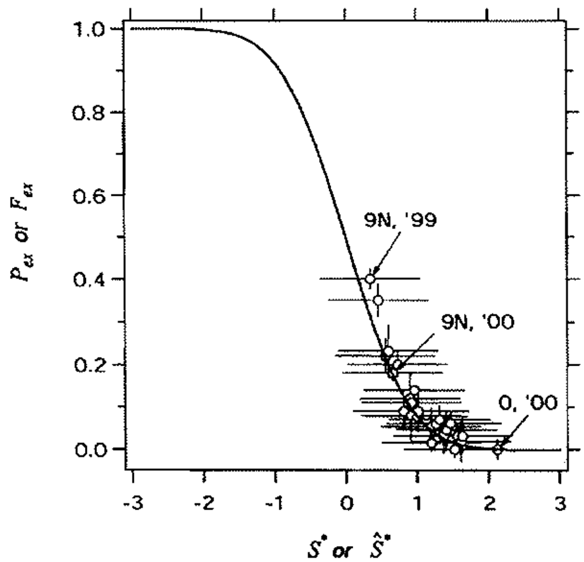

FIGURE 2. Predicted relationship between the probability P_{ex} that samples will exceed a single-sample standard and the dimensionless parameter S^* (solid line, eq 2a). Data points represent observed relationship between the fraction F_{ex} of samples collected at Huntington Beach that exceeded the single-sample standard for ENT plotted against the parameter $\hat{S}^*$. The vertical and horizontal error bars correspond to $\pm\sqrt{F_{ex}(1-F_{ex})/n}$ and $\pm 1/\sqrt{2}$, respectively, where n is the number of data points in each data bin (n = 45–65)

In these equations, erfc is the complementary error function, and $\mu_{\log C}$ and $\sigma_{\log C}$ represent the mean and standard deviation of the log-transformed bacterial concentrations, respectively. This simple theoretical result predicts that the exceedence probability decreases with increasing values of the parameter S^*, ranging from $P_{ex} > 99\%$ when $S^* < -2$ to $P_{ex} < 1\%$ when $S^* > 2$ (solid line in Figure 2). In turn, the value of S^* depends on local water quality ($\mu_{\log C}$ and $\sigma_{\log C}$) and the magnitude of the single-sample standard (log s).

These theoretical predictions compare well with observations of single-sample exceedences at Huntington Beach. To compute the latter, summertime measurements of ENT in the surf zone at Huntington Beach were grouped, or binned, by station and year. For example, one bin constituted all ENT measurements collected at surf zone station 9N during the summer of 1999; for the purposes of this analysis, summer is defined as the time period June 1–August 31. From each data bin, we calculated the fraction F_{ex} of samples that violated the single-sample standard for ENT and an empirical approximation of the parameter S^* denoted here as $\hat{S}^*$ (see Supporting Information, note that the circumflex or "hat" denotes empirical approximations of population parameters). Values of F_{ex} and $\hat{S}^*$ track the theoretical prediction closely (compare solid line with data points in Figure 2); hence, eq 2a appears to capture the relationship between measured water quality ($\hat{\mu}_{\log C}$ and $\hat{\sigma}_{\log C}$) and the fraction of samples that exceed a single-sample standard (F_{ex}). At Huntington Beach, the percentage of samples exceeding the single-sample standard for ENT ranges from a low of 0% ($F_{ex} = 0$) at surf zone station 0 during the summer of 2000 to a high of 40% ($F_{ex} = 0.4$) at station 9N during the summer of 1999 (see arrows in Figure 2).

From the shape of the theoretical curve in Figure 2, a marginal change in water quality can result in a very large or a very small change in the number of signs posted at the beach, depending on the absolute magnitude of the parameter S^*. In particular, eq 2a predicts that P_{ex} is sensitive to marginal changes in water quality when $|S^*| < 1$ and insensitive to marginal changes in water quality when $|S^*| > 1$. This observation helps to explain why the number of signs posted during the summer at 9N decreased precipitously from 26 in 1999 to 12 in 2000, despite the relatively small change in the log-mean of ENT over these two summers (1.7−1.4 units of log(MPN/100 mL)) (see two arrows in Figure 2).

Probability of Binary Advisory Posting Errors

Question: *How is the posting error rate influenced by the degree to which bacterial concentrations in consecutive samples are correlated? At Huntington Beach, are bacterial concentrations in consecutive samples correlated or independent realizations?*

Answer: *Theory predicts very high posting error rates when S^* is close to zero, even in the case where the concentrations of fecal indicator bacteria in consecutive samples are moderately correlated. This prediction is borne out by an analysis of ENT measurements in the surf zone at Huntington Beach. Posting error rates at Huntington Beach are indistinguishable from the predictions of Bernoulli trial theory, which is premised on the idea that test outcomes are independent realizations. When S^* is close to zero, posting error rates are predicted to range from 35% (for moderately correlated samples) to 50% (for weakly correlated samples).*

Let $c(t_i)$ represent the measured concentration of fecal indicator bacteria at a particular site at time t_i. The single-sample standard posting protocol can be stated succinctly as follows. A water sample is collected at a particular site at time t_{i-1}, and the concentration of fecal indicator bacteria in that sample $c(t_{i-1})$ is compared to a single-sample standard s. If $c(t_{i-1}) > s$, a sign is posted at the site; if $c(t_{i-1}) < s$, a sign is not posted at the site (or an existing sign at the site is removed). If t_i denotes the time at which a sign is posted (or removed), four possible outcomes can be identified: (1) $c(t_{i-1}) > s$ and $c(t_i) > s$, (2) $c(t_{i-1}) < s$ and $c(t_i) < s$, (3) $c(t_{i-1}) < s$ and $c(t_i) > s$, and (4) $c(t_{i-1}) > s$ and $c(t_i) < s$. No error occurs in cases 1 and 2 because the public has been correctly informed that water quality exceeds standards (case 1) or does not exceed standards (case 2). Posting error occurs when the public is incorrectly informed that water quality meets standards (case 3), or incorrectly informed that water quality does not meet standards (case 4). Consistent with the discussion of Figure 1A (see above), we refer to cases 3 and 4 as underprotection and overprotection posting errors, respectively.

Letting the superscripts U, O, and T represent underprotection, overprotection, and total posting error (defined as the sum of underprotection and overprotection errors), the following expressions can be derived for the probability that posting errors will occur at a particular site (see Supporting Information):

$$P^{\mathrm{U}}_{\mathrm{err}} = P^{\mathrm{O}}_{\mathrm{err}} = \frac{1}{2}\mathrm{erfc}\ S^*\left[1 - \frac{1}{2}\mathrm{erfc}\ S^* G(S^*,\rho(1))\right] \quad (3a)$$

$$P^{\mathrm{T}}_{\mathrm{err}} = \mathrm{erfc}\ S^*\left[1 - \frac{1}{2}\mathrm{erfc}\ S^* G(S^*,\rho(1))\right] \quad (3b)$$

The function $G(S^*,\rho(1))$ depends on the value of S^* and $\rho(1)$, where S^* has been defined previously (see eq 2b) and $\rho(1)$ is the correlation coefficient between fecal indicator bacteria concentrations at times t_{i-1} and t_i. A mathematical definition and graphical representation of $G(S^*,\rho(1))$ is included with the Supporting Information (Figure S2). The total posting error predicted by eq 3b is plotted against S^* in Figure 3. The different lines in the figure correspond to different choices of the correlation coefficient $\rho(1)$, ranging from strong positive correlation ($\rho(1) = 0.95$) to strong negative correlation ($\rho(1) = -0.95$). For the choice of $\rho(1) = 0$, the concentrations of bacteria in consecutive samples are completely uncorrelated (i.e., every concentration measurement is independent of the one before it), and the function $G(S^*,\rho(1))$ reduces to unity for all choices of S^*. In this limit, referred to here as the Bernoulli trial theory limit, the probability of an overprotective or underprotective posting error peaks at $P^{\mathrm{T}}_{\mathrm{err}} = 0.5$ when $S^* = 0$ (dashed line in Figure 3). Put another way, when the concentrations of bacteria in consecutive samples are uncorrelated ($\rho(1) = 0$) and half of samples exceed the single-sample standard ($S^* = 0$, $P_{\mathrm{ex}} = 0.5$), theory predicts that beach signage will be posted (or not posted) in error 50% of the time ($P^{\mathrm{T}}_{\mathrm{err}} = 0.5$).

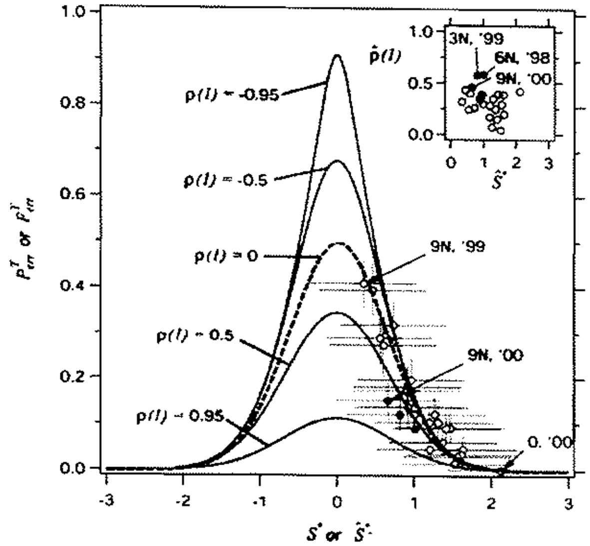

FIGURE 3. Predicted relationship between the probability of a posting error $P^{\mathrm{T}}_{\mathrm{err}}$ and the dimensionless parameter S^* (solid and dashed curves, eq 3b). The different curves correspond to different choices of the correlation coefficient $\rho(1)$. Points represent posting error rates calculated from the surf zone monitoring data at Huntington Beach. Vertical and horizontal error bars correspond to $\pm F^{\mathrm{T}}_{\mathrm{err}}\sqrt{F^{\mathrm{T}}_{\mathrm{err}}(1-F^{\mathrm{T}}_{\mathrm{err}})/n}$ and $\pm 1/\sqrt{2}$, respectively, where n represents the number of points in a particular data bin (see text). Inset is a plot of the correlation coefficient between bacterial concentrations in consecutive samples $\hat{\rho}(1)$ against the parameter $\hat{S}^*$. The red points correspond to surf zone stations and years where the posting error rate fell substantially below the error rate predicted by Bernoulli trial theory ($\rho(1) = 0$).

The peak error rate, which always occurs at $S^* = 0$, decreases with increasing $\rho(1)$, approaching zero (i.e., no error) in the limit where the concentrations of bacteria in consecutive samples are perfectly correlated ($\rho(1) = 1$) (Figure 3). In the event that concentrations of bacteria in consecutive samples are negatively correlated ($\rho(1) < 0$), the peak error rate exceeds 50%, approaching 100% in the extreme limit where $\rho(1) = -1$. Even in cases where the concentrations of bacteria in consecutive samples are reasonably well-correlated, say $\rho(1) = 0.5$, relatively large peak posting error rates are predicted ($P^{\mathrm{T}}_{\mathrm{err}} = 0.35$).

To test the theory presented above, the fraction $F^{\mathrm{T}}_{\mathrm{err}}$ of samples that generated underprotection or overprotection posting errors was calculated for each of the Huntington Beach data bins described in the last section (see Supporting Information). Values of $F^{\mathrm{T}}_{\mathrm{err}}$ track closely the theoretical line for $\rho(1) = 0$ (i.e., the Bernoulli trial theory limit, compare data points with dashed line in Figure 3), consistent with the idea that the concentrations of ENT in consecutive samples at Huntington Beach are uncorrelated. To explore this issue further, for each data bin, we calculated the correlation coefficient between ENT concentrations in consecutive samples, referred to here as $\hat{\rho}(1)$ (see inset in Figure 3). Empirical correlation coefficients range from $\hat{\rho}(1) = 0.04$ to 0.58; averaging across all bins, we obtain $\overline{\hat{\rho}(1)} = 0.32 \pm 0.13$. The concentration of ENT in consecutive samples are not completely uncorrelated (i.e., $\hat{\rho}(1) \neq 0$); however, the sample-to-sample correlation is sufficiently weak such that total posting error rates are indistinguishable, within the resolution of our estimates of F^T_{err}, from the predictions of Bernoulli trial theory. An exception may be the three data bins with the highest correlation coefficients: station 6N in 1998, station 3N in 1999, and station 9N in 2000 (compare red points and dashed line in Figure 3).

Can Increasing the Sampling Frequency Reduce Posting Errors?

Question: *Would the sample-to-sample correlation be higher and the rate of single-sample posting errors be lower if surf zone samples were collected more frequently?*

Answer: *An analysis of ENT data at Huntington Beach reveals that posting decisions would have to be updated every 40 min (or more frequently) to significantly reduce posting errors. Even if posting decisions were revised every 10 minutes, when S^* is close to zero as much as 30% of the signage would be in error. This result will likely apply to any shoreline site where the sampling time interval is longer than the persistence time of pollution patches in the surf zone.*

The question is motivated by the growing interest in developing rapid fecal indicator bacteria tests that could, in principle, dramatically reduce the time between when a sample is taken and the bacterial indicator concentration is known (22). The answer derives from an analysis of autocorrelation functions computed from four different time series (Figure 4): (1) Routine ENT monitoring data at station 9N, subsampled to yield a sampling frequency of once per week ($\Delta t = 1$ week, panel A). (2) Routine ENT monitoring data at station 9N subsampled to yield a sampling frequency of once per 3 days ($\Delta t = 3$ days, panel B). (3) A special ENT monitoring study at station 3N in which water samples were collected every hour, 24 h per day, for 2 weeks ($\Delta t = 1$ h, panel C). (4) A second special ENT monitoring study at station 6N in which water samples were collected every 10 min for a total of 12 h ($\Delta t = 10$ min, panel D) (7). The autocorrelation functions in Figure 4 represent the correlation $\hat{\rho}(\Delta t_j)$ between a time series and itself after introducing a lag of j points or, equivalently, a time lag of $\Delta t_j = j\Delta t$. For comparison, also plotted in each panel of the figure are autocorrelation functions calculated from a sequence of random numbers ranging in magnitude from 1 to -1 (black lines in each panel).

Correlation $\hat{\rho}(\Delta t_j)$ falls off very rapidly with increasing lag Δt_j for sampling intervals of $\Delta t = 1$ week and 3 d (red curves in panels A and B, respectively). When $\Delta t = 1$ week (panel A), a broad peak is evident at time lags of 40–50 weeks (i.e., approximate 1 yr), presumably due to the influence of seasonal rainfall on bacterial concentrations in the surf zone. Spring–neap cycling of bacterial concentration is apparent in panel B where the correlation values peak every 2 weeks. Apart from the seasonal (panel A) and spring–neap (panel B) patterns, the correlation coefficients calculated for these two cases are generally within the range calculated from a sequence of random numbers (black lines). Correlation peaks are present at multiples of 24 h when the surf zone is sampled every hour ($\Delta t = 1$ h, panel C). This diurnal cycle probably arises from the germicidal affect of sunlight (7), although tidal processes may also play a role (e.g., during the summer at Huntington Beach there is typically just one large ebb tide per day). Remarkably, the sign of $\hat{\rho}(\Delta t_j)$ in Figure 4C is periodically negative, implying that posting error rates might *increase* if the sampling frequency is increased, for example, from once per day to once every 12 h (see peak error rates when $\rho(1) < 0$ in Figure 3). Compared to the other autocorrelation functions, $\hat{\rho}(\Delta t_j)$ decays with Δt_j more slowly when samples are collected every 10 min (panel D). Even in this case, however, the correlation coefficient for a lag of 10 min ($\hat{\rho}(\Delta t_1 = 10\text{min.}) = 0.6$) is such that substantial posting errors ($\approx 30\%$) are predicted when $S^* \approx 0$ (see Figure 3). Put another way, if the time interval between when a sample is taken and a sign is posted (or removed) was reduced to just 10 min, as much as 30% of the signage could be in error. The technology for rapid detection of fecal indicator bacteria is maturing such that near real-time measurements of these organisms may be feasible soon. Even if bacterial measurements could be carried out instantaneously (i.e., $\Delta t \to 0$ and $\rho(1) \to 1$), however, it is not clear how that information would be used in practice. Given the highly variable nature of the coastal water quality signal, health advisories would have to be updated on a minute-by-minute basis, creating an untenable

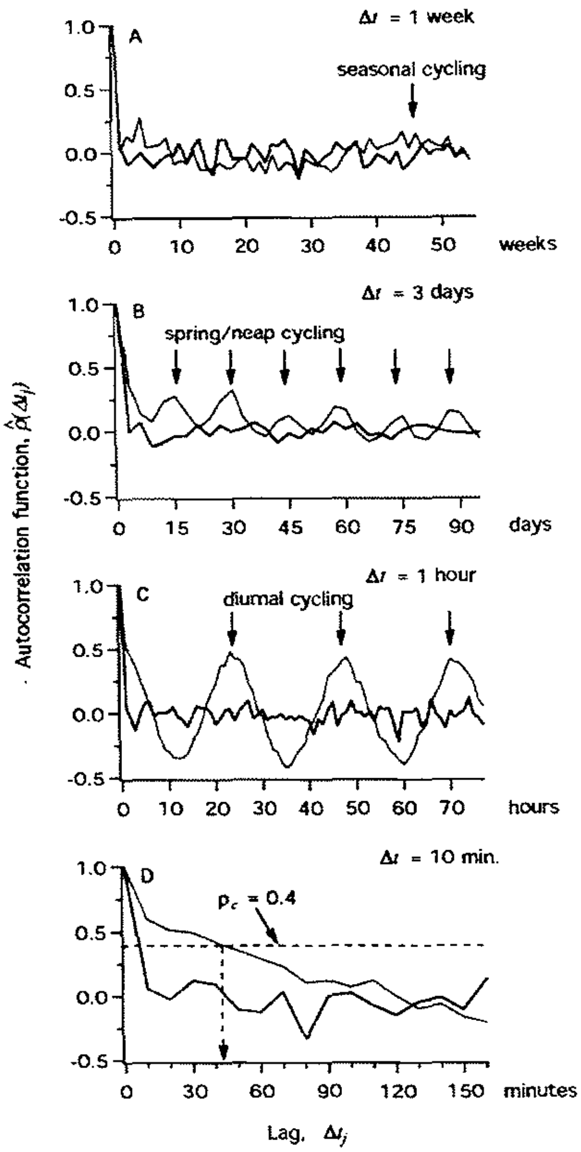

FIGURE 4. Autocorrelation functions calculated from four different time series (red lines in each panel) and from a sequence of uncorrelated (random) numbers (black lines in each panel). The time interval between samples (Δt) is noted in each panel (see text for details). The number of samples used for the analysis in each panel is $n = 216$ (panel A), 528 (panel B), 337 (panel C), and 61 (panel D).

situation for both local officials who issue health advisories and the beach-going public.

On the basis of the autocorrelation functions presented in Figure 4A,B and the correlation values calculated from ENT monitoring data at Huntington Beach (inset in Figure 3), it appears that $\hat{\rho}(1) \leq 0.4$ is typical of sampling frequencies in the once-per-day to once-per-week range. Because the posting error rates calculated from the daily to weekly monitoring data closely follow the predictions of Bernoulli trial theory (see Figure 3), it seems reasonable to adopt 0.4 as the critical correlation value above which Bernoulli trial theory begins to break down ($\rho_C \approx 0.4$). Referring to Figure 4D, the correlation $\hat{\rho}(\Delta t_j)$ falls below 0.4 for lag times greater than approximately 40 min (see dashed arrow in figure). This lag is very close to the time it takes tidally generated patches of fecal pollution in the Huntington Beach surf zone to advect past a fixed location by wave-driven long-shore currents ($\tau = L/u = 50$ min, where $L = 1$ km is the approximate length of an average pollution patch and $u = 0.3$ m/s is a typical long-shore advective velocity) (7, 8). Therefore, the concentration of bacteria in a water sample from the surf zone appears to have little memory of previous samples (and hence Bernoulli trial theory applies) so long as the time interval between samples is longer than the persistence time of pollution patches. While the persistence time scale will vary by site, given the highly dynamic nature of ocean currents at most coastal sites it is unlikely that the persistence time scale will exceed 1 d, the sampling frequency of the most aggressive shoreline monitoring programs. Hence, the large posting error rates reported in this paper are probably not unique to Huntington Beach. Rather, large errors can be expected at any marine or freshwater beach when $|S^*| < 1$ and the time interval between samples is greater than the persistence time scale for patches of contaminated water.

Toward an Analog Public Health Advisory System

Question: *Can less error-prone approaches be developed for assessing current water quality and reporting that information to the public?*

Answer: *Several different approaches can be adopted to predict (or "now-cast") current coastal water quality and report that information to the general public. At Huntington Beach, the current concentration of ENT at a particular surf zone station is generally more correlated with the maximum daily tide range, than with the concentration of bacteria in the last sample.*

An approach that follows naturally from the probability theory presented above involves computing analog (i.e., continuously varying) estimates of current water quality and/or human health risk, periodically updated as new information becomes available. Predictions of current water quality, or now-casts, could utilize a variety of data resources including recent water quality test results and real-time (or near-real-time) measurements of quantities known to correlate with local surf zone water quality. Now-casts, in turn, could be conveyed to the public through a combination of web sites, newspaper reports, and/or beach signage either in raw form or using a grading scale like that employed by Heal the Bay (23).

As an example, below we present a prototypical algorithm that now-casts three different measures of water quality. For the sake of simplicity and to demonstrate the power of even a modest algorithm, our prototype requires only estimates of tide range (predicted from WXTide32 (24)) and water quality measurements collected over the previous 30 d. At the heart of the algorithm is the assumption that current water quality is conditioned on maximum daily tide range, as expressed quantitatively through the conditional probability density function, $f_{\log C|L}(\log c|\ell)$. Here, log C and L represent random variables for the log-transformed bacterial concentration and maximum daily tide range, respectively, and log c and ℓ are specific realizations of the random variables. For a fixed value of the tide range ℓ, the probability of single-sample exceedence and expected value of the bacterial concentration can be calculated as follows:

$$P_{\text{ex}|\ell} \equiv P[\log C > \log s | L = \ell] = \int_{\log s}^{\infty} f_{\log C|L}(x|\ell)\, dx \quad (4a)$$

$$\mu_{\log C|\ell} \equiv E[\log C | L = \ell] = \int_{-\infty}^{\infty} x f_{\log C|L}(x|\ell)\, dx \quad (4b)$$

An analysis of maximum daily tide-range predicted for Huntington Beach reveals that L conforms reasonably well to a normal distribution (based on Kolmogorov–Smirnov normality tests; maximum difference $K - S = 0.04$ at the significant level $\alpha < 0.01$) as do the log-transformed concentrations of fecal indicator bacteria (see earlier). This implies that eqs 4a,b can be written explicitly as follows (see Supporting Information):

$$P_{\text{ex}|\ell}(S^*, \ell^*, \rho_{\text{CL}}) = \frac{1}{2}\text{erfc}[(S^* - \rho_{\text{CL}}\ell^*)/\sqrt{1 - \rho_{\text{CL}}^2}] \quad (5a)$$

$$\mu_{\log C|\ell} = \mu_{\log C} + (\rho_{\text{CL}}\sigma_{\log C}/\sigma_L)(\ell - \mu_L) \quad (5b)$$

$$\sigma_{\log C|L} = \sigma_{\log C}\sqrt{1 - \rho_{\text{CL}}^2} \quad (5c)$$

where $\ell^* = (\ell - \mu_L)/(\sqrt{2}\sigma_L)$, ρ_{CL} is the correlation coefficient between log C and L, μ_L and σ_L are the mean and standard deviation of the maximum daily tide range, and the other parameters have been defined previously. Equation 5b is a linear model for the dependence of $\mu_{\log C|\ell}$ on ℓ, with coefficients equivalent to those obtained by a mean-square regression of $\mu_{\log C|\ell}$ against ℓ (25). To develop an expression for bather illness rate, we utilized the linear relationship for gastrointestinal illness rate per 1000 bathers (Y) reported by Cabelli et al. (9): $Y = a + b \log \text{GM}_C$, where GM_C represents the geometric mean of ENT measurements, and $a = -5.1$ and $b = 24.2$ are empirical constants. Rewriting Cabelli et al.'s model in terms of the conditional log-mean $\mu_{\log C|\ell}$ we obtain:

$$Y = a + b\mu_{\log C|\ell} \quad (6a)$$

$$\sigma_Y = \sqrt{{\sigma_a}^2 + \sigma_b^2{\mu_{\log C|\ell}}^2 + b^2{\sigma_{\log C|L}}^2} \quad (6b)$$

The estimate of uncertainty in Y (eq 6b) was derived by propagating uncertainties in the variables on the RHS of eq 6a (17). This set of expressions (eqs 5a–c and 6a,b) were employed to now-cast water quality at surf zone station 6N over the period May 1–October 31, 1999 (i.e., the same period of time encompassed by Figure 1A). Now-casts for a particular day were generated using the current day's maximum tide range ℓ (estimated from WXTide32) and updated estimates for $\hat{\mu}_{\log C}$, $\hat{\sigma}_{\log C}$, $\hat{\mu}_L$, $\hat{\sigma}_L$, and $\hat{\rho}_{\text{CL}}$ calculated from daily tide range and ENT measurements collected over the previous 30 d (see Supporting Information). Values for $\hat{\sigma}_a = 6.35$ and $\hat{\sigma}_b = 4.15$ were estimated from data reported in Cabelli et al. (9).

Now-casts of the ENT concentration at 6N correctly capture the magnitude and spring–neap cycling of actual ENT measurements (compare red and blue curves, second panel of Figure 5). Over the 6-month period, 56% and 92% of the ENT measurements at 6N fell in the predicted range $\hat{\mu}_{\log C|\ell} \pm 1\hat{\sigma}_{\log C|L}$ (blue band in second panel) and $\pm 2\hat{\sigma}_{\log C|L}$, respectively. Now casts of illness rate (third panel) are generally above the threshold level of 19 in 1000 (dashed

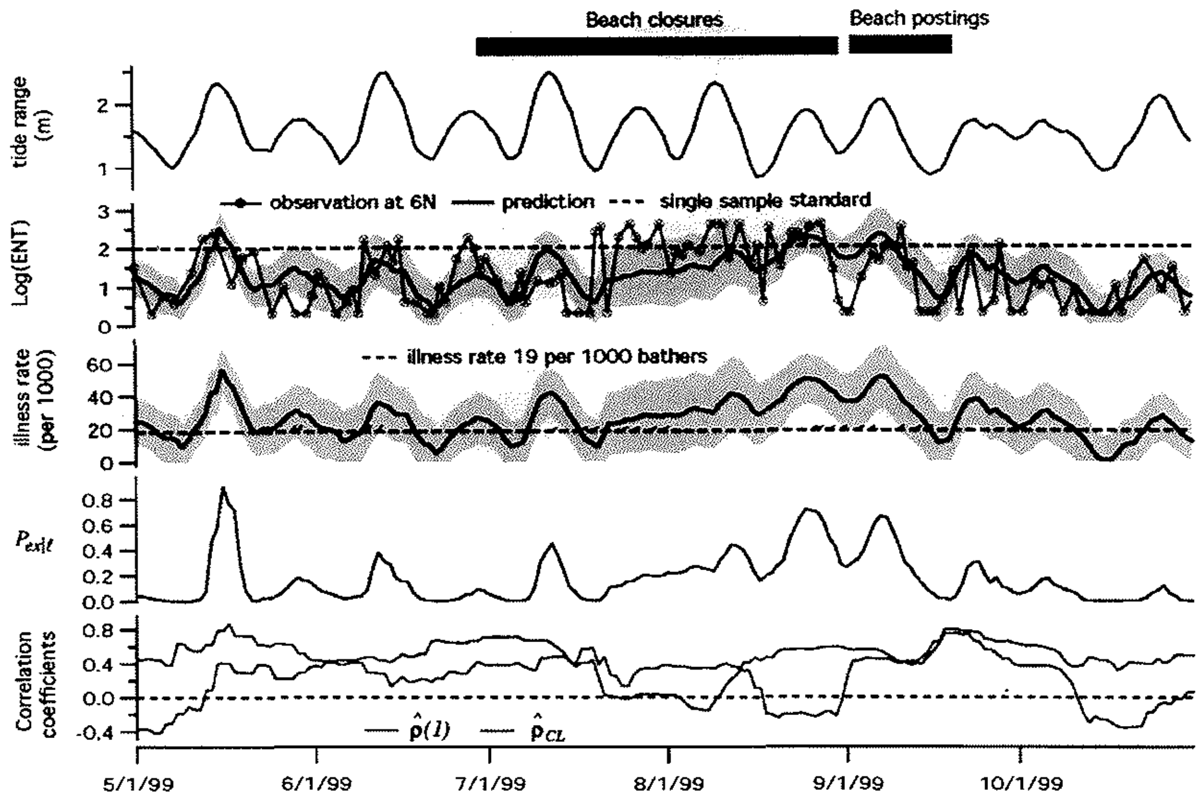

FIGURE 5. Time series plots of maximum daily tide range (top panel), now-casts of ENT concentration (blue line, second panel), illness rates (blue line, third panel), and probability of exceeding the single-sample standard for ENT (black line, fourth panel) at surf zone station 6N in Huntington Beach. Blue bands in second and third panels represent ±1 SD (see eqs 5c and 6b); red curve in second panel is actual measurements of ENT at 6N. Bottom panel is a plot of the correlation coefficients between ENT in consecutive samples ($\hat{\rho}(1)$) and between ENT and maximum daily tide range ($\hat{\rho}_{CL}$).

horizontal line), peaking at about 60 excess illnesses per 1000 bathers; these illness attack rates are in the range estimated by other researchers for the Huntington Beach area (*26*). Now-casts for the exceedence probability (fourth panel) exhibit spring–neap cycling, and importantly, periods of high exceedence probability (e.g., $P_{ex|\ell} > 0.2$) generally coincide with single-sample violations (compare second and fourth panels, Figure 5). The last panel in the figure is a plot of the correlation coefficient between ENT concentrations in consecutive samples ($\hat{\rho}(1)$, red line) and between ENT concentration and maximum daily tide range ($\hat{\rho}_{CL}$, blue line). These two coefficients were updated every day in the 6-month period encompassed by Figure 5, using ENT and maximum daily tide range data collected (ENT) or calculated (tide range) over the previous 30 d. In general, $\hat{\rho}_{CL}$ is larger than 0.4 (i.e., $\hat{\rho}_{CL} > \rho_C$, see earlier) and larger than the correlation between the concentrations of bacteria in consecutive samples ($\hat{\rho}_{CL} \geq \hat{\rho}(1)$). The exception is an approximately 1-month period, centered around August 1, when $\hat{\rho}_{CL} \approx 0$. Not surprisingly, this was also the period when our now-cast model performed least well. In general, at Huntington Beach, the current concentration of bacteria at a particular surf zone station is more correlated with the maximum daily tide range than with the concentration of bacteria in the last sample.

The model presented above could be improved by utilizing all physical variables known through past experience to correlate with coastal water quality and/or by adopting alternative now-casting methodologies (e.g., artificial neural networks (*27–29*)) that tolerate nonlinear relationships between dependent and independent variables. As mentioned above, the ideal advisory system will be analog in nature; however, even if the current binary approach is retained, posting decisions based on now-cast methodologies, like the one described here, would be an improvement over the status quo. If the now-casts of ENT presented in Figure 5 had been used as the basis for posting decisions at Huntington Beach during the summer of 1999, for example, the total posting error rate there would have been reduced between 7.5% and 50% (depending on the particular surf zone station of interest).

Acknowledgments

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Supporting Information Available

Mathematical derivations and additional data. This material is available free of charge via the Internet at http://pubs.acs.org.

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ES034382V

National Center For Environmental Research

Final Report: Identification and Control of Non-Point Sources of Microbial Pollution in a Coastal Watershed

NCER Research Project Search

EPA Grant Number: R828011
Title: Identification and Control of Non-Point Sources of Microbial Pollution in a Coastal Watershed
Investigators: Sanders, Brett , Grant, Stanley B. , Horne, Alex , Keller, Robin , Sobsey, Mark D.
Institution: **University of California - Irvine** , University of California - Berkeley , University of North Carolina at Chapel Hill
EPA Project Officer: Stelz, Bill
Project Period: August 1, 2000 through July 31, 2003 (Extended to January 31, 2005)
Project Amount: $895,234
RFA: Water and Watersheds (1999)
Research Category: Water and Watersheds

Description:

Objective:

The objectives of this study were to: (1) characterize the magnitude and variability of fecal indicator bacteria (FIB) loads in the watershed along an inland to coastal gradient that includes street gutters, storm channels, tidal channels, and the surf-zone at Huntington Beach; (2) examine linkages between FIB and other indicators of human pathogens; (3) develop strategies to control FIB loads during nonstorm periods; and (4) aid decisionmaking by examining the perspectives of stakeholders, including beachgoers, environmentalists, local businesses, public health officials, and wastewater utility managers on various aspects of beach pollution problems, such as the causes, health risks, and responsibility to pay.

California beaches are a critical component of the culture and economy of California and are threatened by coastal pollution. Beach recreation in California accounts for $5.5 billion of the Gross State Product (King and Symes, 2003). Nowhere has there been greater attention on beach pollution than at Huntington Beach in southern California.

Huntington Beach, consisting of Huntington State Beach and Huntington City Beach, is located along a northwest to southeast striking section of the Pacific coastline between Los Angeles and San Diego, in Orange County, California. Several areas of Huntington State Beach have suffered chronic beach postings and closures over the past several years as a result of elevated concentrations of FIB in the surf zone (Kim and Grant, 2004). This beach is very popular (more than 5 million visitors per year), and the combination of surf zone pollution and significant beach usage implies that a large number of people (perhaps as many as 50,000) may acquire highly credible gastroenteritis from swimming and surfing in this area each year (Turbow, et al., 2003). FIB pollution at Huntington State Beach is thought to be caused by a combination of sources, including dry and wet weather runoff from the surrounding community, bird droppings deposited in the Talbert Marsh, and regrowth of bacteria on vegetation and marsh sediments (Grant, et al., 2001; Reeves, et al., 2004). Additional potential sources of FIB include the offshore discharge of partially treated sewage effluent (Boehm, et al., 2002a), the offshore discharge of power plant cooling water that contains FIB from plant wash-down and other activities (Boehm, et al., 2002b), resuspension of contaminated sediments (Sanders, et al., 2004), bather shedding, the accumulation of bird droppings along the shoreline and offshore, the exfiltration of sewage-contaminated groundwater, and contributions from watershed outlets located north and south of the study area, including the Los Angeles River, the San Gabriel River, and outlets for Huntington Harbor and Newport Bay (Kim, et al., 2004).

This project focuses on the Talbert Watershed in Huntington Beach and Fountain Valley, California, which drains to Huntington Beach and is a significant stressor of Huntington Beach water quality. The Talbert Watershed encompasses 3,400 hectares in the cities of Huntington Beach and Fountain Valley. The watershed is urbanized and consists of residential developments, commercial districts, plant nurseries, and light industry. This area of southern California has separate storm water and sanitary sewer systems, therefore, dry and wet weather runoff flows to the ocean without treatment. Runoff from the Talbert Watershed is conveyed along street gutters to inlets that connect to underground storm water pipelines. These pipelines connect to a network of three flood control channels (Fountain Valley, Talbert, and Huntington Beach) that converge near the ocean at a constructed wetland known as the Talbert Marsh. Ocean water floods both the Talbert Marsh and the lower reaches of the open channels during rising tides (flood tides), and a brackish mixture of ocean water and runoff drains from the system during falling tides (ebb tides). The Talbert Watershed is nearly flat and only a few feet above sea level. This geographical setting hinders drainage by gravity alone, so a system of transfer stations is used in the lower reaches of the Talbert Watershed to pump runoff into the open channels from storm water pipelines. Each transfer station, or pump station, consists of a forebay, where runoff can be stored, and several pumps. Pumping of runoff to the channels occurs intermittently during dry weather periods and continuously during storms. Talbert Marsh is a 10-hectare remnant of what used to be an extensive (1,200 hectare) saltwater wetland and dune system in coastal Orange County. The majority of this wetland system was drained and filled over the past century for agricultural reclamation and urban development. Most of what remained of the historical wetland, including Talbert Marsh, was cut off from tidal flushing by the construction of the Pacific Coast Highway and channelization of the surrounding area for flood control. As part of a habitat restoration effort, tidal flushing in the Talbert Marsh was restored in 1990 when a new tidal inlet was constructed. Since its restoration, Talbert Marsh has become a typical southern California tidal saltwater marsh with open water, wetland, and upland habitats (Grant, et al., 2001). Pickle weed (*Salicornia virginica*) is the dominant macrophytic vegetation, and the marsh is utilized by several special-status bird species, including the California least tern, brown pelican, and Beldings savannah sparrow.

Summary/Accomplishments (Outputs/Outcomes):

To achieve the objectives, extensive monitoring of Talbert Watershed surface waters was conducted to measure the spatio-temporal variability of FIB loads (total coliform, *Escherichia coli*, and *Enterococcus*) and analysis was performed to examine the factors that control fate and transport. Monitoring also was performed to examine the association between FIB and other indicators of fecal pollution. Both one-dimensional and two-dimensional hydrodynamic models were developed to analyze the FIB loads in tidal channels and into the surf-zone and to develop a predictive tool that can be used to examine how bacteria loads would be altered by operational changes to the infrastructure. Surveys were performed to measure stakeholder preferences in the context of multistakeholder, multi-objective beach pollution problems and to support decisionmaking analysis.

Closure and posting of Huntington Beach, California, during the study period was the source of widespread media attention. In response, members of the research team redirected efforts and/or engaged in a number of additional studies to better understand the factors controlling surface water quality in the Huntington Beach surf zone, as well as the response of stakeholders to the unfolding pollution problem. For example, co-principal investigator (PI) Keller focused attention on the decisionmaking of beachgoers (to swim or not to swim) in response to warning signs posted on the beach. Co-PI Keller also focused attention on the decisionmaking of public agencies, who were under great public pressure to remedy the pollution problem but had little understanding of its cause. To better understand the pollution problem, co-PI Grant analyzed short- and long-term FIB monitoring data to identify trends in Huntington Beach bathing water quality. The observed variability was examined in the context of historical management measures, such as passage of the Clean Water Act, construction of a new ocean outfall, and efforts to prevent urban runoff from draining directly to the beach. Co-PI Grant also developed a method to identify and rank the sources of pollution to the surf-zone using high-frequency monitoring data collected along the beach. PI Sanders teamed with University of California (UC) Irvine and UC San Diego researchers to examine the potential for Orange County Sanitation District effluent, discharged roughly 7 km offshore of Huntington Beach, to be transported onshore by internal tides. After the Talbert Marsh was identified as a contributor of FIB to the Huntington Beach surf zone, co-PI Sobsey focused attention on potential health risks associated with water contaminated with bird feces. In particular, marsh bird feces and surface water was examined for *Campylobacter*, *Salmonella,* and male-specific coliphages.

During dry weather, concentrations of FIB were highest in inland urban runoff, intermediate in tidal channels harboring variable mixtures of urban runoff and ocean water, and lowest in ocean water at the base of the watershed. This inland-to-coastal gradient is consistent with the hypothesis that urban runoff from the watershed contributes to coastal pollution. On a year-round basis, the vast majority (> 99%) of FIB loading occurs during storm events when runoff diversions, the management approach of choice, are not operating. During storms, the load of FIB in runoff follows a power law of the form $L \sim Q^n$, where L is the loading rate (in units of FIB per time), Q is the volumetric flow rate (in units of volume per time), and the exponent n ranges from 1 to 1.5. This power law and the observed range of exponent values are consistent with the predictions of a mathematical model that assumes FIB in storm runoff originate from the erosion of contaminated sediments in drainage channels or storm sewers. (Reeves, et al., 2004)

During dry weather periods, urban runoff controls surface water concentrations of FIB in channels where flushing is weak, and resuspension of FIB from the sediment/water interface controls surface water concentrations near the mouth where flushing by ocean water occurs once per day. The reservoir of FIB at the sediment/water interface is probably linked to settling of bacteria from both dry and wet weather urban runoff, deposition of animal feces, decaying vegetation, and bacterial regrowth. It is not clear whether the FIB are primarily attached to sediments, suspended in pore water, or incorporated into microbial biofilms. Nevertheless, surface water concentrations of FIB are rapidly amplified as turbulence in water column increases. A result is that dry weather urban runoff has little direct impact on surf zone water quality, but significant indirect impact given FIB loads from runoff accumulate at the sediment/water interface and are subsequently resuspended and exported to the surf-zone by tidal currents (Grant, et al., 2001; Arega and Sanders, 2004; Sanders, et al., 2004).

During the project period, dry-weather diversions of urban runoff to the sanitary sewer system were implemented to mitigate impacts to the surf-zone, at a cost of at least $6 million to the County of

Orange and City of Huntington Beach. The efficacy of this approach is unclear, because the vast majority of watershed loads are shed during wet weather, whereas during dry weather, the tidal channels and marsh serve to dissipate loads by promoting die-off and settling. On the other hand, diversions presumably serve to reduce loads of other contaminants, including oil, grease, heavy metals, and so forth and, therefore, may be justified on these grounds. To evaluate whether the diversions are justified on the basis of FIB control, a better understanding of the cycling of FIB in sediments is needed. The alternative is to focus management efforts on wet weather controls. For example, if erosion of sediments is driving the loading of FIB, then regular removal of contaminated sediments accumulating in the storm sewer system might be an appropriate management strategy. The creation of distributed wetland treatment systems, in which contaminants in urban runoff are removed near their source, might also prove useful for reducing downstream impacts (Reeves, et al., 2004).

Research lead by PI Sanders shows that numerical modeling can be performed to predict FIB loads in tidal wetlands, analytes that are notoriously difficult to model because of poorly characterized non-conservative processes. The key parameters needed for accurate predictions of FIB loads, using a validated hydrodynamic model, are: (1) the load as a result of urban runoff; (2) sediment erodibility parameters; and (3) sediment concentrations and surface water die-off rates of enteric bacteria. For channels in the Talbert Watershed, literature values for sediment erodibility and water column die-off rates were used and average concentrations of indicator bacteria were predicted within one-half log unit of measurements. Total coliform were predicted more accurately than *E. coli* or enterococci, both in terms of magnitude and tidal variability. This work is important because it represents the first case where first-principle models were successfully applied to predict FIB in an estuarine setting with significant nonpoint sources. The approach adopted here is highly transferable and could benefit both wetland restoration and water quality compliance efforts on a widespread basis (Sanders, et al., 2004).

Plume tracking studies conducted by UC Irvine and UC San Diego researchers, including PI Sanders, show that Orange County Sanitation Department (OCSD) effluent occasionally moves shoreward toward Huntington Beach into water less than 20 m deep. Analyses of current and temperature observations indicate cold water is regularly advected crossshelf, into and out of the nearshore, at both semi-diurnal and diurnal frequencies. Isotherms typically associated with the wastefield near the outfall are observed just outside the Huntington Beach surf zone, where the total depth is less than 6 m, highlighting the extent of the cross-shelf transport. This advection is attributed to a mode 1 internal motion, or internal tide. Based on this analysis, it is not possible to rule out the possibility that the OCSD plume contributes to poor bathing-water quality at Huntington Beach (Boehm, et al., 2002a). Concerned over potential shoreline impacts, OCSD began a disinfection program in 2002 and initiated a roughly $300 million program to build the necessary infrastructure for full secondary treatment.

Analysis of Huntington Beach monitoring data lead by co-PI Grant shows that the concentration of FIB varies over time scales that span at least seven orders of magnitude, from minutes to decades. Sources of this variability include historical changes in the treatment and disposal of wastewater and dry weather runoff, El Niño events, seasonal variations in rainfall, spring-neap tidal cycles, sunlight-induced mortality of bacteria, and nearshore mixing. On average, total coliform concentrations have decreased over the past 43 years, although point sources of shoreline contamination (storm drains, river outlets, and submarine outfalls) continue to cause transiently poor water quality. These transient point sources typically persist for 5 to 8 years and are modulated by the phase of the moon, reflecting the influence of tides on the sourcing and transport of pollutants in the coastal ocean. Indicator bacteria are very sensitive to sunlight; therefore, the time of day when samples are collected can influence the outcome of water quality testing. These results demonstrate that coastal water quality is forced by a complex combination of local and external processes and raise questions about the efficacy of existing marine bathing water monitoring and reporting programs (Boehm, et al., 2002b). Further analysis led by co-PI Grant reveals that protocols used to decide whether to post a sign are prone to error. Errors in public notification (referred to here as posting errors) originate from the variable character of pollutant concentrations in the ocean, the relatively infrequent sampling schedule adopted by most monitoring programs (daily to weekly), and the intrinsic error associated with binary advisories in which the public is either warned or not. We derived a probabilistic framework for estimating posting error rates, which at Huntington Beach range from 0 to 41 percent, and show that relatively high sample-to-sample correlations (> 0.4) are required to significantly reduce binary advisory posting errors. Public misnotification of coastal water quality can be reduced by utilizing probabilistic approaches for predicting current coastal water quality, and adopting analog, instead of binary, warning systems (Kim and Grant, 2004).

Research lead by co-PI Sobsey on the potential health risks of bathing water contaminated by bird feces has lead to only preliminary findings. Specifically, *Campylobacter* and male specific coliphages were identified in Talbert Marsh bird feces and in marsh surface waters near the marsh. *Salmonella* was found only in bird feces samples and not water samples. Analysis continues to understand the relationship between microbes in bird feces and surrounding surface waters, and potential health impacts.

Research lead by co-PI Keller indicates that stakeholders share diverse opinions about the causes of beach pollution, the risks to beachgoers, and the responsibility to pay. In the context of a multi-objective decision model, stakeholders disagree on the appropriate weights of objectives. For example, local businesses heavily weigh economics whereas beachgoers heavily weigh health risks. Stakeholders also disagree on the severity of pollution problems. For example, environmentalists believe the probability of an environmental health problem is high when beaches are posted, but beachgoers do not. Relative to beachgoers' perceptions of potential health risks, surveys showed a peer effect: decisions to enter the water at posted beaches were strongly affected by whether or not others were in the water (Biswas and Keller, 2004; Biswas, et al., 2004).

Conclusions:

The vast majority of FIB loads in runoff from the Talbert Watershed are shed during storms and are associated with particles that appear to be scoured from the water collection system, including street gutters, storm pipes, and storm channels. Loads in runoff during dry weather periods account for roughly 1 percent of the annual runoff load and dissipate within the tidal channels by a combination of die-off and settling.

Loads exported from the watershed to the surf zone during dry weather period are deflected along the shoreline by wave driven currents and can cause exceedances of water contact recreation standards. Model predictions show the origin of such loads is the scouring by tidal currents of FIB at the sediment/water interface of tidal channels and Talbert Marsh. FIB at the sediment/water interface are linked to urban runoff FIB loads during both dry and wet weather periods, bird droppings, decaying vegetation, and bacterial regrowth. Because intertidal wetlands are to some extent natural generators of FIB, these results call into question the exclusive use of FIB as the basis of water contact recreation standards at beaches near the outlet of these water bodies.

On the basis of FIB control, the efficacy of dry weather diversions in Talbert Watershed is unclear, although diversions presumably serve to mitigate other types of pollution as well. A better understanding of the cycling of FIB between the water column and sediments is needed to evaluate the linkages between wet weather and dry weather loads in relation to sediment interactions.

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Journal Article	Boehm AB, Sanders BF, Winant CD. Cross-shelf transport at Huntington Beach. Implications for the fate of sewage discharged through an offshore ocean outfall. *Environmental Science & Technology* 2002;36(9):1899-1906	R828011 (2001) R828011 (Final)	• Full-text: ACS Publications Full Text EXIT Disclaimer • Other: ACS Publications PDF EXIT Disclaimer
Journal Article	Grant SB, Sanders BF, Boehm AB, Redman JA, et al. Generation of enterococci bacteria in a coastal saltwater marsh and its impact on surf zone water quality. *Environmental Science and Technology* 2001;35(12):2407-2416.	R828011 (2000) R828011 (2001) R828011 (Final)	Not available
Journal Article	Reeves RL, Grant SB, Mrse RD, Copil-Oancea CM. Scaling and management of fecal indicator bacteria in runoff from a coastal urban watershed in southern California. *Environmental Science & Technology* 2004;38(9):2637-2648.	R828011 (Final)	• Full-text: ACS Full Text EXIT Disclaimer • Other: ACS PDF EXIT Disclaimer
Journal Article	Sanders BF, Arega F, Sutula M. Modeling the dry-weather tidal cycling of fecal indicator bacteria in surface waters of an intertidal wetland. *Water Research*. 2005;39(14):3394-3408.	R828011 (Final)	

Supplemental Keywords:

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Enviro Science & Technology

Bird Droppings Are Blamed for Bacteria

By Stanley Allison June 02, 2001 in print edition B-9

A team of UC Irvine researchers has concluded that waterfowl and other animal droppings from a saltwater marsh and the Santa Ana River are a significant source of bacteria contaminating the ocean waters off Huntington Beach.

In a report that will be published in the June 15 issue of Environmental Science and Technology, the researchers point to inherent flaws in the design of the man-made saltwater Talbert Marsh.

Stanley Grant, the UCI professor who led the 18-month study of the ocean contamination problem at Huntington Beach, said water containing fecal bacteria, pesticides, nutrients and other materials filters through the marsh and then flows into the ocean in about 40 minutes–which is too fast.

For the marsh to act as a natural cleanser and remove contaminants, the water must spend at least a week filtering through the wildlife preserve, Grant said.

Even though other sources such as urban runoff from the Santa Ana River may have contributed to the contamination that resulted in four miles of beach closures for most of the summer of 1999, the levels of bacteria from the marsh were hundreds of times more than the state limits, the researchers said.

The team's conclusions contradict the accepted environmental theory that wetlands purify contaminated water flowing into the ocean.

The findings suggest that approximately 4.6 million saltwater marshes in the U.S. could be similarly affected, Grant said.

Mark Gold, a spokesman for the conservation group Heal the Bay, said that finding animal droppings in a nature preserve is nothing new, and insists that marshes still serve as a cleanser for other, more hazardous, contaminates.

"It's not surprising that wetlands are sources of fecal bacteria," Gold said. "What wetlands are great at doing is removing nutrients and metals."

The 25-acre wetlands preserve is on the inland side of Pacific Coast Highway at Brookhurst Street. Part of the Talbert watershed that encompasses 12 square miles in Huntington Beach and Fountain Valley, it attracts thousands of migratory birds and other wildlife each year.

The UCI researchers also say that the nearby AES power plant contributes to the shore's contamination. The study suggests that partly treated sewage released four miles offshore from the Orange County Sanitation District treatment plant is being pulled back to the shore by tides and the plant as it draws water to cool its towers.

O-10

SUPPORTING SOUND MANAGEMENT OF OUR COASTS AND SEAS

Bacterial Contamination at Huntington Beach, California—Is It From a Local Offshore Wastewater Outfall?

During the summers of 1999 and 2000, beaches at Huntington Beach, California, were repeatedly closed to swimming because of high bacteria levels in the surf zone. The city's beaches are a major recreational and commercial resource, normally attracting millions of visitors each summer. One possible source of the bacterial contamination was the Orange County Sanitation District's sewage outfall, which discharges treated wastewater 4.5 miles offshore at a depth of 200 feet. Scientists from the U.S. Geological Survey and cooperating organizations have been investigating whether [illegible]


In most years, more than 5 million people visit the wide sandy beaches at Huntington Beach, California. Unfortunately, in recent summers, stretches of these beaches had to be repeatedly posted with advisories against swimming or even closed (inset), because levels of live bacteria in the surf zone exceeded State public-health standards. These closures resulted in serious economic losses to local business communities.

The wide sandy beaches at Huntington Beach, California, just south of Los Angeles, attract residents and visitors alike. Typically, more than 5 million people visit these beaches each summer, helping to support a regional tourism industry of $80 million annually.

During the summers of 1999 and 2000, stretches of these beaches had to be repeatedly closed to swimming or posted with advisories against swimming, because levels of live bacteria in the surf zone exceeded beach sanitation standards in the California Health and Safety Code (Assembly Bill 411, or AB411). Because people stayed away from the beaches, local recreational and beachfront business communities suffered serious economic losses.

Local agencies conducted a variety of studies in 1999 and 2000 to try to determine the cause of the beach contamination at Huntington Beach. They investigated known sources of bacteria, such as broken sewer pipes, outflow from the Santa Ana River, feces of bird populations in coastal marshes, and the plume of treated wastewater from the Orange County Sanitation District's (OCSD) ocean outfall, 4.5 miles (7 km) offshore at a depth of 200 feet (60 m). The beach closures were caused by elevated levels of three categories of bacteria—total coliform, fecal coliform, and enterococci. These bacteria, which live in the digestive tracts of warm-blooded animals including humans, are also found in the treated effluent discharged from the OCSD outfall. Because of this, it was suspected that coastal ocean processes might be bringing bacteria-rich effluent from the ocean outfall to shore.

To evaluate whether the OCSD outfall could be the source for the bacterial contamination at Huntington Beach, scientists from the U.S. Geological Survey (USGS) and cooperating organizations

This photo shows many enterococci bacteria (magnified more than 5,000 times), one type of bacteria indicative of contamination from feces of humans or other warm-blooded animals. High levels of enterococci have been responsible for many of the beach closures at Huntington Beach. (Photo courtesy Centers for Disease Control and Prevention.)

designed and carried out an extensive study. Begun in the summer of 2001, this study focused on the area's coastal ocean circulation and transport pathways.

It was known from the 1999–2000 studies done by local agencies that the beaches were most often contaminated during "spring" tides in the 2-week tidal cycle. Spring tides occur when the gravitational pulls of the Moon and Sun reinforce each other, resulting in the highest high waters and lowest low waters of this cycle. Additionally, previous field observations and theoretical modeling indicated that, in summer, the effluent plume from the OCSD outfall remains trapped below the thermocline, a zone of rapid change in temperature that divides ocean water into cold dense water below and warmer, less dense water above. In the ocean off southern California, the thermocline is typically about 50 to 65 feet (15 to 20 m) below the sea surface during the summer.

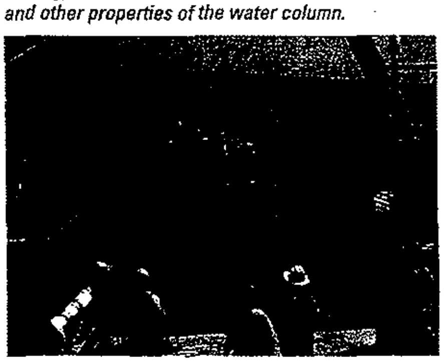
In light of these earlier findings, the USGS-led study focused on coastal ocean processes, such as tides, daily sea breezes, upwelling, and vertical mixing, that could move significant volumes of bacteria-laden OCSD plume water from offshore below the thermocline into the nearshore region and surf zone during summer months. In the summer of 2001, scientists deployed a sophisticated set of oceanographic instruments at 12 mooring sites in the coastal ocean off Huntington Beach and Newport Beach to monitor current velocity, temperature, and salinity at selected depths in the water column every few minutes for 4 months. Other instruments at these sites collected real-time meteorological data at the sea surface. Additional instruments were deployed in very shallow water to monitor the transport pathways between nearshore waters and the surf zone. Surf-zone water samples were collected 5 days a week in the early morning hours to measure bacterial levels from Huntington Beach to Newport Beach.

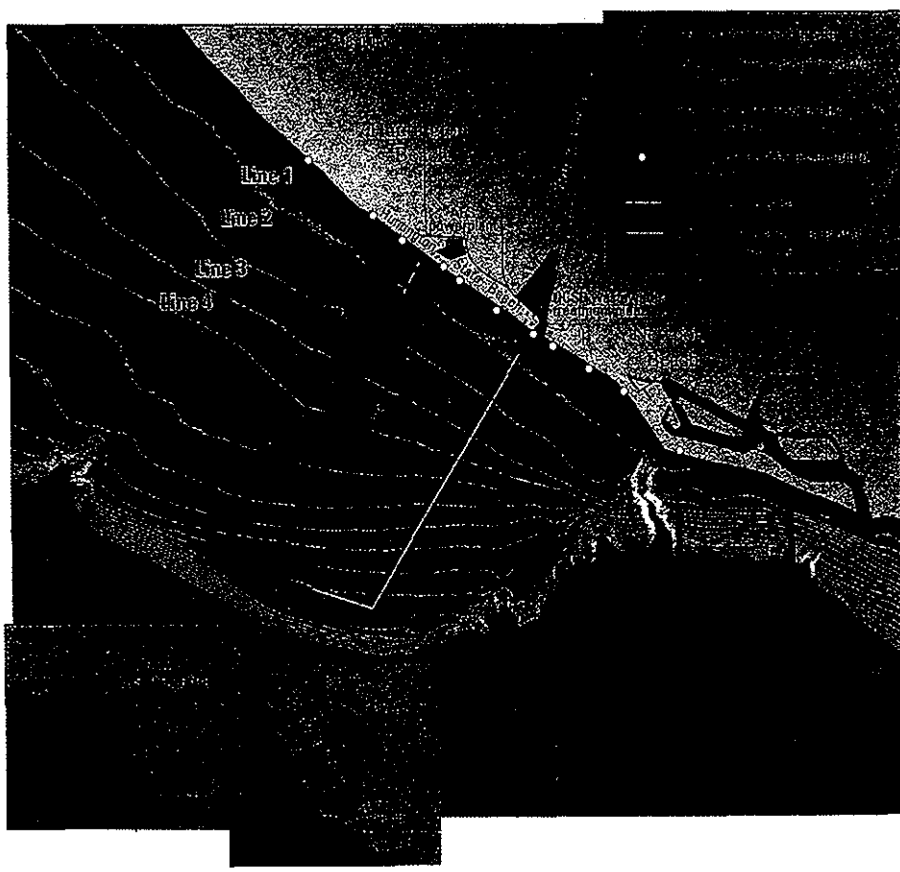
A complementary hydrographic mapping program used arrays of instruments towed or lowered from boats along 10 tow lines and at 40 sites between these lines and the shore during six surveys centered around periods of maximum tidal range (spring tides). The surveys measured the spatial distribution of temperature, salinity, ammonia content, bacteria concentrations, and other properties of the water column. These properties were chosen in part because they could be used to identify and track the relatively low-salinity and ammonia-rich effluent from the OCSD plume. During these surveys, additional surf-zone samples were collected every hour at 11 sites along local beaches to provide additional data to evaluate the spatial and temporal distribution patterns of bacteria from offshore to the surf zone.

Analysis of the data collected in 2001 show that the bacterial beach contamination at Huntington Beach is closely associated with both tidal cycles and time of day. High values of all three categories of fecal indicator bacteria were found preferentially at times of spring tides. Additionally, values were high mainly at night, particularly for enterococci. The data show that bacteria levels in the surf zone decreased to very low levels during sunlit hours, even when beaches were closed or posted for several days. However, previous studies have shown that the categories of bacteria found in the OCSD outfall plume can survive for several days in the deeper, colder water below the thermocline, where they are sheltered from ultraviolet light.

When it enters the ocean, the treated wastewater from the OCSD outfall rises toward the thermocline, because it is fresher, warmer, and therefore less dense than the surrounding ocean water. The

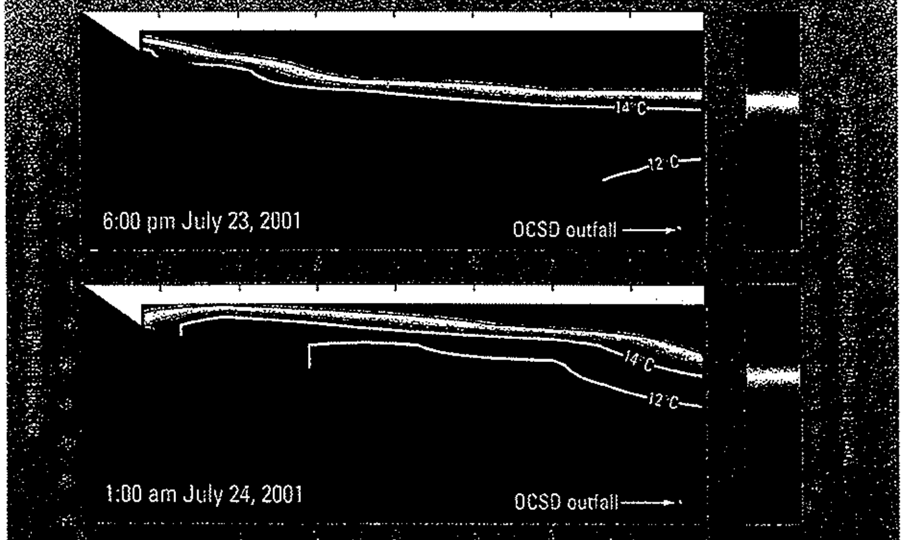
In the summer of 2001, scientists from the U.S. Geological Survey (USGS) and cooperating organizations designed and carried out an extensive study to evaluate whether the Orange County Sanitation District's (OCSD) ocean outfall could be the source of bacterial contamination at Huntington Beach. This study focused on coastal ocean processes that could move significant volumes of bacteria-laden OCSD wastewater into the nearshore region and surf zone during summer months. This map shows locations of instrument-mooring sites, hydrographic survey lines, and surf-zone bacteria sampling stations used in the USGS-led study. At mooring sites, arrays of instruments, such as the tripod shown below being lowered to the sea floor, monitored current velocity, temperature, salinity, and meteorological conditions. Hydrographic surveys used arrays of instruments towed or lowered from boats to measure the spatial distribution of temperature, salinity, ammonia content, bacteria concentrations, and other properties of the water column.



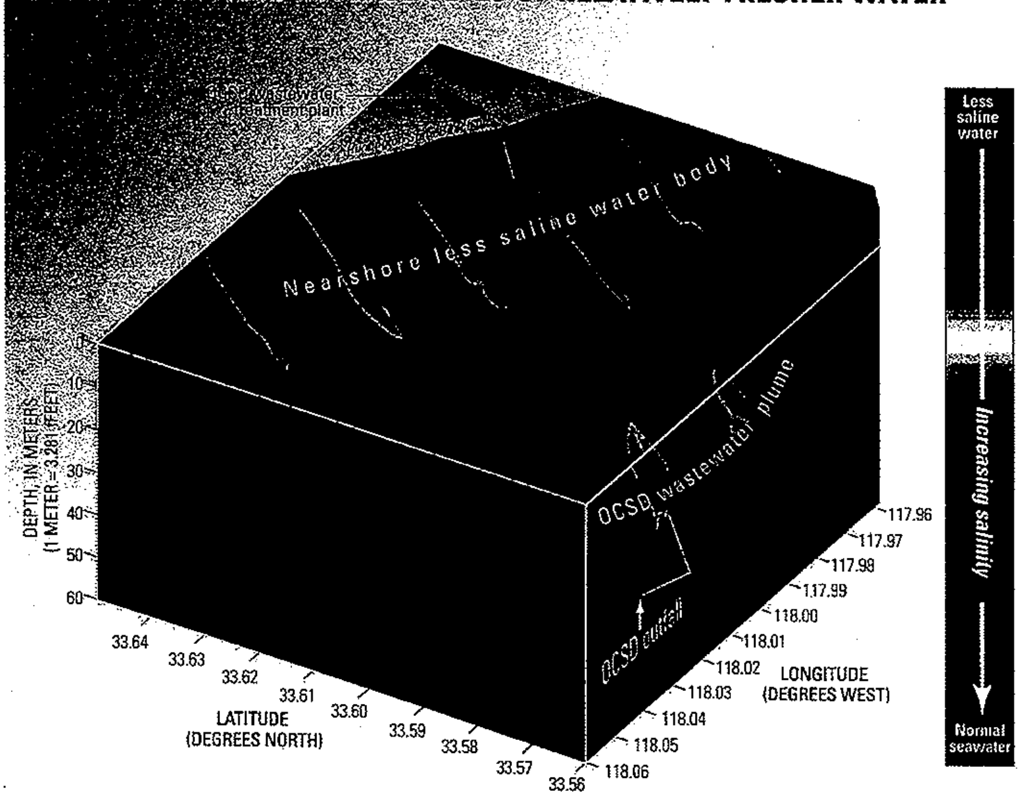


wastewater plume tends to stabilize and mix with a layer of water that has a temperature of 54 to 57°F (12 to 14°C) and for the most part is carried out of the area by alongshore currents. However, data show that water of that temperature was intermittently brought nearshore during July and August 2001. These cold-water pulses were the result of a combination of internal tides (tidal-cycle waves on density boundaries within the water column, like the thermocline) and daily circulation induced by sea breezes. It had been hypothesized that these mechanisms could bring wastewater from the OCSD plume into the proximity of the cooling-water intake and discharge pipes of the Huntington Beach AES Corporation electrical powerplant, where plume water could be sucked up and then be injected by the discharge jet into surface waters. The OCSD effluent could then easily move into the surf zone through buoyant spreading, wind forcing, or other processes. However, the USGS-led study found no association between the timing of nearshore cold-water pulses and beach closures or postings on the shoreline adjacent to the AES facility, which has been a hotspot for bacterial contamination.

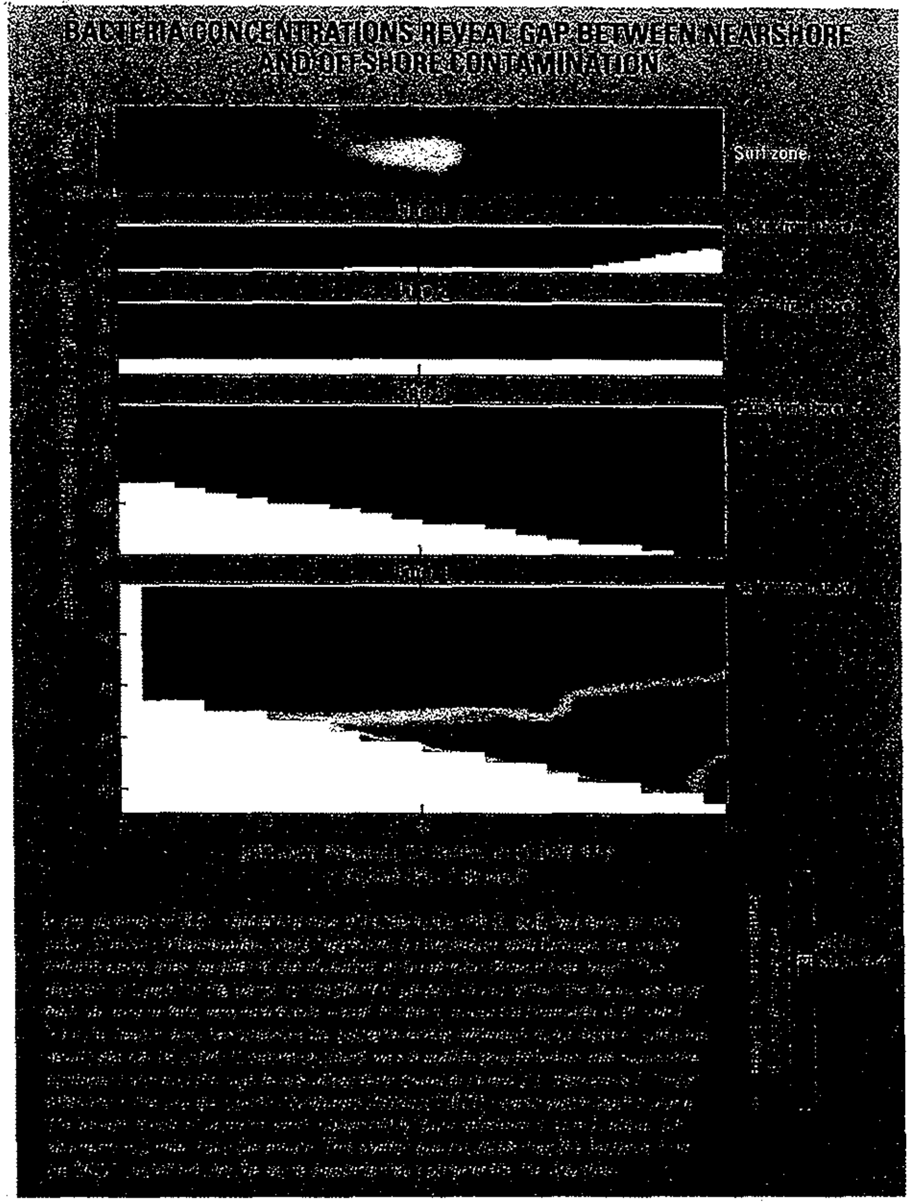
TEMPERATURE PROFILES SHOW NEARSHORE COLD-WATER PULSES



SALINITY PROFILES REVEAL BODIES OF RELATIVELY FRESHER WATER



This diagram shows salinity variations offshore of Huntington Beach in early July 2001, measured along hydrographic survey lines (see map). As shown in this diagram, salinity data from the summer of 2001 clearly show two distinct bodies of relatively less saline water—one in deeper water offshore associated with the Orange County Sanitation District's (OCSD) wastewater outfall plume and one in shallower nearshore waters. The nearshore less saline water could be a possible source of the bacterial contamination that has caused beach closures at Huntington Beach. This water may be coming from the San Gabriel and Los Angeles Rivers, to the north, which carry urban runoff into the ocean.



Observations from the USGS-led study provide no evidence that the bacteria found in the OCSD treated-effluent plume reached any part of the shoreline at Huntington Beach in significant enough quantities to cause beach closures or postings. The data show that when the beaches have high bacteria levels, very nearshore water about 0.25 mile (0.4 km) from the beach also has measurable bacteria levels, although much below AB411 standards. However, there was a spatial gap between this nearshore contamination and the high levels of bacteria measured 2 miles (3.3 km) offshore below the thermocline in the OCSD outfall plume. The lowest levels of bacteria were consistently found offshore about 0.5 mile (0.8 km) from the beach. This spatial gap suggests that the bacteria from the OCSD outfall are not the same bacteria that contaminate the beaches. The bacteria causing the beach postings and closures at Huntington Beach are most likely coming from other sources, such as water from local storm drains, marshes, or some other as yet unidentified source.

Salinity data from the summer of 2001 clearly show two distinct bodies of relatively fresher (less saline) water—one offshore below the thermocline associated with the OCSD plume and one in shallower nearshore waters. This observation corresponds well to the spatial gap seen in the bacterial data. Although additional work needs to be done to identify the source of the nearshore body of fresher water, it may be coming from the San Gabriel and Los Angeles Rivers, to the north, which carry urban runoff into the ocean.

Beach closures from bacterial contamination continued at Huntington Beach in the summer of 2002. Even though the data show that the OCSD outfall plume is almost certainly not the source of that contamination, OCSD began chlorinating its wastewater discharge in the fall of 2002. Samples taken from the OCSD offshore outfall plume since then show low levels of bacteria, often below detection limits. However, periods of bacterial beach contamination at levels above AB411 standards have occurred at Huntington Beach after OCSD implemented chlorination, supporting the conclusion of the USGS-led study that the OCSD outfall plume is not the culprit in the beach contamination.

USGS and its cooperators are continuing to investigate how bacteria, other contaminants, and sediments are transported by coastal ocean processes not only off Huntington Beach but also in the larger region off southern California. Such research is only part of the USGS efforts to protect people's lives and property from geologic and environmental hazards in the coastal zones of the United States.

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O-11

Generation of Enterococci Bacteria in a Coastal Saltwater Marsh and Its Impact on Surf Zone Water Quality

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Elevated levels of enterococci bacteria, an indicator of fecal pollution, are routinely detected in the surf zone at Huntington State and City Beaches in southern California. A multidisciplinary study was carried out to identify sources of enterococci bacteria landward of the coastline. We find that enterococci bacteria are present at high concentrations in urban runoff, bird feces, marsh sediments, and on marine vegetation. Surprisingly, urban runoff appears to have relatively little impact on surf zone water quality because of the long time required for this water to travel from its source to the ocean. On the other hand, enterococci bacteria generated in a tidal saltwater marsh located near the beach significantly impact surf zone water quality. This study identifies a potential tradeoff between restoring coastal wetlands and protecting beach water quality and calls into question the use of ocean bathing water standards based on enterococci at locations near coastal wetlands.

Introduction

Beaches are an important part of the culture and economy in California. An estimated 550 million people visit California's public beaches annually for a total economic benefit to the state of over 27 billion dollars (*1*). To protect beach-goers from exposure to waterborne disease, a new state law mandates the implementation of recreational water quality monitoring programs at public beaches with 50,000 or more annual visitors. Specifically, the law requires monitoring for total coliform (TC), fecal coliform (FC), and the enterococcus (ENT) groups of bacteria, all of which may indicate the presence of fecal contamination. The state also enforces a set of uniform standards for TC, FC, and ENT bacteria including single-sample standards (10,000, 400, and 104 most probable number (MPN) or colony forming units (CFU)/100 mL) and 30 day geometric mean standards (1000, 200, and 35 MPN or CFU/100 mL); a lower single-sample standard for TC of 1000 MPN or CFU/100 mL also applies when the TC/FC ratio falls below 10. The enterococci standard conforms closely to the national guidelines for marine water quality criteria published by the U.S. Environmental Protection Agency (*2*). If indicator bacteria levels in the ocean exceed any of the above standards, the local health officer is required to either post signs that warn against swimming in the water or close the ocean to the public if a sewage spill is suspected. The state standards and U.S. Environmental Protection Agency guidelines are based on a series of epidemiological studies that link gastrointestinal illness and exposure to ocean water containing high levels of indicator bacteria, particularly ENT (*3*–*11*). The origin of ENT in these epidemiological studies was presumed to be anthropogenic sources of fecal pollution, such as sewage, agricultural runoff, and urban runoff.

Huntington State and City Beaches in southern California have been heavily impacted by the passage of the new regulations. According to data provided by the Orange County Health Care Agency, there have been a total of 99 postings at Huntington State and City Beaches between July 26, 1999, when the bill went into effect, and September 5, 2000, approximately 72% and 25% of which were triggered by violations of the ENT single-sample and geometric mean standards, respectively. Persistently high levels of indicator bacteria in the surf zone at Huntington State and City Beaches in the summer of 1999 led to an extensive survey of the local sewage infrastructure (*12*). No significant sewage leaks were discovered, prompting speculation that urban runoff from the nearby Talbert Watershed was a source of fecal pollution (*12*). The present study was designed to test this hypothesis and, more broadly, to characterize the sources and transport of ENT in tidally influenced flood control channels and a saltwater marsh. ENT was the focus of this study because this particular group of indicator bacteria is responsible for the vast majority (97%) of beach advisories issued at Huntington State and City Beaches.

Field Site

The Talbert Watershed encompasses 3400 hectares in the cities of Huntington Beach and Fountain Valley. The watershed drains an urbanized area consisting of residential developments, commercial districts, plant nurseries, and light industry. This area of southern California has separate stormwater and sanitary sewer systems, so dry- and wet-weather runoff flows to the ocean without treatment.

Runoff from the Talbert Watershed is conveyed along street gutters to inlets that connect to underground stormwater pipelines. These pipelines connect to a network of three flood control channels (Fountain Valley, Talbert, and

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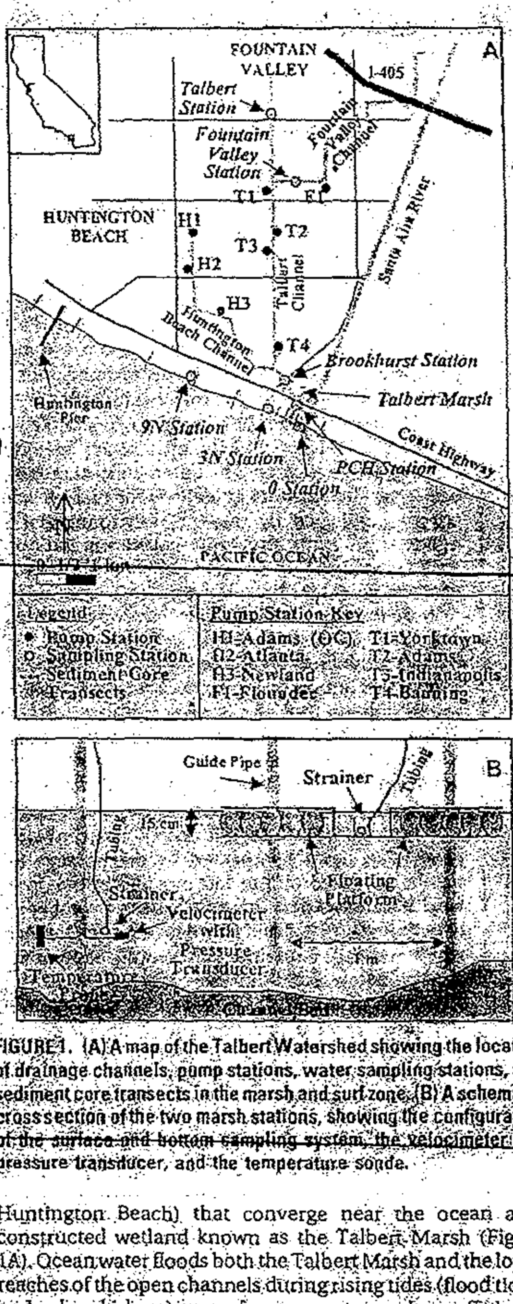

FIGURE 1. (A) A map of the Talbert Watershed showing the location of drainage channels, pump stations, water sampling stations, and sediment core transects in the marsh and surf zone. (B) A schematic cross section of the two marsh stations, showing the configuration of the surface and bottom sampling system, the velocimeter and pressure transducer, and the temperature sonde.

Huntington Beach) that converge near the ocean at a constructed wetland known as the Talbert Marsh (Figure 1A). Ocean water floods both the Talbert Marsh and the lower reaches of the open channels during rising tides (flood tides) and a brackish mixture of ocean water and runoff drains from the system during falling tides (ebb tides).

The Talbert Watershed is nearly flat and only a few feet above sea level. This geographical setting hinders drainage by gravity alone, so a system of transfer stations is used in the lower reaches of the Talbert Watershed to pump runoff into the open channels from stormwater pipelines. Each transfer station, or pump station, consists of a forebay, where runoff can be stored, and several pumps. Pumping of runoff to the channels occurs intermittently during dry weather periods and continuously during storms.

Talbert Marsh is a 10 hectare remnant of what used to be an extensive (1200 hectare) saltwater wetland and dune system in coastal Orange County. The majority of this wetland system was drained and filled over the past century for agricultural reclamation and urban development. Most of what remained of the historical wetland, including Talbert Marsh, was cut off from tidal flushing by the construction of Pacific Coast Highway and channelization of the surrounding area for flood control. As part of a habitat restoration effort, tidal flushing in the Talbert Marsh was restored in 1990 when a new tidal inlet was constructed. Since its restoration, Talbert Marsh has become a typical southern California tidal saltwater marsh with open water, wetland, and upland habitats (13–15). Pickle weed (*Salicornia virginica*) is the dominant macrophytic vegetation, and the marsh is utilized by several special-status bird species including the California Least Tern, Brown Pelican, and Beldings Savannah Sparrow.

At the outset of this study it was not clear what effect the Talbert Marsh had on surf zone water quality at Huntington State and City Beaches. On one hand, wetlands, particularly freshwater wetlands, are natural treatment systems that remove chemical and biological pollutants from domestic and agricultural wastewater and urban runoff (19, 20). On the other hand, coastal marshes are an important bird habitat, and bird feces are a potential source of ENT (21, 22), as is the environmental growth of these organisms in the sediments and on vegetation (23–26).

Methods and Materials

A series of investigations were carried out to (1) quantify the flow of water and ENT into the ocean from the Talbert Marsh and Talbert Watershed, (2) assess the impact of ENT from the marsh and watershed on local surf zone water quality, and (3) identify potential sources of indicator bacteria within these two systems (runoff, birds, vegetation, and sediment). These three different investigations are referred to throughout the paper as the Marsh Study, the Surf Zone Study, and the Source Study, respectively. The methods employed in these investigations are described below.

Marsh Study. The goal of the Marsh Study was to measure the flow of water and ENT from the Talbert watershed into the Talbert Marsh and from the Talbert Marsh into the ocean. Measurements were carried out for 15 days starting on May 2, 2000. During the 15 day study, pump stations in the Talbert Watershed were operated in two different modes: during the first 8 days the pump stations were offline, and for the following 7 days the pump stations were online. When the pump stations were offline, runoff that would normally be discharged into the drainage channels was either diverted into the regional sanitary sewer system or stored in the pump station forebays. When the pump stations were online, runoff was intermittently discharged into the drainage channels following normal operating procedures. The impact of these operational changes was monitored at two locations: (i) the junction of the drainage channel network and the marsh at the Brookhurst street bridge (Brookhurst Station) and (ii) the junction of the marsh and the ocean at the Pacific Coast Highway bridge (PCH Station) (see Figure 1A). Two additional sites (Talbert Station and Fountain Valley Station, see Figure 1A) were monitored to characterize the flow of runoff into the drainage channels from the upper reaches of the watershed where there are no pump stations. Methods for monitoring the flow of water and ENT concentrations at these four sites are described below.

Flow Measurements. The velocity and level of water at the Brookhurst Station and the PCH Station were measured using acoustic Doppler velocimeters outfitted with pressure transducers (4250 Area Velocity Flow Meter, Isco, Lincoln, NE). The velocimeters were suspended approximately 5 cm above the sediment bed (Figure 1B) and positioned so that the Doppler cone, or area over which the velocity is averaged, was pointing upward and in an inland direction. Data from the velocimeters was electronically logged every five minutes and downloaded onto a laptop computer. The velocity and water level data were used to calibrate a hydrodynamic model for the marsh and channel network (27). The calibrated model

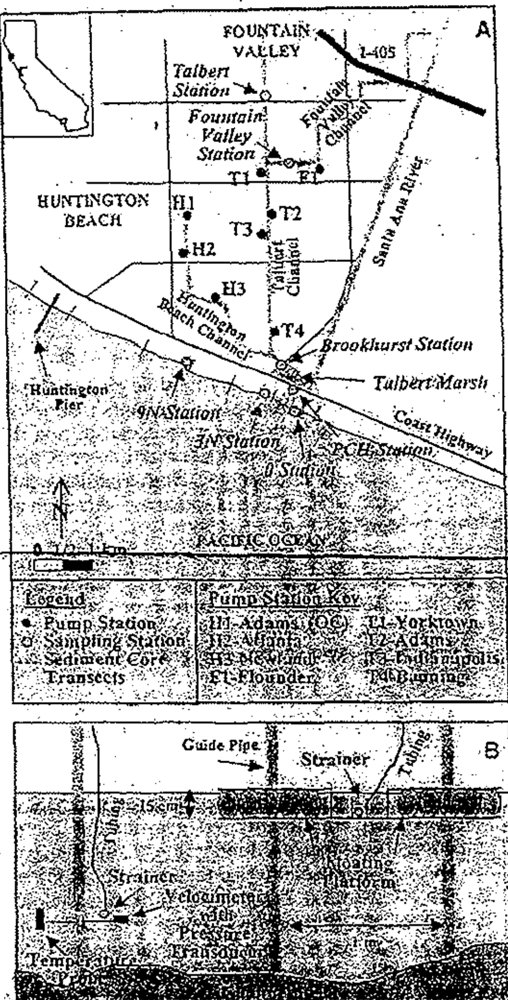

FIGURE 1. (A) A map of the Talbert Watershed showing the location of drainage channels, pump stations, water sampling stations, and sediment core transects in the marsh and surf zone. (B) A schematic cross section of the two marsh stations, showing the configuration of the surface and bottom sampling system, the velocimeter and pressure transducer, and the temperature sonde.

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The flow of urban runoff into the upstream reaches of the Talbert and Fountain Valley channels was too low to measure using acoustic Doppler technology. Consequently, flow rates at the Talbert and Fountain Valley Stations were estimated by recording the time 10 different pieces of submerged debris took to travel a fixed distance. Volumetric flow rates were then obtained by multiplying this average velocity by the estimated cross sectional area of the flowing water.

No water was discharged from the pump station forebays during the first 8 days of the Marsh and Surf Zone Studies. The volume of water discharged during the last 7 days of the study was estimated from City of Huntington Beach records of water volumes diverted into the sanitary sewer during the first 8 days of the study. The conductivity of forebay water at several pump stations was elevated (30 mS/cm), reflecting the fact that some fraction of the forebay water is ocean water that traveled up the channels during flood tides and spilled into the forebays through leaking flap gates. We computed the fraction of water discharged from the pump stations that was runoff (i.e., not ocean water) as follows

$$F = 1 - (C - C_R)/(C_O - C_R) \tag{1}$$

where C_O and C_R are the conductivity of ocean water and runoff (taken as 53.5 and 3 mS/cm, respectively) and C is the measured conductivity of samples from the pump stations.

The volume of runoff exiting the channel network through the outlet to the ocean was quantified from the magnitude of the conductivity depressions and the volumetric flow rate at the PCH Station by numerically evaluating the following integral

$$\int F(t)Q(t)dt \tag{2}$$

where $F(t)$ represents the fraction of freshwater computed by applying eq 1 to the conductivity signal measured at the PCH Station and $Q(t)$ is the volumetric flow rate at the PCH Station computed using the calibrated hydrodynamic model (see above). The integral was taken separately over the first 8 days and last 7 days of the study.

ENT Measurements. At both the Brookhurst Station and the PCH Station, hourly water samples were collected from the surface and bottom of the water column using programmable sampling units (Isco models 3700 and 6700, Lincoln, Nebraska) (Figure 1B). Surface samples were obtained by drawing water over the lip of an acrylic box that was submerged approximately 1 cm below the water surface and supported by a floating platform (Figure 1B). Bottom samples were drawn through a strainer suspended approximately 5 cm above the sediment bed by a pole attached to the bridge. To obtain an average measure of water quality over each hour-long sampling interval, the automated samplers were programmed to collect 200 mL of water every 15 min for a total sampling volume of 800 mL per bottle per hour. Sample bottles consisted of a disposable plastic liner (Isco ProPak sample bags) supported by a plastic cage (Isco ProPak holder); the liners were used once and then discarded. A purge cycle was executed before and after each sampling event, and the sampling units were filled with ice to reduce bacterial die-off. Samples were retrieved from the Brookhurst and PCH Stations every 6 h and transported to a laboratory at the Orange County Sanitation District (Fountain Valley, CA) where 10 mL was immediately analyzed for ENT using a defined substrate test (IDEXX Enterolert test implemented in a 97 well Quanti-tray format), pH, turbidity, and conductivity (temperature-corrected to 20 °C). A total of 1416 samples were collected using the automated samplers. Automated samplers were employed here because they allowed us to collect hourly water samples in a reproducible manner from precisely the same locations in the water column, 24 h per day, 7 days per week. One potential disadvantage of the automated systems is that the tubing and sampling system (e.g., strainers) are not sterilized between sampling events, so there is a possibility that sample-to-sample cross-contamination might occur. A recent study of sources of *E. coli* in an estuarine system in Florida (*26*) found that automated samplers did not cause significant cross-contamination when a purge step was executed between sampling events, as was done here.

Solar Radiation. To assess possible relationships between sunlight and bacterial levels in the marsh, hourly measurements of solar radiation were recorded during the 15 day study period using a thermopile radiometer (Kipp & Zonen, CM3 Thermopile Radiometer, Netherlands), located at the San Joaquin Marsh, which is approximately 6 km west of the Talbert Marsh.

Surf Zone Study. Dye experiments and intensive surf zone water quality monitoring were carried out to quantify the impact of ENT from the Talbert marsh and watershed on surf zone water quality at Huntington State Beach. The methods employed for this element of the study are described below.

Dye Study. During ebb tides, water from the Talbert Watershed flows into the drainage channels (Huntington Beach, Talbert, and Fountain Valley), through the Talbert Marsh, and into the ocean. To determine how ebb flow from the Talbert marsh and watershed interacts with the surf zone, separate dye experiments were conducted on May 1 and May 10, 2000, as follows. Rhodamine WT dye (Keystone, Santa Fe Springs, California) was added for approximately 30 min to effluent from the Talbert Marsh during an ebb tide. The spatial distribution of the dye was recorded at a series of times post release by a four channel radiometer (DMSV MK-1 SpecTerra Sys., Nedlands, Australia) flown at approximately 1500 m above sea level. The dye field in these images was visualized by forming the ratio of emission and absorption maxima (570 and 550 nm, respectively) of Rhodamine WT.

Surf Zone Monitoring. To assess the impact of ENT from the marsh and watershed on surf zone water quality, hourly samples were collected at the PCH Station (to characterize the concentration of ENT entering and leaving the marsh) and at three locations in the surf zone (stations 0, 3N, and 9N; see Figure 1A). The Surf Zone Study was carried out during the same period of time (May 2–16, 2000) as the Marsh Study (see above). However, the methods used to collect and analyze samples in the Surf Zone Study differed from those described above for the Marsh Study. For the Surf Zone Study, hourly grab samples (total volume of approximately 1L) were collected in sterile Nalgene bottles at the PCH and the surf zone stations 24 h per day, 7 days per week, for 2 weeks. Within 6 h of collection, samples were transported to Sierra Laboratories, Inc. (Laguna Hills, California) on ice where 10 mL of each sample was immediately analyzed for ENT using multiple tube fermentation (MTF) (EPA Method 9230B). To characterize cross-shore variability of the ENT signal, separate samples were collected from ankle and waist depths at each surf zone station. A total of 2021 grab samples were collected for this element of the study.

ENT Source Study. Additional studies were carried out to identify specific sources of ENT in the marsh and watershed. Specific sources examined included urban runoff, bird feces in the marsh, marine vegetation, and marsh and surf zone sediments, as described below.

Bird Feces. To assess the amount of ENT present in bird feces, bird feces were collected, along with any attached sediment from mud flats, in the Talbert Marsh where birds congregate. The nature of the feces (wet or dry) was noted at the time of collection. Sediment that appeared to contain no bird feces was also collected to determine background levels of ENT. The sediment and feces samples were weighed and placed in acid washed Nalgene bottles with 500 mL of marsh water. The suspensions were shaken vigorously to disperse the feces and sediment and then allowed to settle for 15 min. Depending on the experiment, between 0.1 and 10 mL of supernatant was tested for ENT using the Enterolert protocol described in the Marsh Study. Control experiments were conducted to rule out the possibility that chemicals present in the feces and/or sediment might interfere with the Enterolert system. Specifically, Enterolert analyses were conducted on autoclaved suspensions of sediment and bird feces.

Bird Census. To quantify the input of ENT into the marsh from birds, a bird census was carried out as follows. Digital cameras (Kodak Model DC-290, Rochester, New York) were installed at three different locations along the northeastern margin of the marsh. These cameras were positioned so that, together, they provided a complete picture of the upland, wetland, and open water habitat areas. Images were shot hourly at a resolution of 2240 × 1500 pixels in 256 colors, 24 h per day, over the same period of time when samples were being collected in the marsh and in the surf zone (May 2–16, 2000). The images were uploaded to a desktop PC where they were analyzed with Adobe Photoshop (Adobe, San Jose, California). The birds in each image were enumerated manually to obtain an estimate for the total number of birds present in the marsh each hour of the 2-week study.

Urban Runoff. To characterize the concentration of ENT in urban runoff, daily grab samples were collected from all 11 pump stations in the Talbert Watershed and from the upstream reaches of the watershed at the Talbert and Fountain Valley Channel Stations (Figure 1A). Runoff sampling occurred over the same period of time that the Marsh and Surf Zone Studies were carried out (May 2–17, 2000). Prior to sampling the pump station forebays, water in the forebay was mixed by cycling the station pumps on and off. Sterile Nalgene bottles were lowered into the underground forebays, and approximately 1 L of water was collected. Five hundred mL samples of runoff at the Talbert and Fountain Valley Channel Stations were collected by manually placing a sterile Nalgene bottle directly in the flowing stream. All samples were stored on ice immediately after collection and transported to the Orange County Sanitation District where they were analyzed for pH, turbidity, conductivity, and ENT using the Enterolert protocol described in the Marsh Study.

Sediment and Vegetation. To assess the levels of ENT present in sediments, cores were collected from the marsh and surf zone with a Brandford 5024 Pneumatic Vibrator (Brandford Co., New Britain, CT) outfitted with a 1.52 m barrel (OD 4.4 cm) and Butyrate plastic liners (AMS Inc., American Falls, ID). Each core was cut into three 15 cm segments which were sealed at the ends with Teflon lined caps and transported to Sierra Laboratories, Inc. (Laguna Hills, CA) for bacterial analysis. Upon arrival at the laboratory, 50 g of each core section was suspended in 450 mL of phosphate buffered saline (PBS) (0.3 mM KH_2PO_4, 2 mM $MgCl_2$) in accordance with Standard Method 9221 A-3 (28). The clarified supernatant was analyzed for ENT using MTF following the protocol outlined in the Surf Zone Study. Seaweed samples were collected from the marsh, stored in disposable plastic bags, and transported on ice to Sierra Laboratories, Inc. Upon arrival at the lab, 50 g of vegetation was placed in a sterile container to which 450 mL of PBS was added. The solution was shaken vigorously and allowed to settle for 15 min and then reshaken. A 100 mL sample of the supernatant was analyzed for ENT using the MTF method described in the Surf Zone Study.

Results and Discussion

Marsh Study: Dynamics. The Talbert Marsh is a highly dynamic system, primarily because the flow of water through the marsh is dominated by the tides (Figure 2). Because Southern California has semidiurnal unequal tides (*29, 30*), there are four different tidal extrema each day including high–high, low–high, high–low, and low–low tide levels. Furthermore, the tide range, which is the difference between the high–high and low–low levels, oscillates over a 14–15 day period. The Marsh and Surf-Zone Studies were carried out over a 15 day period that began shortly before a spring tide when the tide range is maximal, passed through a neap tide when the tide range is minimal, and returned back to a spring tide again. The four daily tide stages and the spring-neap-spring transition are evident in the water levels measured at the Brookhurst and PCH Stations (top panel in Figure 2).

During flood tides (indicated by negative velocities in the second panel of Figure 2), the water levels at the Brookhurst and PCH Stations increase as water flows from the ocean, through the marsh, and inland along the channel network. During ebb tides (indicated by positive velocities) the water levels at the two stations decrease as water flows out of the channel network, through the marsh, and into the ocean. When ebb tides occur during daylight hours, solar heating of water flowing out of the channel network causes a significant increase in the temperature of the marsh water (compare first, third, and fourth panels). The conductivity measured at the Brookhurst and PCH Stations (fifth panel) corresponds to pure ocean water during flood tides (53.5 mS/cm) and a brackish mixture of ocean water and urban runoff at the end of the ebb tides (conductivity depressions).

The next panel in Figure 2 is a plot of the ENT concentrations measured at the Brookhurst and PCH Stations. ENT concentrations in the marsh varied from below the detection limit (10 MPN/100 mL) to a high of 2142 MPN/100 mL. A total of 218 (15%) and 655 (46%) of the marsh samples exceeded the single-sample and geometric mean standards for ENT (104 MPN/100 mL and 35 MPN/100 mL, shown as dark and light blue lines in the plot), respectively. A total of 247 (17%) of the marsh samples fell below the detection limit of 10 MPN/100 mL; all values falling below the detection limit were arbitrarily assigned the detection limit value. The log-transformed ENT concentrations at the top and bottom of the water column in the marsh are correlated ($r = 0.7$ and $r = 0.72$ at the Brookhurst and PCH Stations, respectively). Comparing the conductivity and ENT curves in Figure 2, we find that elevated ENT values frequently occur in the marsh during periods of time when runoff from the drainage channels, as indicated by the conductivity depressions, is not present.

The last panel in Figure 2 is a plot of the total number of birds that visited the Talbert Marsh during the course of our study. The birds followed a daily routine in which their numbers started out low in the morning, peaked in the afternoon, and tapered off in the evening. Gulls and Elegant Terns constituted the majority (80%) of birds visible in the images. The largest congregation of birds, 1180 individuals, occurred at 2:00 in the afternoon on May 5.

Marsh Study: ENT Source or Sink? A primary objective of this study was to determine if the marsh functions as a net source or sink of ENT as water flows out of the Talbert Watershed drainage channels, through the marsh, and into the ocean during ebb tides. To this end, we segregated all of the marsh ENT data into two groups based on whether the samples were collected during ebb tides (Figure 3A,B) or

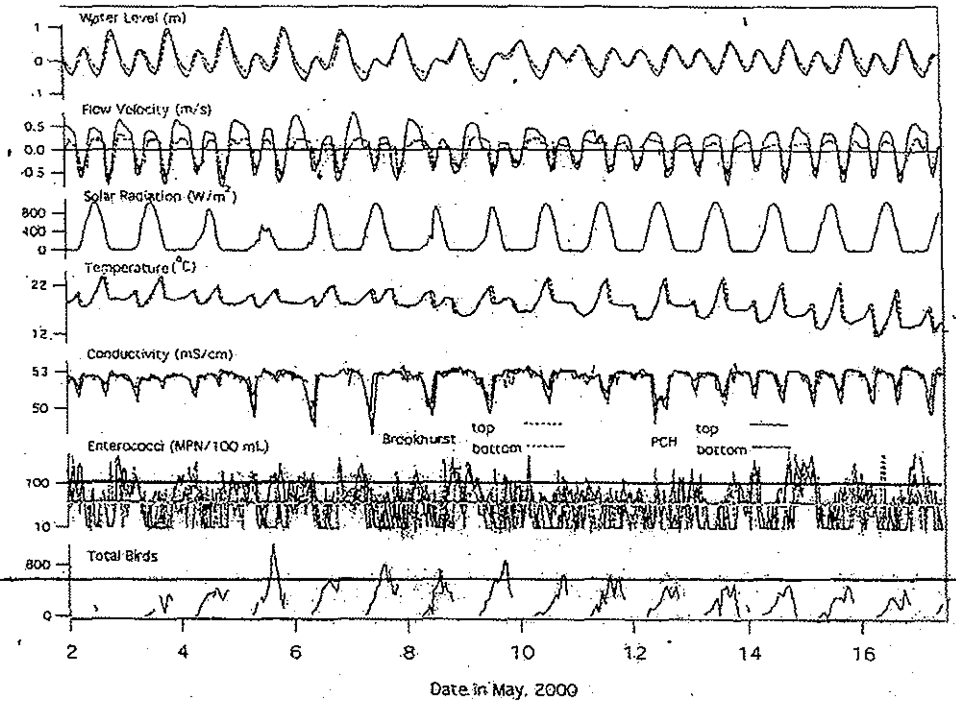

FIGURE 2. The dynamics of marsh parameters measured during the 15 day study period. The solid and dashed lines (water level, flow velocity, temperature panels) correspond to the PCH and Brookhurst stations, respectively. The key for conductivity and ENT traces is indicated in the figure. The dark and light blue lines denote the single sample and geometric mean standards for ENT. Water level is referenced to mean sea level. Positive and negative velocities correspond to shoreward and landward flow, respectively. The gray vertical stripes represent night-time conditions.

flood tides (Figure 3C,D). These data were further segregated based on whether the samples were collected during the first 8 days of the study (when the pump stations were offline) or the last 7 days of the study (when the pump stations were online) and based on the vertical location of samples in the water column (top or bottom). For each subgroup of data we computed a geometric mean and tabulated the percentage of samples that exceeded the single-sample standard for ENT. The results of this analysis identify the marsh, not urban runoff from the Talbert Watershed, as the primary source of ENT in the water flowing into the ocean. During ebb tides, the geometric mean of ENT (Figure 3A) and the percentage of samples exceeding the single-sample standard (Figure 3B) approximately double as the water flows through the marsh from the Brookhurst to PCH Station. The trend is reversed during flood tides when the geometric mean of ENT (Figure 3C) and percentage of single-sample exceedences (Figure 3D) increase as water flows through the marsh from the PCH to Brookhurst Station. With the exception of two flood-tide cases, water enters the marsh below the geometric mean standard for ENT (35 MPN/100 mL, dashed line in the figure) and exits the marsh in exceedence of the standard. In several cases, the ENT concentrations measured at the top of the water column are higher than the ENT concentrations measured at the bottom of the water column.

The idea that the marsh is a net source of ENT is also supported by Figure 3E, where we plot the hour-by-hour difference between the ENT concentrations measured at the Brookhurst and PCH Stations (ΔENT). On average, the ENT concentration is higher at the PCH Station during ebb tides (mean ΔENT = −29 ± 7 MPN/100 mL) and higher at the Brookhurst Station during flood tides (mean ΔENT = 27 ± 6 MPN/100 mL). A direct comparison of the ENT concentrations at the Brookhurst and PCH Stations is valid only if the residence time of water in the marsh is less than our sampling interval of 1 h. This condition appears to be satisfied based on a dye study conducted on the morning of May 19, 2000, which found that the residence time of water in the marsh during a weak spring tide is less than 40 min (*27*).

Surf Zone Study: Dye Experiment. The above analysis demonstrates that the Talbert Marsh is a net source of ENT, but it is not clear that ENT generated by the marsh negatively impact surf zone water quality. To characterize how ebb flow from the Talbert Marsh interacts with the ocean, a set of experiments were conducted in which dye (Rhodamine WT) was injected into the outlet of the Talbert Marsh during two separate ebb tides, one on May 1 and the other on May 10, 2000. The spatial pattern of dye released from the Talbert Marsh during the May 1 experiment is displayed in Figure 4. The dye pulse split into two plumes as it flowed into the ocean. One plume was entrained in the surf zone where it rapidly advected upcoast at velocities exceeding 0.2 m/s; a portion of this plume was subsequently taken offshore by a rip current. The second plume was carried directly offshore by a momentum jet located at the mouth of the marsh. The portion of the dye entrained in the surf zone on May 1 was advected in an upcoast direction because, on that day, ocean waves with average significant heights of 0.7 m were from the south (*31*). During the second release on May 10, ocean waves with significant heights of 1.4 m were from the west, and the portion of the dye entrained in the surf zone was advected rapidly (0.3 m/s) in a down coast direction (data not shown). Hence, water flowing out of the marsh during ebb tides can impact surf zone water quality at Huntington State and City Beaches directly upcoast of the Talbert Marsh outlet, provided that ocean waves strike the beach in an upcoast direction. Interestingly, wave conditions similar to those observed during the May 1 experiment were also present during the summer of 1999 when large stretches of Huntington State and City Beaches were closed to the public

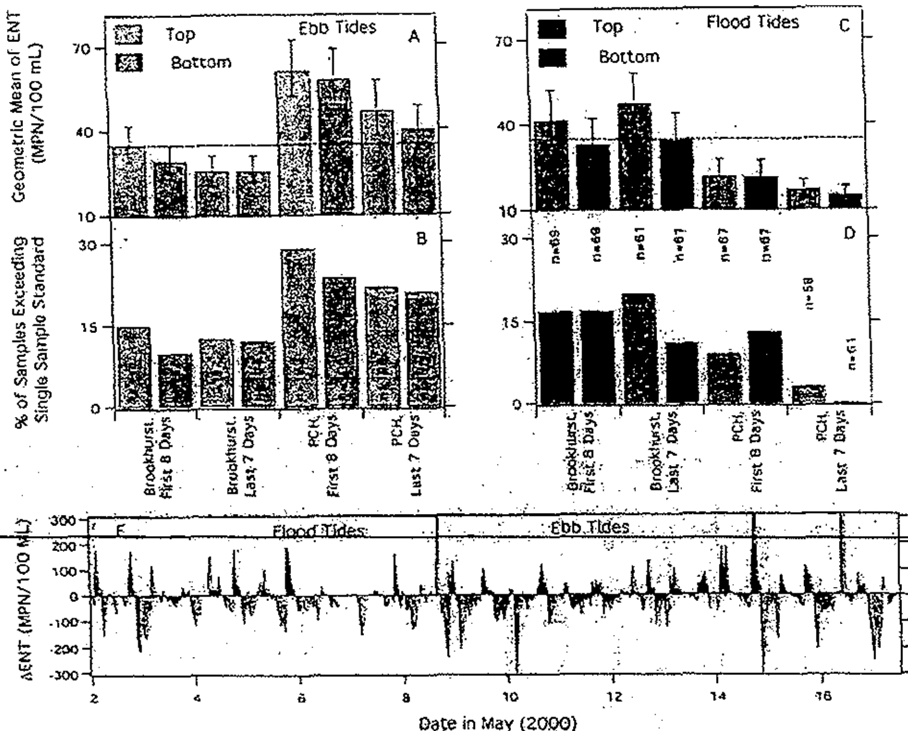

FIGURE 3. Geometric means of ENT in samples collected during ebb tides (A) or during flood tides (C). The dashed line in these figures represents the geometric mean standard for ENT (35 MPN/100 mL). Also shown are the percentage of samples collected during ebb tides (B) or flood tides (D) that exceeded the single sample standard for ENT (104 MPN/100 mL) and the difference in ENT concentrations at Brookhurst and PCH (E). Error bars represent 95% confidence intervals. The number of samples used to calculate geometric mean values are indicated in the figure.

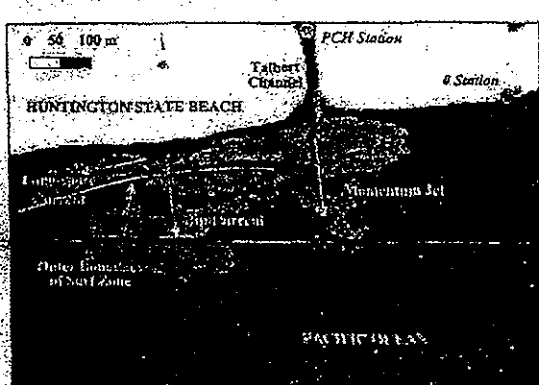

FIGURE 4. An areal image showing the near shore distribution of Rhodamine WT dye at 11:51 PDT, approximately 25 min into a release from the Talbert outlet during ebb tide on May 1, 2000.

(personal communication City of Huntington Beach lifeguards, 2000).

In addition to providing qualitative information about the fate of marsh effluent as it enters the ocean, the dye experiments can also be used to estimate the dilution that occurs as ebb flow from the marsh becomes entrained in the surf zone. Concentrated dye was released into the Talbert Marsh outlet at a rate of $Q_{dye} = 8 \times 10^{-6}$ m^3/s. From the calibrated hydrodynamic model, we estimate that the volumetric flow of water out of the marsh during the dye study on May 1 was relatively steady and equal to $Q_{effluent} = 11.6$ m^3/s. Photographs of the dye release indicate that the dye plume mixed over approximately one-half of the channel cross section before reaching the surf zone (*31*). Taking this observation into account, we estimate that the initial dilution of the dye plume into the marsh effluent stream was approximately 7.0×10^5 $((Q_{effluent}/2)/Q_{dye})$. The volume of the dye field at 11:51 PDT (the time at which the DMSV image in Figure 4 was shot) was approximately 7×10^4 m^3 assuming a 1.5 m mixing depth. Therefore, the dilution of the plume at 11:51 PDT, which includes both the initial and the surf zone dilution, is the volume of the dye field (7×10^4 m^3) divided by the volume of the dye released (6.51×10^{-2} m^3) or 1.1×10^6. Taking the ratio of this total dilution (1.1×10^6) and the initial dilution (7.0×10^5) indicates that the marsh effluent stream was diluted by a factor of 1.6 as it became entrained in the surf zone. Hence, effluent leaving the Talbert Marsh during ebb tides suffers approximately a factor two dilution as it is entrained in the surf zone.

Surf Zone Study: Bacterial Monitoring. To measure the actual impact of ebb flow from the marsh on surf zone water quality, an intensive surf zone monitoring program was carried out in parallel with the 15 day Marsh Study described above. ENT measurements in the surf zone varied from below detection limit (10 MPN/100 mL) to a high of 5700 MPN/100 mL. A total of 69 (3%) and 298 (15%) surf zone samples exceeded the single-sample and geometric mean standard for ENT, respectively. A total of 1067 (53%) of the surf zone samples fell below the detection limit. As with the data collected in the Marsh Study, samples falling below the detection limit were arbitrarily assigned the detection limit value.

Figure 5 displays the geometric mean and 95% confidence intervals of ENT measured at surf zone stations (9N, 3N, and 0, see Figure 1A) and at the PCH Station during either rising or falling tides. These data are also segregated based on whether samples were collected in the first 8 days of the study or the last 7 days of the study (indicated in the figure

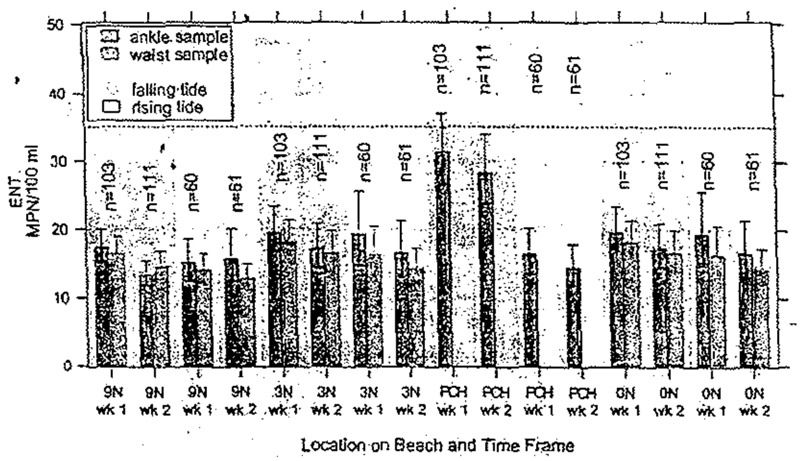

FIGURE 5. Geometric means and 95% confidence intervals of ENT concentrations (MPN/100 mL) at the PCH and surf zone stations measured during falling (blue background) and rising (white background) tides. The stations are displayed from north (left) to south (right): 9N, 3N, PCH, and 0 (see map in Figure 1). At each station, the geometric means are shown for the first 8 days and last 7 days (denoted wk 1 and wk 2, respectively). For the surf zone stations, geometric means for samples taken at ankle and waist depth are indicated. At the PCH site, only a surface sample was analyzed. The sample sizes are shown above the bars. The dotted line represents the geometric mean standard for ENT (35 MPN/100 mL).

as "wk 1" and "wk 2", respectively), whether the samples were collected at ankle or waist depth, and whether the samples were collected during rising or falling tides. As described in more detail in the Methods and Materials section, all of the ENT data plotted in Figure 5 were obtained by performing MTF analysis on grab samples, while the ENT data collected for the Marsh Study were obtained by performing an Enterolert analysis on samples collected with an automated sampling system. Comparing the PCH Station data in Figure 3A with the PCH Station data in Figure 5, we find that during ebb tides the geometric mean of ENT estimated using the Enterolert/automated sampling system is approximately 60 MPN/100 mL, compared to 30 MPN/100 mL using MTF/grab samples. ENT values estimated by the two approaches are weakly correlated ($r = 0.5$), but the magnitude of the ENT values estimated by the MTF/grab sample method appear to be lower. This difference could arise due to differences in the analytical technique employed (MTF versus Enterolert) and/or the sampling methodology employed (grab versus automated). A strong correlation between Enterolert and MTF measurements of ENT in marine samples ($r = 0.927$) has been previously reported (*32*). Hence, the differences reported here are probably due to the differences in the sample collection protocols employed in the Marsh and the Surf Zone Studies.

Because all of the data presented in Figure 5 were collected and analyzed using the same procedure (MTF on grab samples), we can directly compare the ENT signal leaving the marsh during ebb tides with the ENT signal measured in the surf zone over the same period of time. Figure 5 reveals that during falling tides, when ebb flow from the marsh enters the ocean, the geometric mean of ENT at the PCH Station is approximately two times higher compared to the geometric mean of ENT measured at the surf zone stations. With one exception, the geometric means of surf zone samples collected at waist depth are slightly lower than the geometric mean of samples collected at ankle depth. Based on these data, the ENT signal at stations 0, 3N, and 9N could have been caused by ebb flow from the Talbert Marsh provided that the following conditions were met: (1) near complete surf zone entrainment of the marsh effluent as it flows over the beach and into the ocean during falling tides, (2) no more than a factor of 2 dilution as effluent from the marsh is entrained in the surf zone, and (3) littoral flow in the surf zone directed in an upcoast direction. The first two conditions appear to be met based on the results of the dye study described above. Based on wave azimuth data recorded at Huntington Beach during the 15 day study (*31*), wave-induced flow in the surf zone was directed in an upcoast direction 60% of the time, including long stretches of time between May 4 and 8 and again between May 12 and May 16. Hence, ENT generated in the marsh appear to have at least a localized impact on surf zone water quality at Huntington State Beach.

ENT Source Study: Urban Runoff. No more than trace levels of rainfall were measured in Huntington Beach either during, or 14 days prior to, our 15 day study. Therefore, all runoff generated by the Talbert Watershed during this period was from dry weather sources, including landscape irrigation, street cleaning, car washing, and other activities that lead to surface water flow. To determine if the Talbert Watershed might be a significant source of ENT, samples of runoff were collected from pump station forebays and upstream at the Talbert and Fountain Valley Channel Stations (Figure 1A) and then analyzed for ENT using the Enterolert system (Table 1). The largest concentration of ENT (81,310 MPN/100 mL) was detected in a sample collected from the Flounder pump station on 5/8/00 (data not shown). The geometric mean of ENT in the runoff ranged from 23.1 MPN/100 mL at the Indianapolis pump station to 9477 MPN/100 mL in the upstream reaches of the Fountain Valley Channel (Table 1). Despite the high concentration of ENT measured in most urban runoff samples, the activation of pump stations during the last 7 days of our study did not appear to negatively impact downstream water quality. Indeed, the geometric means of ENT at the Brookhurst and PCH Stations during ebb tides (Figure 3A) actually *decreased* when pump stations came online. Likewise, the geometric means of ENT at all surf zone stations (Figure 5) were either unchanged when the pump stations went from offline to online or declined slightly.

There are several possible reasons why the discharge of pump station water did not lead to higher ENT concentrations in the marsh and surf zone. Mathematical modeling of tidal flow in the channel network reveals that water discharged from a particular pump station may or may not be flushed to the ocean in a single tide cycle, depending on the tidal range, when in the tide cycle the discharge occurred, and the pump station's inland distance from the shore. Specifically, the model predicts that at least 50% of runoff discharged during the last 7 days of our study was temporarily trapped

TABLE 1. Quality of Water That Enters the Channel Network from Either Uncontrolled Sources of Runoff (Talbert (T.) and Fountain Valley (F.V.) Channels) or from Pump Stations (p.s.)[a]

source	conduct. [mS/cm]	pH [−]	turbidity [NTU]	ENT (×10³) geometric mean [MPN/100 mL]	ENT (×10³) mean [MPN/100 mL]
Adams p.s.	4.5 (±1.3)	7.7 (±0.3)	10.2 (±5.1)	1.6 (+1.7/−0.8))	3.6 (±6.0)
Atlanta p.s.	32.3 (±6.9)	7.3 (±0.3)	22.1 (±4.8)	1.6 (+0.75/−0.51)	2.0 (±1.3)
Banning p.s.	36.3 (±3.8)	7.4 (±0.3)	9.3 (±2.0)	0.7 (+0.7/−0.3)	1.8 (±3.2)
OC Adams p.s.	3.0 (±0.8)	7.6 (±0.2)	24.7 (±11)	2.87 (+2.8/−1.4)	5.2 (±6.1)
Flounder p.s.	3.5 (±2.4)	7.4 (±0.4)	13.8 (±19.8)	1.9 (+6.1/−1.5)	12.5 (±17)
Indianapolis p.s.	11.1 (±1.9)	7.6 (±0.4)	11.5 (±5.3)	0.023 (+0.06/−0.02))	0.012 (±0.02)
Yorktown p.s.	8.0 (±2.6)	7.4 (±0.4)	27.2 (±9.9)	2.2 (+5.1/−1.6)	9.7 (±11)
Newland p.s.	19.7 (±4.5)	7.5 (±0.3)	10.4 (±4.9)	1.2 (+1.1/−0.6)	2.1 (±2.4)
F. V. channel	3.1 (±4.8)	9.0 (±0.5)	2.1 (±0.8)	3.5 (+2.0/−1.3)	5.2 (±6.3)
T. channel	2.5 (±4.9)	8.8 (±0.5)	3.22 (±2.0)	0.5 (+0.4/−0.2))	0.9 (±1.1)

[a] Standard deviations and 95% confidence intervals are given in parentheses for mean and geometric mean values, respectively.

in the channel network due to the tidally driven oscillation of water flow in the drainage channels.

By integrating the conductivity depressions evident in Figure 2 (see Methods and Materials), we estimate that the volume of runoff flowing into the ocean at the PCH Station during the first 8 days and last 7 days was 5000 m³ and 4000 m³, respectively. Furthermore, we estimate the amount of flow entering the upper reaches of the channels at the Fountain Valley and Talbert Stations to be approximately 8000 m³ (first 8 days) and 7000 m³ (last 7 days), and we estimate the amount of runoff discharged from pump stations the last 7 days of the study to be 16 000 m³. Hence, the net inflow and outflow of runoff roughly balance during the first 8 days (8000 and 5000 m³, respectively), while the net inflow and outflow of runoff do not balance during the last 7 days (22 000 and 4000 m³, respectively). These volume estimates support the conclusion that the majority of the pump station water discharged in the last 7 days of the study was trapped in the channel network. Importantly, the 7000 m³ per week of runoff continuously entering the drainage channels from the upper reaches of the Talbert Watershed had relatively little impact on downstream water quality, at least compared to the ENT signal generated by the Talbert Marsh. Die-off of ENT and the relatively long residence time (~1 week) of runoff in the drainage channels may limit the downstream impact of urban runoff (*33–35*). The fate and transport of bacterial pollutants in the drainage system at Huntington Beach is a subject of ongoing investigations.

ENT Source Study: Sediment and Vegetation. Sediment cores were collected from May 22 to June 6, 2000 along a set of transects (dotted lines in Figure 1A) located both in the marsh and surf zone. ENT levels in the sediment cores are consistent with the marsh being a significant source of these bacteria. Nineteen percent of sediment samples from the marsh ($n = 96$) were positive for ENT, compared to 2% of the sediment samples from the surf zone ($n = 121$). A total of 65% of the surface sediment samples in the marsh were positive for ENT. Vertical profiles of ENT in the marsh sediments indicate that the bacteria are concentrated in the top 1 cm of the cores (Figure 6). The largest concentration of ENT in the sediment cores (50 000 MPN/100 g) was from a surface sample collected from the northeast corner of the marsh. Of the sediment collected from the surf zone, only one sample had significant levels of ENT (800 MPN/100 mL), and this was a surface sample collected directly upcoast of the Talbert Marsh outlet.

High levels of ENT, ranging from 18 to 450 000 MPN/100 g (geometric mean of 2284 MPN/100 g, $n = 9$), were also found on seaweed collected from the marsh. The fact that sediments and vegetation are enriched in ENT suggests that these organisms are surviving, and perhaps even growing, in the marsh environment. Marine vegetation supports the

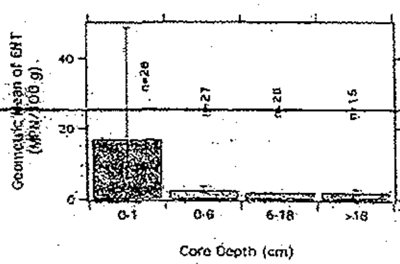

FIGURE 6. The vertical distribution of ENT in marsh sediments. Error bars represent 95% confidence intervals. The number of cores used to calculate the geometric mean values are indicated.

growth of certain strains of ENT in New Zealand, and estuarine sediments can apparently support the growth of ENT in tropical settings such as Hawaii and Guam (*21, 22*), although there are no published reports of this occurring in Mediterranean climates such as southern California.

ENT Source Study: Bird Feces. Bird feces are a significant source of ENT in the marsh environment. This conclusion was arrived at by measuring the ENT levels in the following: (1) marsh water alone, (2) 500 mL of marsh water after addition of approximately 10 g of marsh sediment, and (3) 500 mL of marsh water after addition of approximately 10 g of marsh sediment containing bird feces that were either wet or dry at the time of collection. The concentration of ENT was below the detection limit (100 MPN/100 mL) in samples of pure marsh water and in marsh water containing feces-free sediment. However, when marsh water was exposed to sediment containing feces that were wet at the time of collection, the ENT concentrations ranged from 9090 to 24 192 000 MPN/100 mL ($n = 10$). Likewise, marsh water exposed to sediment containing feces that were dry at the time of collection had ENT concentrations ranging from 100 to 241 920 MPN/100 mL ($n = 10$). The geometric mean and 95% confidence intervals of the ENT measured in marsh water exposed to wet and dry feces were $1.8 \times 10^5 + 6.2 \times 10^5/-1.4 \times 10^5$ and $6.8 \times 10^2 + 3.3 \times 10^3/-5.6 \times 10^2$ MPN/100 mL, respectively. Expressing these geometric means and confidence intervals on a per feces basis, we obtain $8.9 \times 10^5 + 3.1 \times 10^6/-6.9 \times 10^5$ and $3.4 \times 10^4 + 1.6 \times 10^4/-2.8 \times 10^3$ MPN/feces for wet and dry feces, respectively.

The majority of the bird feces are deposited on low-lying mud flats in the marsh which become submerged to varying degrees during high tides. To determine if bird feces deposited in the marsh can account for the observed increase of ENT in water as it flows through the marsh, we performed a simple mass balance calculation as follows:

$$G = C_{out}Q_{out} - C_{in}Q_{in} \tag{3}$$

Here G is the rate of generation of bacteria in the marsh with units of [MPN/T], C_{out} and C_{in} are the concentrations of ENT at the outlet and inlet of Talbert Marsh, respectively, with units of [MPN/L^3], and Q_{out} and Q_{in} are the volumetric flow rates of water at the outlet and inlet of Talbert Marsh with units [L^3/T], where L and T represent length and time scales, respectively.

During ebb tides, in-situ measurements of flow velocity and water elevation at Brookhurst and PCH Stations indicate that the flow in and out of Talbert Marsh roughly balance so that $Q_{out} \approx Q_{in}$ and eq 3 simplifies as follows:

$$G = Q(\Delta C) \tag{4}$$

The parameter ΔC is the increase in ENT measured in water as it flows through Talbert Marsh.

Using average ebb tide values of $\Delta C = 29$ MPN/100 mL (see Figure 3E) and $Q = 8.37$ m^3/s from the calibrated hydrodynamic model, we estimate a generation rate for ENT in the marsh to be $G \approx 10^{10}$ MPN/h. Assuming each bird dropping has 10^6 MPN/feces (the geometric mean for wet bird feces), then 10^4 wet feces/h would be needed to account for the estimated generation rate. Our bird census indicates that, at most, 10^3 birds are present in the marsh, which corresponds to a deposition rate of more than 1 feces per bird every six minutes. If instead we use the maximum number of ENT liberated from the wet bird feces (10^8 MPN/feces) and the average number of birds present in the marsh during the day (228 birds), the deposition rate required decreases to approximately 1 feces per bird every 3 h. This latter deposition rate is comparable to rates observed for the same bird species in captivity, typically one dropping every 3 h (personal communication, J. Pavlat, Wildlife Care Facility, Huntington Beach, CA).

The above analysis does not consider the potential contributions of older, dried, bird feces, which were also found to contain significant levels of ENT. Portions of the mud flats in Talbert Marsh may remain exposed over many tide cycles, allowing the quantity of bird feces deposited there to increase. During a spring tide, when higher than average high tides occur, these older feces may become suspended in the marsh water and thereby increase the concentration of ENT in the water column. This idea is consistent with the fact that the highest level of ENT recorded at the Brookhurst and PCH Stations occurred during spring tides when the mud flats are most likely to be washed by tidal action (see Figure 2). Vegetation in the Talbert Marsh may also contribute to the levels of ENT in the water column, as could the growth of these organisms at the sediment/water interface. Indeed, growth at the sediment/water interface is supported by the distribution of ENT in cores taken from Talbert Marsh (see Figure 6). While bird droppings are clearly a significant source of ENT in the marsh, other sources may also contribute to the generation of ENT in the marsh including urban runoff, sediment, and vegetation.

Implications. ENT generated in the Talbert Marsh appear to be at least partially responsible for the frequency with which surf zone samples in Huntington State and City Beaches exceed state bathing water standards. This conclusion is based on two findings from our study: (i) ENT concentrations are increased above ENT standards (both single-sample and geometric mean) as water passes through the marsh and (ii) water flowing out of the marsh can be transported by littoral currents to the region of Huntington State and City Beaches where ENT standards are routinely exceeded. The ENT appear to enter the marsh from birds and runoff, and once there these organisms accumulate, and perhaps even grow, on marsh vegetation and sediments.

While ENT flowing into the surf zone during ebb tides may be responsible for beach postings that occur near the marsh outlet, the marsh is probably not the only source of ENT at Huntington State and City Beaches. During the summers of 1999 and 2000, for example, surf zone station 9N (see Figure 1) was frequently posted or closed (total of 70 days) due to elevated levels of ENT, even during periods of time when the concentration of ENT at stations near the Talbert Marsh outlet were relatively low (31). Given this spatial distribution of ENT, it is unlikely that the bacteria at 9N are coming solely from the Talbert Marsh, and their exact source is a matter of ongoing investigation. Indeed, we anticipate that the impact of marsh effluent on surf zone water quality will be relatively localized, given the factor two dilution that occurs as the marsh water mixes into the surf zone, and the fact that ENT die-off in ocean water (34, 35).

Based on the results presented in this paper, there may be a tradeoff between the restoration of coastal wetlands and compliance with marine water contact standards. This tradeoff could be ameliorated by specifically designing wetlands to remove bacteria from the water column. For example, freshwater wetlands remove bacterial pollutants most efficiently when the flow velocities are slow (<0.7 m/s) and the residence time of water is long (10 h) (36, 37). While the flow velocities in the Talbert Marsh are within the recommended range, the residence time of water (~1 h) is not. On the other hand, if there are no human health risks associated with ENT from wetland effluent, then marine water contact standards may need to be modified to account for the existence of both benign and nonbenign sources of these bacteria. An epidemiological study could help to define the human health risks associated with human exposure to nonanthropogenic sources of ENT such as marsh effluent. These issues are especially timely, as a Federal law has recently been enacted that mandates national monitoring and reporting of coastal water quality (38).

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