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A predictive model for microbial counts on beaches where intertidal sand is the primary source

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ABSTRACT

Human health protection at recreational beaches requires accurate and timely information on microbiological conditions to issue advisories. The objective of this study was to develop a new numerical mass balance model for enterococci levels on nonpoint source beaches. The significant advantage of this model is its easy implementation, and it provides a detailed description of the cross-shore distribution of enterococci that is useful for beach management purposes. The performance of the balance model was evaluated by comparing predicted exceedances of a beach advisory threshold value to field data, and to a traditional regression model. Both the balance model and regression equation predicted approximately 70% the advisories correctly at the knee depth and over 90% at the waist depth. The balance model has the advantage over the regression equation in its ability to simulate spatiotemporal variations of microbial levels, and it is recommended for making more informed management decisions.

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1. Introduction

In response to the U.S. Environmental Protection Agency (EPA) guidelines (US EPA, 1986, 2012) and federal laws, such as Beaches Environmental Assessment and Coastal Health Act (BEACH Act) of 2000, beach monitoring programs have been adopted and implemented around the U.S. coasts as well as the Great Lakes to protect beachgoers from health risks caused by potentially harmful bacteria. Implementation traditionally includes sparse water sampling with time-consuming laboratory analysis. For instance, under the Florida Healthy Beaches Program, all 34 coastal counties in the State of Florida collect beach water samples weekly, and report beach advisories on the basis of enterococci and fecal coliform measurements 24–48 h after sample collection. The water sample analysis is useful in terms of guiding beach warnings and advisories; however, due to the minimum one-day laboratory time requirement by the culture method and

the high spatiotemporal variability associated with fecal indicator bacteria (FIB) in the nearshore water (Boehm, 2007; Ge et al., 2012a; Enns et al., 2012), this method may not be timely and sufficient for decision making, thereby potentially causing unnecessary beach closures or human health risks for beaches that remain open. Recently, many beach managers have begun to utilize predictive tools, of which the most widely applied are models developed through multivariable linear regression (e.g., Olyphant, 2005; Nevers and Whitman, 2005; Frick et al., 2008). In addition, process-based models, which couple hydrodynamic models with a microbe transport-fate model involving microbial loading, transport and fate processes (e.g., Sanders et al., 2005; Hipsey et al., 2008; Feng et al., 2013; Thupaki et al., 2013) can in principle be used to make predictions. However, for beach managers or public health agencies, these process-based models are usually challenging to build and less accessible due to their intrinsic complexity and high computational demands.

In the past, pollutant transport models along with mass balance analysis have been utilized to quantify bacterial source loading rates from well-defined point sources (e.g., rivers and tidal inlets), and to predict FIB levels in the surf zone of marine beaches (Kim et al., 2004; Grant et al., 2005), as well as in the freshwater beaches

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of the Great Lakes (Thupaki et al., 2010). However, these models are not directly applicable to many beaches without known point sources.

In the U.S., about half of the beach closing/advisory days (12,596 and 11,588 days in 2010 and 2011, respectively) were attributed to unknown sources of pollution (National Resources Defense Council, 2011, 2012). Stormwater runoff with high bacterial levels has been identified as one type of nonpoint source of pollutants to beach water (e.g., Reeves et al., 2004; Ahn et al., 2005; Parker et al., 2010). More significantly, beach sediments have been found to be ubiquitous nonpoint sources of FIB (e.g., Whitman and Nevers, 2003; Shibata et al., 2004; Yamahara et al., 2007; Halliday and Gast, 2011; Byappanahalli et al., 2012) and also to harbor potentially pathogenic microbes (Goodwin et al., 2009; Shah et al., 2011; Yamahara et al., 2012). A multi-beach survey suggested that beaches with a relatively higher abundance of enterococci in the sand would also have higher exceedance rates (Phillips et al., 2011a).

In view of this, a model capable of simulating time-varying microbial levels at a nonpoint source beach is imperative, especially given the significance of intertidal sediments and stormwater observed in various studies. Only recently, mass balance models have been applied to estimate bacterial levels at nonpoint source beaches. Models developed from California beaches were used to identify sand and groundwater as major contributors to fecal indicator bacteria (Boehm et al., 2009; Russell et al., 2013).

The first and main objective of this study was to develop a new physics-based yet simple microbial mass balance model for nonpoint source beaches, taking into account loading, transport, and decay mechanisms. This balance model is a simplification of a prior process-based water quality model developed by Feng et al. (2013). The new balance model is not only computationally efficient, but also retains dominant physical and microbiological processes that govern the microbial balance in the nearshore water, where water quality monitoring and most recreational activities occur. As a case study, this model was optimized and validated using a 10-day monitoring dataset at a well-studied subtropical municipal beach near Miami, Florida, USA (Fig. 1). Multivariable linear regression equations that predict beach enterococci levels were also developed for comparative purposes. The second objective was to compare the balance model and regression equation in their ability to accurately predict beach advisories.

2. Materials and methods

2.1. Simplified microbial mass balance model

2.1.1. Simplification of model equations

The ultimate goal of this model is to predict time-varying microbial levels in order to guide water quality monitoring and beach advisories. We expressed the two-dimensional (2D) depth-averaged microbial mass balance with the advection–diffusion–reaction equation. This equation consists of terms representing microbial advection, diffusion, source loading, and a first-order biological decay (e.g., Sanders et al., 2005; Liu et al., 2006; Feng et al., 2013):

$$\frac{\partial hC}{\partial t} + \frac{\partial uhC}{\partial x} + \frac{\partial v hC}{\partial y} = \sum_{i=1}^{N_s} S_i \delta(x - x_s^i) \delta(y - y_s^i) + \frac{\partial}{\partial x} \left[\varepsilon_x h \frac{\partial C}{\partial x} \right] + \frac{\partial}{\partial y} \left[\varepsilon_y h \frac{\partial C}{\partial y} \right] - k_d h C \quad (1)$$

where C is the depth-averaged microbial concentration (in Colony Forming Unit, CFU or Most Probable Number, MPN per m^3), h is water depth (in m), and u and v are depth-averaged cross-shore and alongshore velocities (in $m s^{-1}$) respectively. The first term on

the right hand side (RHS) represents influx of microbes from sources of multiple types, where N_s is the number of source types, S_i is influx rate (in $CFU s^{-1}$) of the i th type, δ is the Kronecker delta function (in m^{-1}), and x_s^i and y_s^i are coordinates of the source points. ε_x and ε_y are diffusivities (in $m^2 s^{-1}$) in cross-shore and alongshore directions, and k_d is a first-order decay rate (in s^{-1}). Note that Eq. (1) may be applicable to all sorts of microorganisms (e.g. fecal bacteria, pathogen, and protozoan); in this study, we only focused on culturable enterococci, which are recommended by EPA for water quality indication and has been monitored for decades (US EPA, 1986, 2012).

It is not possible to directly solve Eq. (1) for microbial concentration without resolving detailed hydrodynamics (i.e., u , v , h , ε_x and ε_y) from momentum and continuity equations, so we simplified this equation based on the characteristics of the beach setting and hydrodynamic conditions outlined below.

Three transport-related assumptions can be made for many low-energy straight beaches: (1) the beach is quasi-uniform along-shore ($\frac{\partial}{\partial y} \approx 0$); (2) cross-shore velocity is negligible ($u \approx 0$); and (3) the diffusion is assumed isotropic and homogenous. The first assumption applies only to the central part of the beach away from the lateral boundaries. Assuming a straight long beach with neither significant longshore bathymetric variation nor flow convergence/divergence, the central portion of the beach can be considered longshore uniform if the microbial sources along the shoreline are more or less uniform. A uniform shoreline is characteristic of many low-energy beaches, and would allow for applying this simplification to areas where flow patterns are approximately the same along the shoreline. The second assumption is also a reasonable assumption for low-energy beaches where nearshore currents are very weak. This is supported by both theoretical analysis of an ideal enclosed beach (Grant and Sanders, 2010), and the observations and other modeling efforts at the beach of this study (Zhu et al., 2011; Fiorentino et al., 2012; Feng et al., 2013). Finally, since the majority of the fecal bacteria originate from the beach shoreline and intertidal zone, there must be cross-shore gradients of bacterial levels. In this study, the diffusion process consisted of both turbulent diffusion and tidal dispersion, and a constant diffusivity, $0.03 m^2 s^{-1}$, was used based on previous modeling efforts (Feng et al., 2013).

On the basis of the above assumptions, Eq. (1) can be reduced to a one-dimension (1D) equation:

$$\frac{\partial hC}{\partial t} = \sum_{i=1}^{N_s} S_i \delta(x - x_s^i) + \frac{\partial}{\partial x} \left[\varepsilon_x h \frac{\partial C}{\partial x} \right] - k_d h C \quad (2)$$

where S_i represents source influx per meter shoreline (in $CFU m^{-1} s^{-1}$).

2.1.2. Quantification of source loads and decay

Our model took into account the contributions of two types of nonpoint sources: beach sediment and stormwater runoff. The influx terms of two sources are empirically provided by:

$$\sum_{i=1}^{N_s} S_i \delta(x - x_s^i) = \alpha \beta H_1^2 \delta(x - x_s^1) + \gamma I_r \delta(x - x_s^2) \quad (3)$$

where α is a wave pick-up coefficient (in $CFU m^{-3} s^{-1}$), β is a dimensionless tidal modulation factor, and γ is rainwater-runoff loading coefficient (in $CFU m^{-2}$). H_1 represents significant wave height (in m) at a reference depth and I_r is the rainfall rate (in $m s^{-1}$). In addition, the source zones (x_s^1 and x_s^2) for both sediment and runoff loading were considered to be a 10 m wide cross-shore transect from the waterline to offshore (see Fig. 1c red dots), equivalent to a typical width of intertidal zone at the study beach. Notice that the

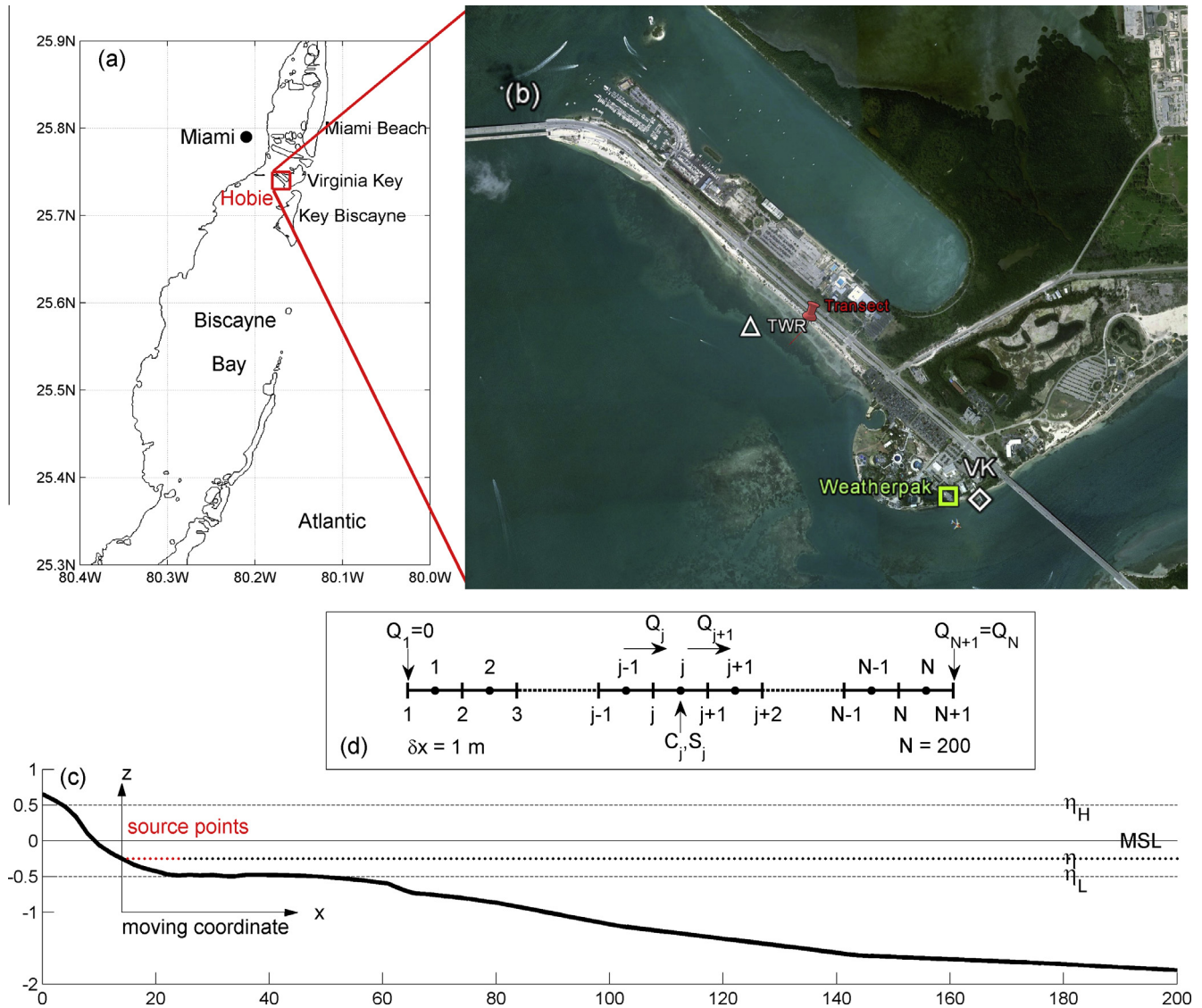


Fig. 1. The study site, beach profile and model grid. (a) Geographical location of Hobie Beach, located on Virginia Key, in the northern Biscayne Bay, and to the southeast of Miami, FL. (b) Google Earth aerial photo of the beach (imagery date of March 31, 2010), which is 1600 m long and northwest-southeast oriented. The positions of tide and wave recorder (TWR), meteorological station (Weatherpak), and NOAA Virginia Key station (VK) are indicated by the triangle, square, and diamond symbols, respectively. The approximate location of water sampling transect is illustrated by a red pushpin. (c) Cross-shore beach profile (heavy solid line) and model coordinate. The coordinate origin is always set at the waterline, moving with the tides. The mean sea level (MSL), reference supratidal (η_H) and subtidal (η_L) lines are illustrated by light solid and dashed lines. The actual tidal elevation (η) is indicated by the black dots with each dot as a grid point. The first ten grid points (red dots) are microbial source points in the model. (d) Illustration of the 1D staggered grid used by the numerical computation. The grid is equally spaced ($\Delta x = 1$ m). The microbial level (C_j) and source term (S_j) are positioned at the cell centers ($j = 1, 2, \dots, N-1, N$) whereas flux terms (Q_j) are at the cell edges ($j = 1, 2, \dots, N, N+1$) and N is the total number of grid cells. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

source points in this model were not fixed because the waterline changes with rising and falling tides.

The first term on the RHS of Eq. (3) quantified the microbial influx from the sand due to the wave effect under the influence of tides. Waves have been found to be an important agent to transport microbes shoreward or directly release them from the sand (Ge et al., 2012a, 2012b; Feng et al., 2013; Thupaki et al., 2013). Since wave-related bed shear stress suspends beach sands and releases attached microbes, we assumed that the influx of microbes from the sands is linearly proportional to wave energy, or squared significant wave height. The above assumption also implies that bed sediments are infinite sources so that losses of enterococci are continuously compensated by settlement and/or regrowth of enterococci in the sediment. The coefficient α is a fitting parameter, determined by minimizing least square errors

between predictions and observations (see Supplement). The reference depth was set at 1 m, consistent with typical waist depth where water samples are routinely collected. Measured wave heights (H_0) are converted to the reference depth using a shoaling factor (K_s). The wave height at reference depth (H_1) is:

$$H_1 = H_0 K_s = H_0 \sqrt{\frac{C_{g0}}{C_{g1}}} \quad (4)$$

where C_{g0} and C_{g1} are wave group velocities at instrument and reference depths, derived from the linear wave theory assuming that incident wave direction is normal to the shoreline (Dean and Dalrymple, 1991). In addition, a minimal wave height of $H_{1,\min} = 0.05$ m is imposed to represent a background enterococci loading due to processes not present in the model, such as tidal

washing and groundwater discharge (Boehm and Weisberg, 2005; Phillips et al., 2011b; Russell et al., 2012, 2013; Feng et al., 2013).

The tidal modulation factor β was introduced to account for the fact that microbial levels in the sand decrease from supratidal to subtidal zones, as observed in many beaches (e.g., Whitman and Nevers, 2003; Yamahara et al., 2007; Wright et al., 2011; Piggot et al., 2012). We assumed this factor has a maximum value of 1 at a reference supratidal point in the permanently dry upper beach, a minimum value of 0 at a reference subtidal point (i.e., permanently wet), and is linearly interpolated in between

$$\beta = \begin{cases} 1, & \text{if } \eta \geq \eta_H \\ (\eta - \eta_L)/(\eta_H - \eta_L), & \text{if } \eta_L < \eta < \eta_H \\ 0, & \text{if } \eta \leq \eta_L \end{cases} \quad (5)$$

where η_H (= 0.5 m) and η_L (= -0.5 m) denote elevations of the supratidal and subtidal reference lines (see Fig. 1c).

The second term on the RHS of Eq. (3) represents loading flux of the stormwater runoff as a prior study observed high levels of fecal indicator bacteria in runoff water at this beach (Wright et al., 2011). The coefficient γ ($=2.6 \times 10^8$ CFU m^{-2}) was determined based on the rational formula (Lindeburg, 1986) for estimating the runoff volume, coupled with a typical level of enterococci in the runoff (Wright et al., 2011). The derivation procedure of the coefficient value can be found in Feng et al. (2013).

The solar inactivation effect, represented by the last term on RHS of Eq. (2), was formulated according to a first-order exponential decay, in which the decay coefficient is linearly proportional to solar insolation (Sinton et al., 2002):

$$k_d = \kappa I_s \quad (6)$$

where the κ is the solar inactivation coefficient (in $\text{m}^2 \text{J}^{-1}$) and I_s is the solar insolation (in W m^{-2}). In this model, a constant $\kappa = 3.68 \times 10^{-7} \text{m}^2 \text{J}^{-1}$ was adopted based upon the work of Zhu et al. (2011).

The final balance model equation is:

$$\frac{\partial hC}{\partial t} = \alpha \beta H_1^2 \delta(x - x_s^1) + \gamma I_r \delta(x - x_s^2) + \frac{\partial}{\partial x} \left[\epsilon_x h \frac{\partial C}{\partial x} \right] - \kappa I_s h C \quad (7)$$

2.1.3. Numerical computation

Eq. (7) was discretized in a 1-D staggered grid (Fig. 1d). During the computation, the whole grid moved simultaneously with the tides so that model origin can always be placed at the waterline. This was done to circumvent the wetting and drying of grid points in the intertidal zone. The transformation from traditional fixed grid to the moving grid had negligible effects on the computations due to the slowly varying nature of local tide and its small tidal range (see Supplement).

To update microbial levels in time, i.e. solving the first term of Eq. (7), we used a fourth-order Runge–Kutta method (RK-4), chosen for its high accuracy and large stability region (Hundsdoerfer and Verwer, 2003). The updated microbial level was imposed with a minimum value of 1 CFU/100 mL, assumed to be the background enterococci level of the beach water. This was set due to the detection limit of traditional membrane filtration method using a typical dilution of 100 mL water. The time step was 15 s to meet the numerical stability criterion (see Supplement). The model was initialized from the minimum level of 1 CFU/100 mL in the entire model domain. We applied Dirichlet boundary condition at the closed waterline boundary assuming zero flux value (i.e., $Q_1 = 0$; see Fig. 1d). A Neumann-type offshore boundary was used by imposing zero flux gradients (i.e., $Q_{N+1} = Q_N$). In this case, offshore boundary allows microbes to leave the computational domain because the sources and high microbial levels are in the beach side, but not the ocean side. Additional details about the numerical

methods can be found in the Supplement. The total computational time for a 10-day period trial was less than 2 min on a laptop computer using computational codes written in Matlab (MathWorks, Natick, MA).

2.2. Multivariable linear regression equation

In this study, we utilized EPA's Virtual Beach (VB) version 2.3 (Frick et al., 2008; Cyterski et al., 2012) to construct the regression equations from a number of independent explanatory variables:

$$\log_{10} ENT = B_0 + \sum_{i=1}^n B_i V_i + e \quad (8)$$

where $\log_{10} ENT$ is \log_{10} -transformed enterococci level (in $\log_{10}(\text{CFU}/100 \text{ mL})$), n is the number of explanatory variables, V_i and B_i are i th explanatory variable and corresponding regression coefficient, and e is the residual error.

The candidate explanatory variables included: tidal elevation (η in m), squared wave height (H^2 in m^2), solar insolation (I_s in W m^{-2}), and 4-h antecedent cumulative rainfall (I_{4h} in mm). These variables were uncorrelated, satisfying an important assumption when conducting multi-regression analysis (Ge and Frick, 2007). Rather than significant wave height H or other transformations, we chose H^2 because it had the best correlation with dependent variable $\log_{10} ENT$. Interestingly, H^2 is also proportional to wave energy, which likely influences the bacterial release process. Note that although enterococci levels at the study beach can be impacted by a variety of other variables (such as turbidity, wind, animal and human sources), these four variables were reasonably satisfactory for building regression equations and comparing the performance with the aforementioned microbial balance model.

An exhaustive search on all variable combinations was performed, and the best-fit equation was chosen with the smallest Akaike information criterion or AIC value (Akaike, 1974; Cyterski et al., 2012), defined as:

$$AIC = 2p + N * \ln \sum_{i=1}^N (\log_{10} P_i - \log_{10} O_i)^2 \quad (9)$$

where p is the number of variables, N is the number of observations, and P and O represent model-predicted and field-observed enterococci levels. AIC includes a penalty associated with increasing number of variables, and hence discourages over-fitting.

2.3. Field dataset

The data utilized for model training and evaluation purposes were collected in June 1st to 11th, 2010. Water was sampled hourly at knee depth (~ 0.3 m) and every 6 h at waist depth (~ 1.0 m). Enterococci levels of the water samples were measured by membrane filtration method. Tide and wave conditions were measured by a bottom-mounted tide and wave recorder (RBR TWR-2050, Ottawa, ON, Canada). Solar insolation and rainfall rate were recorded by a research weather station (Weatherpak, Seattle, WA) every 2 min at the University of Miami Rosenstiel School campus (Fig. 1b). The model inputs of solar insolation and rainfall rate were hourly moving averages of the raw measurements. The details of this experiment were described in Enns et al. (2012) and Feng et al. (2013).

The whole dataset was split into two subsets, one from 1300 (Eastern Daylight Time or EDT hereafter) June 1st to 2300 June 7th and the other after 0000 June 8th. The first subset was used to identify the optimized coefficient α ($= 1.645 \times 10^4$ CFU $\text{m}^{-3} \text{s}^{-1}$) for the balance model (see Supplement) and also to build regression equation. The second subset was then used to validate both the balance

model and regression equation. Because enterococci levels have high spatial variability, we did not attempt to aggregate the observations at different locations to construct a unified regression equation. Instead, we built the regression relationships for enterococci levels of the knee and waist depths separately.

2.4. Surrogates for model input

Typically, in situ tide and wave measurements are unavailable and therefore surrogates are required. Tides can be acquired from nearby National Oceanic and Atmospheric Administration (NOAA) tidal gauges as water level observations or harmonic tidal predictions (<http://tidesandcurrents.noaa.gov/>). Here, we obtained tidal records at the Virginia Key station which is very close to the beach (see Fig. 1b). In lieu of direct wave measurements, nearshore wave models (e.g., Ge et al., 2010) or, in some particular cases, wind observations can be used. For the study site, a good correlation between onshore wind speed and observed wave height was found ($r = 0.718$, $p < 0.001$). This correlation can be explained by the fact that the beach is mainly impacted by locally generated wind waves, and the onshore wind direction is approximately aligned with the longest wind fetch within the Biscayne Bay. Using the wind and wave measurements, we fitted a linear curve between onshore wind speed and significant wave height (see Supplement):

$$H_1 = \begin{cases} 0.02, & \text{if } u_c < 0 \text{ m s}^{-1} \\ 0.02u_c + 0.02, & \text{if } u_c \geq 0 \text{ m s}^{-1} \end{cases} \quad (10)$$

where u_c is the cross-shore wind speed (in m s^{-1} , positive onshore and negative offshore). Hence, in the absence of direct wave observations or modeling, Eq. (10) can be applied to retrieve wave height from the wind at this beach.

2.5. Statistical analyses

All enterococci levels were \log_{10} -transformed ($\log_{10}ENT$) to achieve normality. One-way analysis of variance (ANOVA) was conducted using the Matlab statistical toolbox to compare significant differences of the model and observational results.

Model performances were assessed by two typical statistical measures, mean absolute error (MAE) and root mean square error (RMSE), which are dimensional measures of average model-performance errors (Willmott and Matsuura, 2005).

$$MAE = \sum_{i=1}^N |\log_{10}P_i - \log_{10}O_i|/N \quad (11)$$

$$RMSE = \left[\sum_{i=1}^N (\log_{10}P_i - \log_{10}O_i)^2 / N \right]^{1/2} \quad (12)$$

2.6. Beach advisory assessment

Based on EPA's single sample threshold, an exceedance (or a positive outcome) occurs when sampled or predicted enterococci level exceeds 104 CFU/100 mL (US EPA, 1986). Type-I error (or a false positive outcome) occurs when the predicted enterococci level is above threshold value but the actual water sample is below the threshold. On the contrary, Type-II error (or a false negative outcome) means that predicted enterococci level is below 104 CFU/100 mL but the actual water sample exceeds this value, which may result in public exposure to microbial contamination when beaches are still open. Three other metrics that evaluate model's performance in predicting advisories are (Cyterski et al., 2012):

$$\text{Accuracy} = (N_{TP} + N_{TN})/N \quad (13)$$

$$\text{Specificity} = N_{TN}/(N_{TN} + N_{FP}) \quad (14)$$

$$\text{Sensitivity} = N_{TP}/(N_{TP} + N_{FN}) \quad (15)$$

where N , N_{TP} , N_{TN} , N_{FP} , and N_{FN} are numbers of observations, true positives, true negatives, false positives, and false negatives. Accuracy is the percentage of correct advisory predictions. Specificity and sensitivity are rates of correctly predicted non-exceedance and exceedance, respectively. All three metrics range from 0 to 1, with 1 being perfect.

3. Results

3.1. Microbial balance model

Model outcomes of enterococci levels using direct measurements of waves, tides, rainfall, and solar radiation demonstrated substantial spatiotemporal variations from the shoreline to 200-m offshore (Fig. 2e). The spatiotemporal patterns of enterococci levels agreed well with those shown by a much more sophisticated process model in Feng et al. (2013). In addition, the contour of $\log_{10}ENT$ calculated using model input surrogates (Fig. 2f) seemed almost identical to that using measurements and their values were highly correlated ($r = 0.995$). This suggests that at the study site, wave and tide surrogates are feasible and satisfactory even without in situ measurements.

The most important spatial pattern was the substantial decrease of enterococci levels from the waterline to offshore boundary. For both model and observation, mean $\log_{10}ENT$ of knee depth were more than one order of magnitude higher than waist depth ($p < 0.001$; Table 1). Such a spatial pattern is due to enterococci release at the shoreline source zone with subsequent offshore transport and dilution.

The most apparent temporal pattern was the diurnal variation, with lower enterococci levels in the daytime and higher in the nighttime, mainly resulting from sunlight inactivation during the day (Fig. 2d). Mean daytime $\log_{10}ENT$ was significantly lower than the nighttime ($p < 0.001$; Table 1). Another temporal pattern was associated with the semidiurnal tidal fluctuations (Fig. 2b). Enterococci levels during the high tides were significantly higher than those during the low tides ($p < 0.001$; Table 1). In addition, enterococci levels in the ebb phase were slightly higher than those in the flood phase, although not very significant ($p = 0.11$ for observation and $p = 0.019$ for model).

Elevated enterococci levels occurred with the presence of local wind waves, shown as red patches in the contour plots (Fig. 2e and f). Mean enterococci levels during large waves were higher than small or no wave conditions (Table 1). The significance level calculated from the observation was not as high as the model, mainly due to the larger standard deviation associated with the observation. Meanwhile, rainfall only functioned as an intermittent source. For instance, the maximum enterococci-loading rate during a summer afternoon thunderstorm with the peak rainfall rate of 10 mm h^{-1} (Fig. 2c) could reach $722 \text{ CFU m}^{-1} \text{ s}^{-1}$. This rate was more than half of the loading rate ($1332 \text{ CFU m}^{-1} \text{ s}^{-1}$) from the sand in a condition of largest observed wave height (of 0.3 m) and spring high tide (of 0.4 m). However, because precipitation in the 10-day period was sporadic and short-lived (Fig. 2c), rainfall was of secondary importance compared to the release of bacteria from sand through wave and tidal actions.

Model and observation agreed particularly well in the middle part of the 10-day period (Fig. 2g). However, the balance model generally underpredicted enterococci levels in the first and last two days when wave and rainfall were predominantly small to

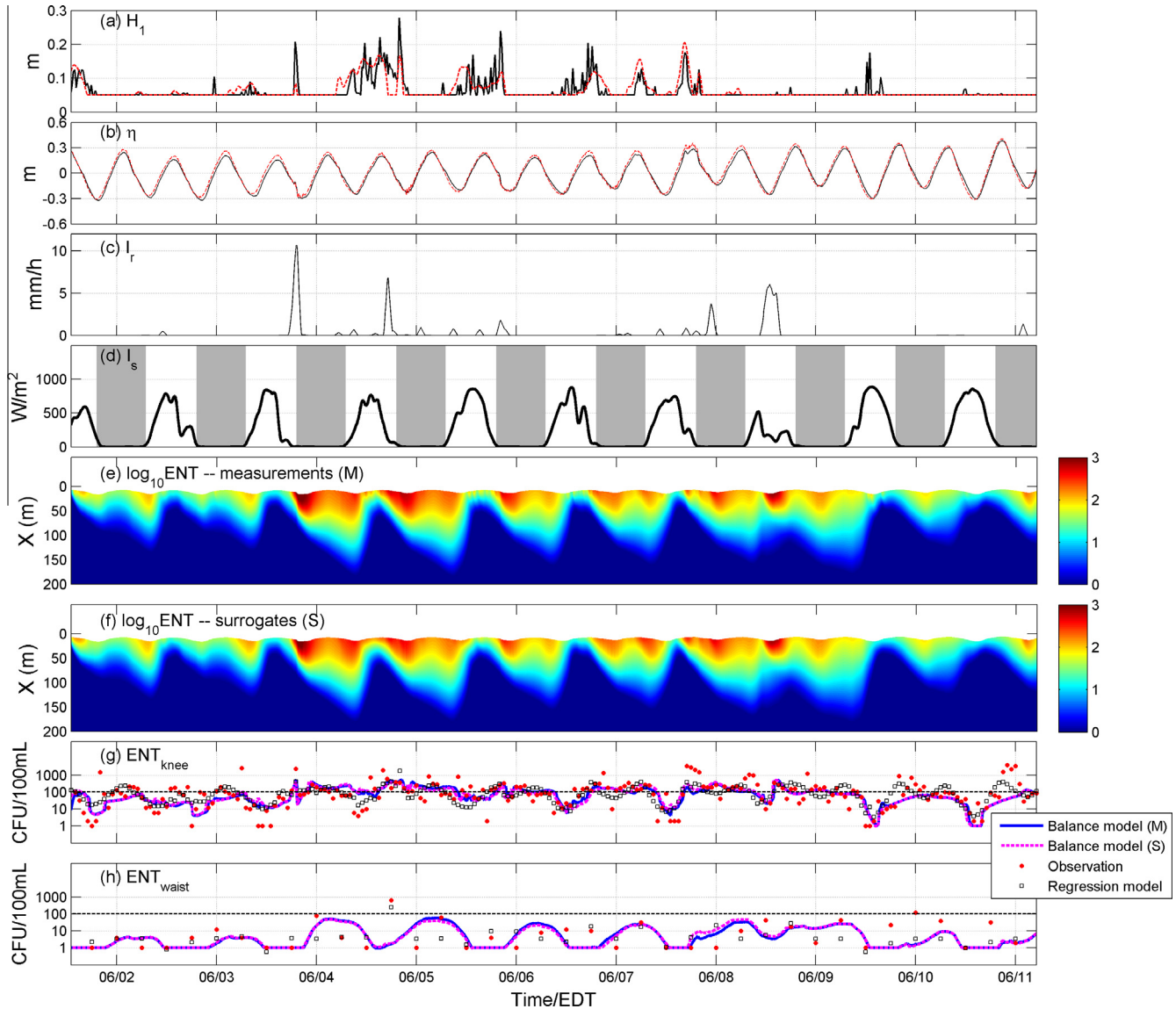


Fig. 2. Microbial balance model input variables and output of enterococci levels from June 1st to 11th, 2010. (a) Measured (black line) and wind-retrieved (red line) significant wave heights. (b) Tides in situ (black line) and at a nearby NOAA station (red line). (c) Hourly moving-averaged rainfall rates. (d) Hourly moving-averaged solar insolation. The grey stripes indicate nighttime conditions. (e) Contour of $\log_{10}ENT$ using direct wave and tide measurements. This figure illustrates the cross-shore transect from waterline to about 200 m offshore. Color bar to the right uses the unit of $\log_{10}(CFU/100\text{ mL})$. (f) Contour of $\log_{10}ENT$ using wind-retrieved wave heights and NOAA tides as model input surrogates. (g) Comparisons of balance model, regression equation and observation at knee-depth locations. Blue solid and magenta dashed lines are balance model outcomes based on measurements (M) and surrogates (S), respectively. Red dots and black squares show observations and regression equation results. Black dashed line is the EPA single sample threshold, 104 CFU/100 mL. (h) Comparisons at waist-depth locations. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

minimal. Also, observed enterococci levels were highly variable with extreme values of the knee-depth samples up to thousands of CFU/100 mL, whereas the modeled maximum enterococci level of knee-depth water never exceeded 600 CFU/100 mL. Such difference was also shown in the box-whisker diagram that observed enterococci levels had a nearly one order of magnitude higher upper whisker than modeled ones (Fig. 3e).

The model performance was further evaluated with statistical measures. The subsets for model training and validation had very similar error values (Table 2). The MAE and RMSE of the knee depth were around 0.5 and 0.65, respectively. In addition, slightly larger errors were found in waist-depth predictions. All these results indicated an approximately half an order of magnitude average error existing in enterococci predictions by the balance model.

3.2. Regression equation

The multivariable linear regression equations to predict the knee- and waist-depth enterococci levels based on Subset-1 can be written as:

$$\log_{10}ENT_{\text{knee}} = 1.9632 + 1.6976\eta + 20.5628H_1^2 - 0.001058I_s + 0.03931I_{4h} \quad (16)$$

$$\log_{10}ENT_{\text{waist}} = 0.5298 + 45.1858H_1^2 - 0.0009412I_s + 0.1486I_{4h} \quad (17)$$

Model fit and variable statistics for two equations above were shown in Table 3. The regression equation could predict 35% and 33% of the variance of knee and waist depths, respectively. For

Table 1

Comparisons of mean (and standard deviation) log₁₀-transformed enterococci levels of knee- versus waist depth, day versus night, high versus low tide, ebb versus flood tide, and large versus small wave.

	Sample	Model
Knee-depth (N = 233)	1.85 (±0.76)	1.76 (±0.52)
Waist-depth (N = 38)	0.73 (±0.77)	0.62 (±0.57)
One-way ANOVA	<i>p</i> < 0.001	<i>p</i> < 0.001
Day (N = 114)	1.54 (±0.82)	1.64 (±0.55)
Night (N = 119)	2.14 (±0.56)	2.00 (±0.41)
One-way ANOVA	<i>p</i> < 0.001	<i>p</i> < 0.001
High-tide (N = 32)	2.24 (±0.67)	1.79 (±0.31)
Low-tide (N = 36)	1.22 (±0.73)	1.17 (±0.67)
One-way ANOVA	<i>p</i> < 0.001	<i>p</i> < 0.001
Ebb-tide (N = 120)	1.93 (±0.81)	1.94 (±0.49)
Flood-tide (N = 113)	1.77 (±0.70)	1.68 (±0.54)
One-way ANOVA	<i>p</i> = 0.11	<i>p</i> = 0.019
Large-wave (N = 25)	2.07 (±0.87)	2.07 (±0.45)
Small-wave (N = 208)	1.82 (±0.74)	1.72 (±0.52)
One-way ANOVA	<i>p</i> = 0.12	<i>p</i> = 0.001

Notes: The unit is log₁₀(CFU/100 mL). The day sample is defined as a sample collected from 7:00 am and 6:59 pm (EDT), when the beach is open to the public. The sample taken after 7:00 pm and before 6:59 am next morning is defined as a night sample. High- (or low-) tide sample is collected when tidal elevation is above 0.2 m (or below −0.2 m) with respect to MSL. Ebb- (or flood-) tide sample is a sample collected at a time when tide is falling (or rising). The cutoff wave height to distinguish large/small wave is 0.1 m. For inter-comparison purposes, model outputs were subsampled to the time when water samples were collected. For day/night, ebb/flood, high/low tide, and large/small wave comparisons, only knee depth data was utilized.

the knee-depth relationship, all four explanatory variables were statistically significant. Surprisingly, tide was not significant in waist-depth relationship. Due to the limited number of samples for regression fitting in waist depth (*N* = 25), the best-fit relationship in Eq. (17) was rather crude and may not capture all significant variables.

3.3. Application in beach advisories

Predicted enterococci levels were visualized against measured values in scatterplots to analyze beach advisory decisions according to EPA's single-sample enterococci criteria of 104 CFU/100 mL (Fig. 3 and Table 4). 43.8% of the total knee-depth samples exceeded the threshold, whereas only 5.3% of the waist-depth samples were above the threshold. Note that the study beach is only open to the public during the daytime from sunrise to sunset according to the regulation of Miami-Dade County, which implies that only advisories using day samples/predictions may be directly relevant to human health protection. In a scenario only accounting for day samples/predictions, lower rates of advisories would be issued no matter whether observations or models are utilized.

The balance model and regression equation yielded accuracy values of 0.691 and 0.648 using knee-depth samples (Table 4). If only daytimes or beach open hours are evaluated, both methods performed better, with accuracy values of 0.719 (balance model) and 0.763 (regression equation), respectively. Waist-depth enterococci levels rarely exceeded the EPA threshold. Notice that neither the balance model nor regression equation can circumvent false positive/negative predictions at the knee depth due to the fact that transient enterococci levels have much larger variability than environmental variables in time and space. Both methods were better in predicting non-exceedances (specificity >0.7) than exceedances (sensitivity <0.6).

4. Discussion

4.1. Model implications for beach management

One major advantage of the balance model compared to the regression equation is that it provides a synoptic view of spatiotemporal distributions of microbial levels in the nearshore water, rather than only predicting microbial levels at relatively fixed sampling locations (for instance, knee or waist depth). Our results confirmed a prior study suggesting that spatial and temporal variations in indicator microbes substantially impact management decisions at recreational beaches (Enns et al., 2012). The high spatiotemporal variability of FIB is not unique at the study beach, but can also be found at other beaches (e.g., Boehm et al., 2002; Boehm, 2007; Ge et al., 2012a).

A prior study at this beach not only identified a variety of pathogens (e.g., yeasts, pathogenic bacteria, protozoa, viruses, and helminthes) in the sand, but also found significant correlations between some indicator microbes and pathogens (Shah et al., 2011). Moreover, epidemiologic studies demonstrated positive dose-response relationships of reported skin illness associated with both the enterococci levels (Fleisher et al., 2010; Sinigalliano et al., 2010) and 24-h antecedent rainfall (Sinigalliano et al., 2010). Another study at 7 U.S. beaches suggested that sand contact activities were positively associated with gastrointestinal illness and diarrhea (Heaney et al., 2009). Our model results showed a substantial increase of enterococci loading to the beach water occurs when local wind waves coincide with high tides. Therefore, it is possible that wave-induced sediment resuspension may release pathogens from the sand reservoir. If that is the case, these environmental phenomena should be taken into account in regular beach monitoring practices.

Abdelzaher et al. (2013) proposed a framework of Comprehensive Toolbox within an Approval Process (CTBAP), with the objective of designing site-specific beach regulation for better public health protection, and for ensuring protection consistency nationwide. The model approach developed in this study can be readily incorporated as a component in this suggested framework. The model may serve as a nowcast tool to provide an independent estimate of enterococci distributions. This model may also be used as a tool to examine beach responses to various “worst-case” scenarios. For example, an overnight heavy rainfall event, although not occurring in study period, will likely flush large amounts of enterococci onto the beach water. Our model can provide a quick estimation of enterococci condition in the early morning, which may help the local beach managers in the decision process. These scenario tests will facilitate decision making in the case of extreme events such as thunderstorms. Last but not least, the model outcome may be useful in guiding sampling strategy (e.g., sampling time and location).

The present balance model may have potential use in estimating counts of microorganisms by culture-independent methods, notably quantitative polymerase chain reaction (qPCR), the next-generation microbiological assay that has been advocated in the new recreational water quality criteria (US EPA, 2012). However, the response of qPCR counts to environmental conditions may be different from culture-based counts (Boehm et al., 2009; Byappanahalli et al., 2010; Sassoubre et al., 2012). Therefore, applying this type of model, based on an advection–diffusion–reaction equation and justified simplification, to predict culture-independent microbiological counts requires further model refinement, parameter tuning and validation using emerging qPCR monitoring datasets.

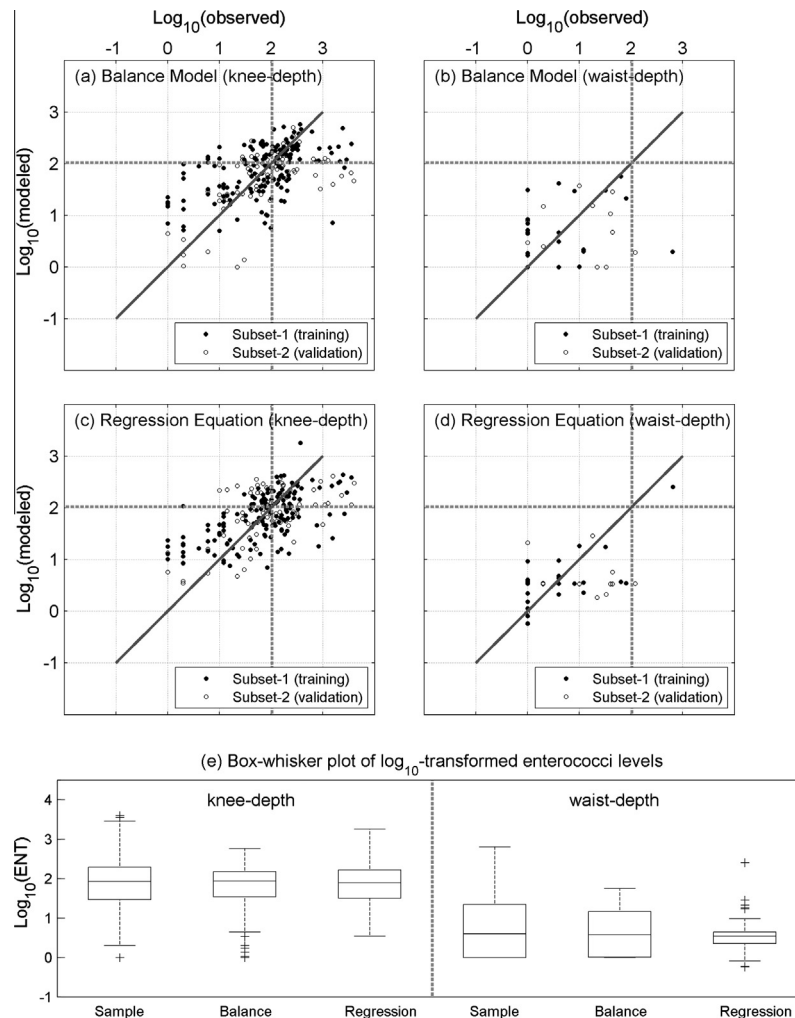


Fig. 3. Scatter and box-whisker plots showing observed versus modeled enterococci levels at the knee- and waist-depth locations. (a) Balance model results at knee depth. Dash line is EPA-recommended single sample threshold of 104 CFU/100 mL and solid line is where model and observation have perfect agreement. (b) Balance model results at waist depth. (c) Regression equation results at knee depth. (d) Regression equation results at waist depth. (e) Box-whisker plots of observed and modeled enterococci levels. On each box, the central line is the median, the edges of the box are the 25th and 75th percentiles, the whiskers extend to the most extreme data points not considered outliers, and outliers are plotted with crosses.

Table 2
Assessment of balance model and regression equation performance using the mean absolute error (MAE) and root mean square error (RMSE). See Eqs. (11) and (12) for definitions.

Subset	Location	Model vs. sample	MAE ($\log_{10}\text{CFU}/100\text{ mL}$)	RMSE ($\log_{10}\text{CFU}/100\text{ mL}$)
Subset-1 (training)	Knee ($N = 155$)	Balance	0.508	0.663
		Regression	0.474	0.604
	Waist ($N = 25$)	Balance	0.569	0.800
		Regression	0.452	0.565
Subset-2 (validation)	Knee ($N = 78$)	Balance	0.474	0.665
		Regression	0.490	0.605
	Waist ($N = 13$)	Balance	0.721	0.920
		Regression	0.739	0.892

4.2. Model application and limitations

The main difference between the balance model of this study and the prior process-based water quality model (Feng et al., 2013) is in the resolution of the detailed hydrodynamic and sediment transport processes. The balance model developed in this study takes an empirical approach to directly parameterize bacterial loading with wave and tidal parameters, which saves considerable time and computational resources. Due to the simplicity, however, the application of the balance model is

restricted to low-energy beaches without significant longshore topographic variations, usually situated in coastal embayments or large lakes. Moreover, sand and stormwater have to be prevailing sources of bacteria, which also implies that the beach is not directly affected by point sources of microbial pollution (such as rivers and sewage outfalls).

The under-prediction of enterococci levels by the balance model in last two days of the 10-day period (see Fig. 2) may suggest that additional sources, likely a thick wrack line observed at the study site, could contribute to the enterococci loading. Prior studies

Table 3

Regression equation parameters and fit statistics based on the Subset-1 data.

Model statistics	Explanatory variable	Coefficient	Standard error	p-Value
Knee (N = 155) AIC = 10.85 Adjusted R ² = 0.35	Constant	1.9632	0.0702	<0.001
	Tidal elevation (η)	1.6976	0.3196	<0.001
	Squared wave height (H_1^2)	20.5628	6.5487	0.002
	Solar radiation (I_s)	−0.001058	0.0002	<0.001
	4-h Rainfall (I_{4h})	0.03931	0.0203	0.0545
Waist (N = 25) AIC = 5.06 Adjusted R ² = 0.33	Constant	0.5298	0.1769	0.0069
	Squared wave height (H_1^2)	45.1858	24.2050	0.0760
	Solar radiation (I_s)	−0.0009412	0.0004	0.0340
	4-h Rainfall (I_{4h})	0.1486	0.0895	0.1118

Table 4

Summary of beach advisory evaluation based on water sample, balance model, and regression equation at the knee- and waist-depth locations.

Location	Method	Positive	Exceedance rate (%)	True Positive	True Negative	Type-I error False Positive	Type-II error False Negative	Accuracy	Specificity	Sensitivity
Knee (N = 233)	Sample	102	43.8	–	–	–	–	–	–	–
	Balance	86	36.9	58	103	28	44	0.691	0.786	0.569
	Regression	96	41.2	58	93	38	44	0.648	0.710	0.569
Knee (day only, N = 114)	Sample	31	27.2	–	–	–	–	–	–	–
	Balance	25	21.9	12	70	13	19	0.719	0.843	0.387
	Regression	26	22.8	15	72	11	16	0.763	0.867	0.484
Waist (N = 38)	Sample	2	5.3	–	–	–	–	–	–	–
	Balance	0	0	0	36	0	2	0.947	1	0
	Regression	1	2.6	1	36	0	1	0.974	1	0.5
Waist (day only, N = 19)	Sample	1	5.3	–	–	–	–	–	–	–
	Balance	0	0	0	18	0	1	0.947	1	0
	Regression	1	5.3	1	18	0	0	1	1	1

suggested that beach wrack is an FIB reservoir and may also provide favorable conditions for FIB survival and regrowth (Whitman et al., 2003; Shibata et al., 2004; Imamura et al., 2011). However, due to a lack of wrack quantification in the field study, the present model does not include the wrack source term. The inclusion of other potential sources into the model could improve model performance in the future.

Model adaption (e.g., parameter tuning) is a prerequisite when applying this model to other similar beaches or if the beach conditions are dramatically changed as a result of episodic events, such as beach renovations (Hernandez et al., 2014; Rippy et al., 2013) and tropical storms (Gast et al., 2011). For example, the release of microbes from the sand, tied to the wave pick-up coefficient α in Eq. (7), is controlled by both sediment and microbial characteristics of the sand (Feng et al., 2013). The sediment characteristics, such as grain size and distribution, affect sediment suspension and deposition (Soulsby, 1997; Reniers et al., 2013). The microbial characteristics (such as biofilm, mineral composition, abundance of microbes in the sand, and attachment of microbes to the sand grains) affect the amount of microbial release under external forcing (Yamahara et al., 2009; Piggot et al., 2012; Russell et al., 2012; Phillips et al., 2014). One particular example at the study site was a recent beach renovation activity in 2010 that significantly reduced enterococci levels within the sand after the renovation (Hernandez et al., 2014). Such a reduction would very likely result in less enterococci release from the shoreline sand to the beach water and thereby smaller wave pick-up coefficient, as indicated by the significant decrease of enterococci levels and exceedances in post-versus pre-renovation periods (Hernandez et al., 2014).

5. Conclusions

We have developed a simplified microbial mass balance model, based on the advection–diffusion–reaction equation, for low-

energy beaches where sands and stormwater are major nonpoint sources. This model has been applied to a typical beach to solve the time-varying cross-shore distribution of enterococci levels. In addition, multivariable linear regression equations were constructed. Both methods demonstrate comparable performance and skills in predicting the need for beach advisories. The advantage of the balance model is in its ability to simulate transient distributions of microbial levels in the nearshore water. Knowledge of this distribution provides added information to beach managers about the meaning of the measured microbial levels, which in turn can be used for making more informed decisions. Because of this advantage, we suggest the integration of this type of model into regular beach management practice as a nowcast modeling tool for beaches with similar conditions (i.e., low-energy straight beaches).

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.marpolbul.2015.03.019>.

References

- Abdelzaher, A.M., Solo-Gabriele, H.M., Phillips, M.C., Elmir, S.M., Fleming, L.E., 2013. An alternative approach to water regulations for public health protection at bathing beaches. *J. Environ. Public Health* 2013. <http://dx.doi.org/10.1155/2013/138521>.
- Ahn, J.H., Grant, S.B., Surbeck, C.Q., DiGiacomo, P.M., Nezin, N.P., Jiang, S., 2005. Coastal water quality impact of stormwater runoff from an urban watershed in southern California. *Environ. Sci. Technol.* 39 (16), 5940–5953.
- Akaike, H., 1974. A new look at the statistical model identification. *IEEE Trans. Autom. Control* 19 (6), 716–723.
- Boehm, A.B., 2007. Enterococci concentrations in diverse coastal environments exhibit extreme variability. *Environ. Sci. Technol.* 41 (24), 8227–8232.
- Boehm, A.B., Weisberg, S.B., 2005. Tidal forcing of enterococci at marine recreational beaches at fortnightly and semidiurnal frequencies. *Environ. Sci. Technol.* 39 (15), 5575–5583.
- Boehm, A.B., Grant, S.B., Kim, J.H., Mowbray, S.L., McGee, C.D., Clark, C.D., Foley, D.M., Wellman, D.E., 2002. Decadal and shorter period variability of surf zone water quality at Huntington Beach, California. *Environ. Sci. Technol.* 36, 3885–3892.
- Boehm, A.B., Yamahara, K.M., Love, D.C., Peterson, B.M., McNeill, K., Nelson, K.L., 2009. Covariation and photoinactivation of traditional and novel indicator organisms and human viruses at a sewage-impacted marine beach. *Environ. Sci. Technol.* 43 (21), 8046–8052.
- Byappanahalli, M.N., Whitman, R.L., Shively, D.A., Nevers, M.B., 2010. Linking non-culturable (qPCR) and culturable enterococci densities with hydrometeorological conditions. *Sci. Tot. Environ.* 408 (16), 3096–3101.
- Byappanahalli, M.N., Nevers, M.B., Korajkic, A., Staley, Z.R., Harwood, V.J., 2012. Enterococci in the environment. *Micobiol. Molec. Biol. Rev.* 76 (4), 685–706. <http://dx.doi.org/10.1128/MMBR.00023-12>.
- Cyterski, M., Galvin, M., Wolfe, K., Parmar, R., 2012. Virtual Beach v. 2.3 User Guide. Center for Exposure Assessment Modeling (CEAM), U.S. Environmental Protection Agency, Athens, GA.
- Dean, R.G., Dalrymple, R.A., 1991. Water wave mechanisms for engineers and scientists. World Sci.
- Enns, A.A., Vogel, L.J., Abdelzaher, A.M., Solo-Gabriele, H.M., Plano, L.R.W., Gidley, M.L., Phillips, M.C., Klaus, J.S., Piggot, A.M., Feng, Z., Reniers, A., Haus, B.K., Elmir, S.M., Zhang, Y., Jimenez, N.H., Abdel-Mottaleb, N., Schoor, M.E., Brown, A., Khan, S.Q., Dameron, A.S., Salazar, N.C., Fleming, L.E., 2012. Spatial and temporal variation in indicator microbe sampling is influential in beach management decisions. *Water Res.* 46 (7), 2237–2246. <http://dx.doi.org/10.1016/j.watres.2012.01.040>.
- Feng, Z., Reniers, A., Haus, B.K., Solo-Gabriele, H.M., 2013. Modeling sediment-related enterococci loading, transport and inactivation at an embayed non-point source beach. *Water Resour. Res.* 49, 693–712. <http://dx.doi.org/10.1029/2012WR012432>.
- Fiorentino, L.A., Olascoaga, M.J., Reniers, A., Feng, Z., Beron-Vera, F.J., MacMahan, J.H., 2012. Using Lagrangian Coherent Structures to understand coastal water quality. *Cont. Shelf Res.* 47, 145–149. <http://dx.doi.org/10.1016/j.csr.2012.07.009>.
- Fleisher, J.M., Fleming, L.E., Solo-Gabriele, H.M., Kish, J., Sinigalliano, C.D., Plano, L., Elmir, S.M., Wang, J.D., Withum, K., Shibata, T., Gidley, M.L., Abdelzaher, A.M., He, G., Ortega, C., Zhu, X., Wright, M., Hollenbeck, J., Backer, L.C., 2010. The BEACHES Study: health effects and exposures from non-point source microbial contaminants in subtropical recreational marine waters. *Int. J. Epidemiol.* 39 (5), 1291–1298. <http://dx.doi.org/10.1093/ije/dyq084>.
- Frick, W.E., Ge, Z., Zepp, R.G., 2008. Nowcasting and forecasting concentrations of biological contaminants at beaches: a feasibility and case study. *Environ. Sci. Technol.* 42 (13), 4818–4824.
- Gast, R.J., Gorrell, L., Raubenheimer, B., Elgar, S., 2011. Impact of erosion and accretion on the distribution of enterococci in beach sands. *Cont. Shelf Res.* 31 (14), 1457–1461. <http://dx.doi.org/10.1016/j.csr.2011.06.011>.
- Ge, Z., Frick, W.E., 2007. Some statistical issues related to multiple linear regression modeling of beach bacteria concentrations. *Environ. Res.* 103 (3), 358–364. <http://dx.doi.org/10.1016/j.envres.2006.11.006>.
- Ge, Z., Nevers, M.B., Schwab, D.J., Whitman, R.L., 2010. Coastal loading and transport of *Escherichia coli* at an embayed beach in Lake Michigan. *Environ. Sci. Technol.* 44 (17), 6731–6737. <http://dx.doi.org/10.1021/es100797r>.
- Ge, Z., Whitman, R., Nevers, M.B., 2012a. Nearshore hydrodynamics as loading and forcing factors for *Escherichia coli* contamination at an embayed beach. *Limnol. Oceanogr.* 57 (1), 362–381. <http://dx.doi.org/10.4319/lo.2012.57.1.0362>.
- Ge, Z., Whitman, R.L., Nevers, M.B., Phanikumar, M.S., 2012b. Wave-induced mass transport affects daily *Escherichia coli* fluctuations in nearshore water. *Environ. Sci. Technol.* 46 (4), 2204–2211. <http://dx.doi.org/10.1021/es203847n>.
- Goodwin, K., Matragrano, L., Wanless, D., Sinigalliano, C.D., LaGier, M.J., 2009. A preliminary investigation of fecal indicator bacteria, human pathogens, and source tracking markers in beach water and sand. *Environ. Res.* 2 (4), 395–417.
- Grant, S.B., Sanders, B.F., 2010. Beach boundary layer: a framework for addressing recreational water quality impairment at enclosed beaches. *Environ. Sci. Technol.* 44 (23), 8804–8813. <http://dx.doi.org/10.1021/es101732m>.
- Grant, S.B., Kim, J.H., Jones, B.H., Jenkins, S.A., Wasyl, J., Cudaback, C., 2005. Surf zone entrainment, along-shore transport, and human health implications of pollution from tidal outlets. *J. Geophys. Res.* 110 (C10), C10025. <http://dx.doi.org/10.1029/2004JC002401>.
- Halliday, E., Gast, R.J., 2011. Bacteria in beach sands: an emerging challenge in protecting coastal water quality and bather health. *Environ. Sci. Technol.* 45, 370–379.
- Heaney, C.D., Sams, E., Wing, S., Marshall, S., Brenner, K., Dufour, A.P., Wade, T.J., 2009. Contact with beach sand among beachgoers and risk of illness. *Am. J. Epidemiol.* 170 (2), 164–172.
- Hernandez, R.J., Hernandez, Y., Jimenez, N.H., Piggot, A.M., Klaus, J.S., Feng, Z., Reniers, A., Solo-Gabriele, H.M., 2014. Effect of full-scale beach renovation on fecal indicator levels in shoreline sand and water. *Water Res.* 48, 579–591.
- Hipsey, M.R., Antenucci, J.P., Brookes, J.D., 2008. A generic, process-based model of microbial pollution in aquatic systems. *Water Resour. Res.* 44 (7), 1–26. <http://dx.doi.org/10.1029/2007WR006395>.
- Hundsdoerfer, W., Verwer, J.G., 2003. Numerical Solution of Time-dependent Advection–diffusion–reaction Equations. Springer, Berlin, Germany, 471p.
- Imamura, G.J., Thompson, R.S., Boehm, A.B., Jay, J.A., 2011. Wrack promotes the persistence of fecal indicator bacteria in marine sands and seawater. *FEMS Microbiol. Ecol.* 77 (1), 40–49.
- Kim, J.H., Grant, S.B., McGee, C.D., Sanders, B.F., Largier, J.L., 2004. Locating sources of surf zone pollution: a mass budget analysis of fecal indicator bacteria at Huntington Beach, California. *Environ. Sci. Technol.* 38 (9), 2626–2636.
- Lindeburg, M.R., 1986. Civil Engineering Reference Manual, fourth ed. Prof. Pub., San Carlos, California.
- Liu, L., Phanikumar, M.S., Molloy, S.L., Whitman, R.L., Shively, D.A., Nevers, M.B., Schwab, D.J., Rose, J.B., 2006. Modeling the transport and inactivation of *E. coli* and enterococci in the near-shore region of Lake Michigan. *Environ. Sci. Technol.* 40 (16), 5022–5028.
- National Resources Defense Council, 2011. Testing the Waters 2011: A Guide to Water Quality at Vacation Beaches, 21st Annual Report.
- National Resources Defense Council, 2012. Testing the Waters 2012: A Guide to Water Quality at Vacation Beaches, 22nd Annual Report.
- Nevers, M.B., Whitman, R.L., 2005. Nowcast modeling of *Escherichia coli* concentrations at multiple urban beaches of southern Lake Michigan. *Water Res.* 39, 5250–5260.
- Olyphant, G.A., 2005. Statistical basis for predicting the need for bacterially induced beach closures: emergence of a paradigm? *Water Res.* 39, 4953–4960.
- Parker, J.K., McIntyre, D., Noble, R.T., 2010. Characterizing fecal contamination in stormwater runoff in coastal North Carolina, USA. *Water Res.* 44 (14), 4186–4194.
- Phillips, M.C., Solo-Gabriele, H.M., Piggot, A.M., Klaus, J.S., Zhang, Y., 2011a. Relationships between sand and water quality at recreational beaches. *Water Res.* 45 (20), 6763–6769. <http://dx.doi.org/10.1016/j.watres.2011.10.028>.
- Phillips, M.C., Solo-Gabriele, H.M., Reniers, A.J.H.M., Wang, J.D., Kiger, R.T., Abdel-Mottaleb, N., 2011b. Pore water transport of enterococci out of beach sediments. *Mar. Pollut. Bull.* 62 (11), 2293–2298.
- Phillips, M.C., Feng, Z., Vogel, L.J., Reniers, A.J.H.M., Haus, B.K., Enns, A.A., Zhang, Y., Hernandez, D.B., Solo-Gabriele, H.M., 2014. Microbial release from seeded beach sediments during wave conditions. *Mar. Pollut. Bull.* <http://dx.doi.org/10.1016/j.marpolbul.2013.12.029>.
- Piggot, A.M., Klaus, J.S., Johnson, S., Philips, M., Solo-Gabriele, H.M., 2012. Relationship between enterococci levels and sediment biofilms at recreational beaches in south Florida. *Appl. Environ. Microbiol.* 78 (17), 5973–5982.
- Reeves, R.L., Grant, S.B., Mrse, R.D., Copil Oancea, C.M., Sanders, B.F., Boehm, A.B., 2004. Scaling and management of fecal indicator bacteria in runoff from a coastal urban watershed in southern California. *Environ. Sci. Technol.* 38 (9), 2637–2648.
- Reniers, A.J.H.M., Gallagher, E.L., MacMahan, J.H., Brown, J.A., van Rooijen, A.A., van Thiel de Vries, J.S.M., van Prooijen, B.C., 2013. Observations and modeling of steep-beach grain-size variability. *J. Geophys. Res.* 118, 1–15. <http://dx.doi.org/10.1029/2012JC008073>.
- Rippy, M.A., Franks, P.J.S., Feddersen, F., Guza, R.T., Warrick, J.A., 2013. Beach nourishment impacts on bacteriological water quality and phytoplankton bloom dynamics. *Environ. Sci. Technol.* 47 (12), 6146–6154. <http://dx.doi.org/10.1021/es400572k>.
- Russell, T.L., Yamahara, K.M., Boehm, A.B., 2012. Mobilization and transport of naturally occurring enterococci in beach sands subject to transient infiltration of seawater. *Environ. Sci. Technol.* 46 (11), 5988–5996.
- Russell, T.L., Sassoubre, L.M., Wang, D., Masuda, S., Chen, H., Soetjito, C., Hassaballah, A., Boehm, A.B., 2013. A coupled modeling and molecular biology approach to microbial source tracking at Cowell Beach, Santa Cruz, CA, United States. *Environ. Sci. Technol.* 47 (18), 10231–10239.
- Sanders, B.F., Arega, F., Sutula, M., 2005. Modeling the dry-weather tidal cycle of fecal indicator bacteria in surface waters of an intertidal wetland. *Water Res.* 39, 3394–3408.
- Sassoubre, L.M., Nelson, K.L., Boehm, A.B., 2012. Mechanisms for photoinactivation of *Enterococcus faecalis* in seawater. *Appl. Environ. Microbiol.* 78 (21), 7776–7785.
- Shah, A.H., Abdelzaher, A.M., Phillips, M., Hernandez, R., Solo-Gabriele, H.M., Kish, J., Scorsetti, G., Fell, J.W., Diaz, M.R., Scott, T.M., Lukasik, J., Harwood, V.J., McQuaig, S., Sinigalliano, C.D., Gidley, M.L., Wanless, D., Ager, A., Lui, J., Stewart, J.R., Plano, L.R.W., Fleming, L.E., 2011. Indicator microbes correlate with pathogenic bacteria, yeasts and helminths in sand at a subtropical recreational beach site. *J. Appl. Microbiol.* 110 (6), 1571–1583.
- Shibata, T., Solo-Gabriele, H.M., Fleming, L.E., Elmir, S., 2004. Monitoring marine recreational water quality using multiple microbial indicators in an urban tropical environment. *Water Res.* 38 (13), 3119–3131.

- Sinigalliano, C.D., Fleisher, J.M., Gidley, M.L., Solo-Gabriele, H.M., Shibata, T., Plano, L.R.W., Elmir, S.M., Wanless, D., Bartkowiak, J., Boiteau, R., Withum, K., Abdelzher, A.M., He, G., Ortega, C., Kish, J., Zhu, X., Wright, M.E., Kish, J., Hollenbeck, J., Scott, T., Backer, L.C., Fleming, L.E., 2010. Traditional and molecular analyses for fecal indicator bacteria in non-point source subtropical recreational marine waters. *Water Res.* 44, 3763–3772.
- Sinton, L.W., Hall, C.H., Lynch, P.A., Davies-Colley, R.J., 2002. Sunlight inactivation of fecal indicator bacteria and bacteriophages from waste stabilization pond effluent in fresh and saline waters. *Appl. Environ. Microbiol.* 68 (3), 1122–1131.
- Soulsby, R.L., 1997. *Dynamics of Marine Sands*. Thomas Telford, London, UK.
- Thupaki, P., Phanikumar, M.S., Beletsky, D., Schwab, D.J., Nevers, M.B., Whitman, R.L., 2010. Budget analysis of *Escherichia coli* at a Southern Lake Michigan Beach. *Environ. Sci. Technol.* 44 (3), 1010–1016.
- Thupaki, P., Phanikumar, M.S., Schwab, D.J., Nevers, M.B., Whitman, R.L., 2013. Evaluating the role of sediment-bacteria interactions on *Escherichia coli* concentrations at beaches in southern Lake Michigan. *J. Geophys. Res. Oceans* 118, 1–17. <http://dx.doi.org/10.1002/2013JC008919>.
- US EPA, 1986. Ambient water quality criteria for bacteria. EPA 440/5-84-002. Washington DC, U.S. Environmental Protection Agency.
- US EPA, 2012. Recreational water quality criteria. Washington DC, U.S. Environmental Protection Agency.
- Whitman, R.L., Nevers, M.B., 2003. Foreshore sand as a source of *Escherichia coli* in nearshore water of a Lake Michigan beach. *Appl. Environ. Microbiol.* 69 (9), 5555–5562.
- Whitman, R.L., Shively, D.A., Pawlik, H., Nevers, M.B., Byappanahalli, M.N., 2003. Occurrence of *Escherichia coli* and enterococci in *Cladophora* (Chlorophyta) in nearshore water and beach sand of Lake Michigan. *Appl. Environ. Microbiol.* 69 (8), 4714–4719.
- Willmott, C.J., Matsuura, K., 2005. Advantage of the mean absolute error (MAE) over the root mean square error (RMSE) in assessing average model performance. *Clim. Res.* 30, 79–82.
- Wright, M.E., Abdelzher, A.M., Solo-Gabriele, H.M., Elmir, S., Fleming, L.E., 2011. The inter-tidal zone is the pathway of input of enterococci to a subtropical recreational marine beach. *Water Sci. Technol.* 63 (3), 542–549.
- Yamahara, K.M., Layton, B.A., Santoro, A.E., Boehm, A.B., 2007. Beach sands along the California coast are diffuse sources of fecal bacteria to coastal waters. *Environ. Sci. Technol.* 41, 4515–4521.
- Yamahara, K.M., Walters, S.P., Boehm, A.B., 2009. Growth of enterococci in unaltered, unseeded beach sands subjected to tidal wetting. *Appl. Environ. Microbiol.* 75 (6), 1517–1524.
- Yamahara, K.M., Sassoubre, L.M., Goodwin, K.D., Boehm, A.B., 2012. Occurrence and persistence of bacterial pathogens and indicator organisms in beach sand along the California coast. *Appl. Environ. Microbiol.* 78 (6), 1733–1745.
- Zhu, X., Wang, J.D., Solo-Gabriele, H.M., Fleming, L.E., 2011. A water quality modeling study of non-point sources at recreational marine beaches. *Water Res.* 45, 2985–2995.