

## **A public health evaluation of recreational water impairment**

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### **ABSTRACT**

Water quality objectives for body contact recreation (REC-1) in Newport Bay, CA are not being attained. To evaluate the health implications of this non-attainment, a comprehensive health-based investigation was designed and implemented. Bacterial indicator data indicate that exceedances of the water quality objectives are temporally sporadic, geographically limited and most commonly occur during the time of the year and/or in areas of the bay where the REC-1 use is low or non-existent. A disease transmission model produced simulated risk estimates for recreation in the Bay that were below levels considered tolerable by the US EPA (median estimate 0.9 illnesses per 1,000 recreation events). Control measures to reduce pathogen loading to Newport Bay are predicted to reduce risk by an additional 16% to 50%. The results of this study indicate that interpreting the public health implications of fecal indicator data in recreational water may require a more rigorous approach than is currently used.

**Key words** | microbial risk assessment, recreational water impairment, water quality evaluation

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### **INTRODUCTION**

Newport Bay (Orange County, California) has been identified by the Santa Ana Region California Regional Water Quality Control Board (RWQCB) as a water quality limited receiving water body. This designation arises because the fecal coliform water quality standards (200 MPN/100 ml log mean and 400 MPN/100 ml 90th percentile) were not being attained or were not expected to be attained with the implementation of technology-based controls. Thus, the bay was included on the 1998 California 303(d) List. Under these conditions, development of a total maximum daily load (TMDL) is required under the Clean Water Act. In California, the development of the TMDL implementation plan must be conducted within the context

of the Porter-Cologne Act (California Water Code), which requires that the potential costs associated with regulatory actions be considered in relation to the benefits projected for those actions.

The Newport Bay fecal coliform TMDL is a phased approach for understanding and controlling the microbiological water quality in the bay and focuses on generating information to ensure the reasonable protection of the bay's beneficial uses (CA RWQCB 1999). The fundamental issue during the first phase of the TMDL was whether occasional exceedances of the fecal coliform objectives for body contact recreation necessarily imply impairment to the beneficial use throughout the entire bay for all seasons. Several issues make answering this question difficult:

- (1) The primary risk to human health from recreational contact most likely comes from exposure to human viruses (Cabelli 1983; Levine and Stephenson 1990; Palmateer *et al.* 1991; Sobsey *et al.* 1995; Fankhauser *et al.* 1998; Mead *et al.* 1999; World Health Organization 1999), yet the water quality objectives are based on bacterial indicator organisms.
- (2) The quantitative link between the fecal coliform concentrations and adverse health effects is weak (Gerba *et al.* 1979; Cabelli 1983; Kay *et al.* 1994; Pruss 1998; McLaughlin and Rose 2000; Wade *et al.* 2003), and the association between bacterial indicators and virus concentrations is debatable (Fleisher 1991, 1993; Sobsey *et al.* 1995; Gantzer *et al.* 2000).
- (3) There are significant temporal and spatial variations in water quality in Southern California waters (Boehm *et al.* 2002) and in recreational use in Newport Bay.
- (4) The low number of people recreating in the bay at most locations make an epidemiological study impractical.
- (5) Limited data are available to distinguish between controllable and uncontrollable sources of coliform.

## METHODS

Mathematical models of disease transmission and water quality were developed and integrated to characterize the human risk of illness from waterborne pathogens in Newport Bay. Existing conditions in the bay were simulated and estimates of illness risk were generated. Each simulation represented an assumed 270-day time period during the dry season in the watershed. Alternative control options were identified as either potential structural modifications (i.e. treatment of stream flow and diversion of stormwater) or programmatic changes (i.e. public information campaigns). Planning level costs were estimated for the control options and the potential improvements to water quality and public health risk estimates were developed via simulation. The existing risk in the bay was then compared with levels of risk considered tolerable by the US Environmental Protection Agency (US EPA), and the potential effectiveness of the alternatives was characterized and presented with planning level costs.

The microbial risk assessment approach builds on previous work (EOA Inc. 1995; EOA Inc. & UC Berkeley 1995, 1999; Eisenberg *et al.* 1996, 1998; Soller *et al.* 2003, 2004), and is consistent with the 1996 EPA framework for microbial risk assessment (ILSI 1996) and the EPA's guidance for pathogen TMDLs (US EPA 2001). The risk characterization methodology employed in this study extends previous population based microbial risk assessment work by characterizing the population based risk from recreational activity in a metric that is comparable to the risk of illness to individual swimmers as specified by the EPA's water quality criteria (US EPA 1986, 2000). The risk assessment methodology makes explicit the mechanistic aspects of the infectious disease process from a population-based perspective and provides a defined structure for clinical, dose response and exposure data to be incorporated into the modelling process. A schematic diagram of the investigation showing how different types of data were used for this investigation is presented in Figure 1.

## Health protective assumptions

A combination of conservative and realistic assumptions was employed to ensure that the result of the assessment would be health protective. This approach is consistent with previous work (Haas *et al.* 1996) and is commonly applied in public health risk assessment. For this study the most important health protective assumptions include the following: (1) rotavirus was employed as the model virus for health effects and infectivity; (2) the model virus was as prevalent and persistent in the environment as male specific coliphage; (3) the boundary conditions in the water quality modelling were based on the maximum observed concentrations; and (4) all microbiological observations reported below the detectable limits were assumed to correspond to concentrations at the detection limit.

## Water quality monitoring

Within the Newport Bay watershed there are a number of potential sources of microbiological pollution. Those sources include tributary inflows, food wastes, sanitary

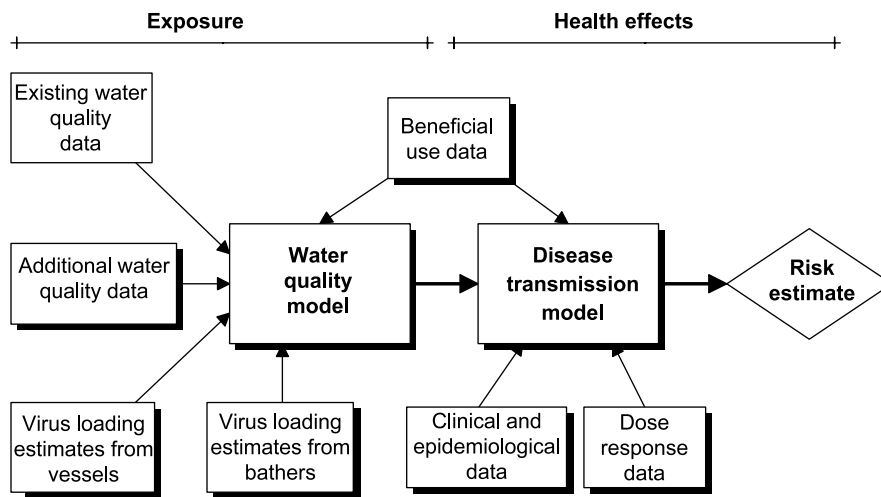


Figure 1 | Fundamental components of health risk characterization.

waste from vessels, wildlife fecal waste (CA RWQCB 1999), leakage of sewer lines (Metcalf and Eddy 1981), recreators (EOA Inc. et al. 1996; Yates et al. 1997), domestic animal wastes (Young and Thackston 1999), and illegal and/or illicit waste discharges from industrial, commercial and/or residential sources.

The County health agency (OCHCA) has monitored total and fecal coliform in Newport Bay for approximately the last 30 years. From the beginning of their monitoring effort to approximately 1998, total and fecal coliform data were collected weekly. Starting in 1999 and continuing to the present, total coliform, *Escherichia coli* and enterococcus data have been collected.

An additional ambient monitoring programme was developed for this investigation to augment the OCHCA bacteriological database, as follows:

- Monitoring for male specific coliphage was added to the existing OCHCA monitoring programme to establish a preliminary set of data, covering Newport Bay and tributaries, that could be used to characterize the relative levels of coliphage in the bay. Approximately 135 water samples were collected and analysed for male specific coliphage, 117 of which were found to be below the detection limit of 1 pfu/100 ml.
- Additional water samples were collected at three types of sites to establish conditions at the boundaries of the bay and in the bay to assist with calibration of the water

quality model for coliphage. A total of 260 samples were collected and analysed for *E. coli*, fecal coliform, total coliform, enterococcus, male specific coliphage, electrical conductivity, flow (major inflows and storm drains), salinity and temperature.

- The distribution of indicator organism density changes throughout the course of a tidal cycle was also investigated. Samples were collected at eight sites over a 2-day period throughout the course of a full tidal cycle and analysed for total coliform, fecal coliform, male specific coliphage and enterococcus.

#### Link between water quality data and risk to public health

Based on research conducted over the last 20 years, it is likely that the primary risk associated with recreational exposure to waterborne pathogens is from viral contaminants (Cabelli 1983; Levine and Stephenson 1990; Palmateer et al. 1991; Sobsey et al. 1995; Fankhauser et al. 1998; Mead et al. 1999; World Health Organization 1999). The recovery and detection of animal viruses in water, however, is technically difficult, time consuming and expensive. Furthermore, the epidemiologically important viruses are difficult to quantify in water (Sobsey et al. 1995). The local water quality objectives for REC-1 contact are delineated in terms of bacterial indicator organism concentrations. The utility of bacterial organisms as indicators for viral pathogens in ambient

waters, however, has been widely questioned (Gerba *et al.* 1979; Fleisher 1991; Fleisher *et al.* 1993; Kay *et al.* 1994; Sobsey *et al.* 1995; Gantzer *et al.* 2000; McLaughlin and Rose 2000).

Based on a review of the literature, male specific coliphages were selected as the indicator of enteric viruses for this investigation. The use of male specific coliphage as a viral indicator began to appear in the literature in the 1980s as researchers reported that these organisms may fulfil many of the essential requirements of a viral indicator (Havelaar 1987; Havelaar *et al.* 1984, 1986, 1993; IAWPRC Study Group 1991; Sobsey *et al.* 1995; Paul *et al.* 1997). A number of researchers report that the survival of male specific coliphages more closely resembles that of animal viruses compared with bacterial indicators (Baldini *et al.* 1978; Borrego *et al.* 1987; IAWPRC Study Group 1991; Armon and Kott 1996; Wommack *et al.* 1996; Paul *et al.* 1997; Sinton *et al.* 1999; Griffin *et al.* 1999; McLaughlin and Rose 2000). A number of factors make phages a reasonable surrogate for enteric viruses in this investigation. First, the structure, composition, size and mode of replication of phages resemble enteric viruses much closer than commonly used bacterial indicators of fecal pollution such as coliforms and enterococci. Second, phages survive longer in natural waters than enteric viruses, and fail to multiply in the environment (Grabow 2001).

From a review of the literature it also appears that male specific coliphages are present in equal or higher numbers than enteric viruses in ambient waters (Kott *et al.* 1974; Dutka *et al.* 1987; Stetler 1984; Gold *et al.* 1992; Morinigo *et al.* 1992; Havelaar *et al.* 1993; Jagals *et al.* 1995; AWWARF 2000). In these studies, the ratio of coliphage to viruses varied significantly, both between and within investigations. However, in all of these studies coliphages were found in numbers equal to or greater than enteric viruses. The ratio of coliphage to viruses reported ranged from ~1:1 to ~1,000,000:1, with levels most commonly reported between 100:1 and 1,000:1. To maintain the health protective nature of this investigation, a ratio of 1:1 (coliphage: enteric virus) was utilized.

### Beneficial use assessment

A beneficial use assessment programme was designed and carried out to describe the population that may be exposed

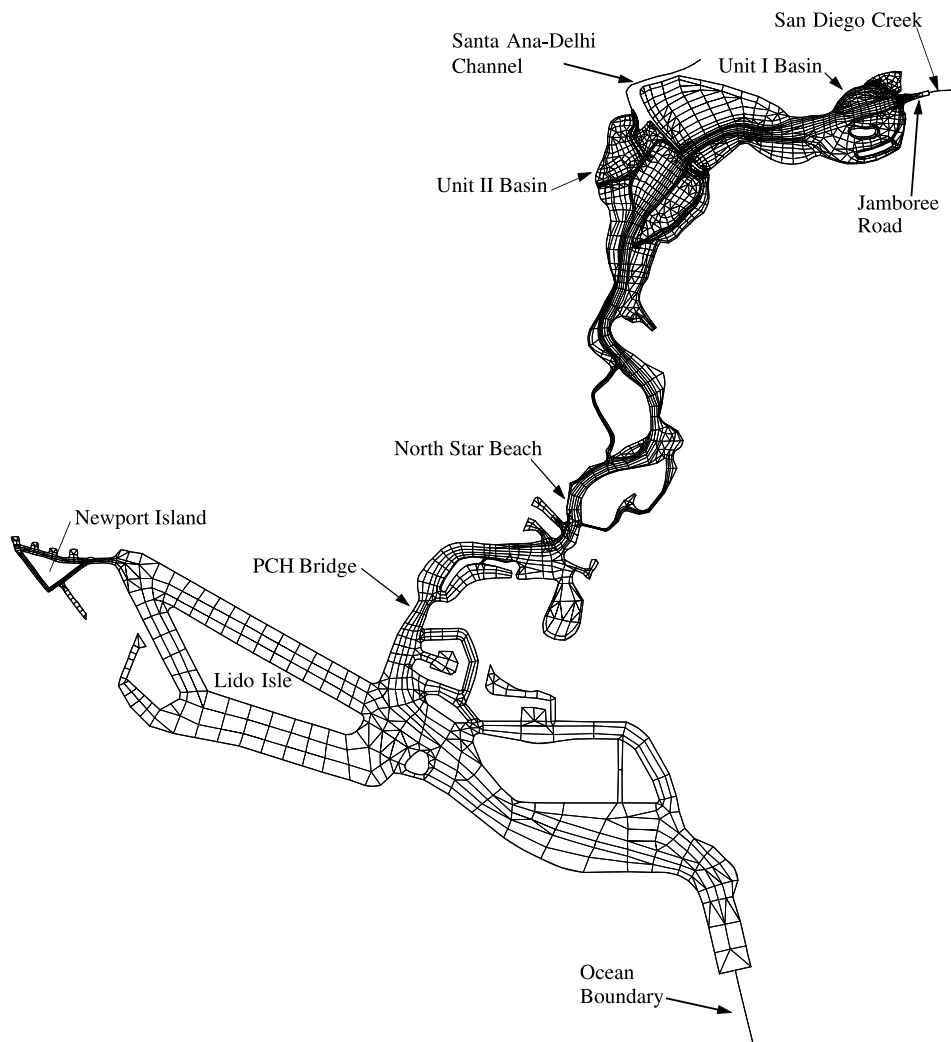
to pathogenic microorganisms during REC-1 activities in Newport Bay (EOA Inc. *et al.* 2001). The assessment programme included: monitoring (counting number of recreators) and surveying (asking recreators a series of questions). The sampling design included 36 days of monitoring and was based on a randomized sampling plan that accounted for various types of day (weekday, weekend, holiday). The design over-sampled during the summer (high use) period; however monitoring also occurred during the autumn, winter and spring seasons. Sites were selected based on those currently monitored by the OCHCA and on discussions with local stakeholders. The sites were categorized according level of use (prohibited, low, medium, high, extra high) by local stakeholders. The information collected during this beneficial use assessment was used in the health risk assessment in three ways: (1) to develop a profile of recreation use in Newport Bay by season, type of day and beach use level; (2) to estimate virus and coliform loading to the bay from bathers; and (3) estimate the size of the potentially exposed population (EOA Inc. *et al.* 2001).

### Water quality modelling

The purpose of the water quality modelling was to provide temporally and spatially varying pathogen concentration data to the disease transmission model. These temporally and spatially varying concentration data are used in the disease transmission model in conjunction with site-specific patterns of beneficial use in Newport Bay to define exposure from recreational contact. A second goal of the water quality modelling was to evaluate alternative control strategies that may affect viral loading. Specific details about the water quality model and the application of the model to Newport Bay are described elsewhere (DeGeorge *et al.* 2003). A very brief summary of the water quality modelling effort is summarized herein.

### Model configuration

The finite element model of Newport Bay extends from the tidal boundary at the entrance to the Lower Bay to San Diego Creek just upstream from Jamboree Bridge (Figure 2). Tributary inflows were specified at San Diego Creek, Santa Ana Delhi Channel, Big Canyon Wash, Back Bay Drain (calibration periods only), Fashion Island Drain and Arches



**Figure 2** | Finite element configuration for Newport Bay, 1997 bathymetric conditions.

Drain. The bay was represented using a two-dimensional depth averaged approximation, with short one-dimensional cross-sectionally averaged segments at the tidal boundary, San Diego Creek and Santa Ana Delhi Channel. Model mixing coefficients were calibrated during a salinity study in Upper Newport Bay.

### Fecal coliform model simulations

Coliphage simulations were used directly in the health risk assessment as the link between the water quality and health risk models. However, the extensive coliform monitoring database was used to develop and test the loading strategy

used in applying the water quality model of Newport Bay for pathogen transport. Dry season conditions were modelled because summer is the high recreation season and thus the period of highest human exposure.

### Boundary conditions

Boundary conditions for the simulations include inflow, temperature and fecal coliform concentration at San Diego Creek, Santa Ana Delhi and other major drains, and temperature and tidal elevation at the ocean boundary. Bather fecal coliform loads were applied at perimeter elements in the Newport Dunes swimming area.



Boat discharge loads were applied at the marina locations (Figure 3). Fecal coliform loads were also applied throughout the Lower Bay to represent the mass loading from the small, unmonitored storm drains. The locations and mass loads were estimated based on location and size of the storm drains, and were fine-tuned to calibrate the model. A distributed load was applied to represent other unknown non-point sources (DeGeorge *et al.* 2003).

### Die-off

Two coliform loss parameters were applied: die-off in darkness and light sensitive die-off. The die-off rate in darkness ranged from 0.02 to 0.038 h<sup>-1</sup>, with temperature dependence. The depth-averaged light sensitive die-off at peak sunlight ranged from approximately 0.16 to 2.4 h<sup>-1</sup>, varying with depth and temperature. Literature values indicate a die-off rate in darkness of 0–0.1 h<sup>-1</sup> (Hydroscience 1977) and a die-off rate in peak sunlight of 1.5–4.6 h<sup>-1</sup> (Fujioka *et al.* 1981).

Two coliphage loss parameters were applied: die-off in darkness and light sensitive die-off. The die-off rate in

darkness ranged from approximately 0.0024 to 0.0045 h<sup>-1</sup>, with temperature dependence. The depth-averaged light sensitive die-off at peak sunlight ranged from approximately 0.0028 to 0.041 h<sup>-1</sup>, varying with depth and temperature. Literature values indicate die-off rates ranging from 0.0028 to 0.096 h<sup>-1</sup> (Fujioka *et al.* 1980; Hurst and Gerba 1980; Raphael *et al.* 1984).

### Flow

Average dry season flows were used for each of the creeks and drains. Boundary flows are shown in Table 1.

### Fecal coliform and coliphage data

Boundary fecal coliform concentrations, based on the OCHCA data were set constant to the median of all available dry season fecal coliform data at each location. Data at the boundaries spanned ranges as wide as 4 to 5 orders of magnitude.

Boundary coliphage concentrations were set to the maximum of all available dry season coliphage data at each

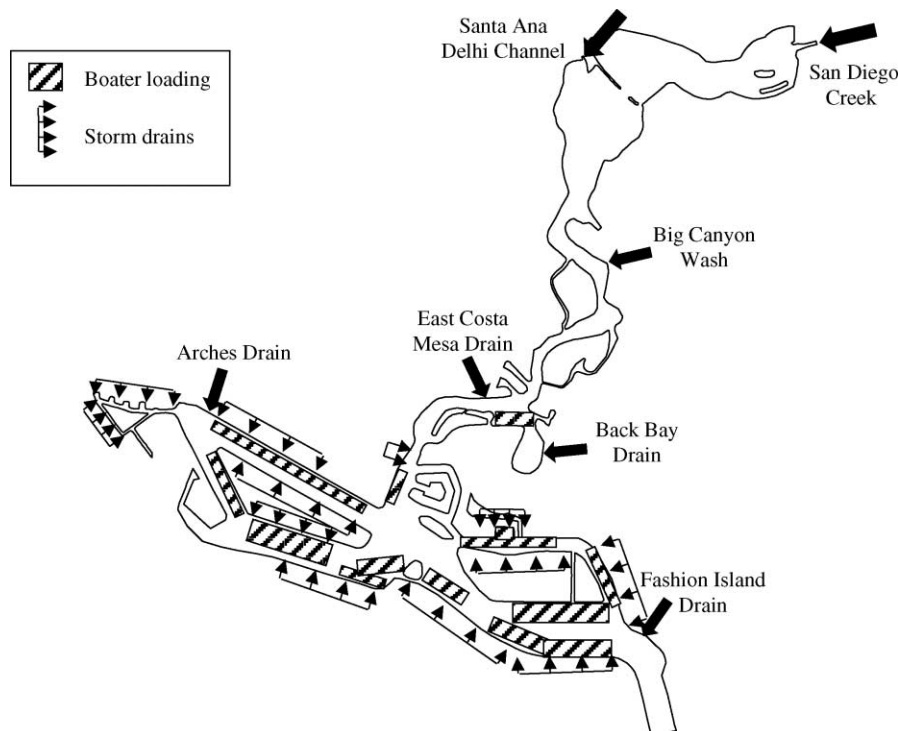


Figure 3 | Locations of creeks, major drains, small storm drains and boat loads.

**Table 1** | Boundary condition flows, fecal coliform and coliphage concentrations

| Source              | Flow ( $\text{m}^3 \text{s}^{-1}$ ) | Fecal coliform   | Coliphage   |
|---------------------|-------------------------------------|--|---|
| San Diego Creek     | 0.4                                 | 1,300 coliform $100 \text{ ml}^{-1}$                               | 3 coliphage $100 \text{ ml}^{-1}$                                 |
| Santa Ana Delhi     | 0.07                                | 1,300 coliform $100 \text{ ml}^{-1}$                               | 6 coliphage $100 \text{ ml}^{-1}$                                 |
| Big Canyon Wash     | 0.00997                             | 400 coliform $100 \text{ ml}^{-1}$                                 | 0.25 coliphage $100 \text{ ml}^{-1}$                              |
| Back Bay Drive      | 0.00177                             | 300 coliform $100 \text{ ml}^{-1}$                                 | 376 coliphage $100 \text{ ml}^{-1}$                               |
| Fashion Island      | 0.00157                             | 3,500 coliform $100 \text{ ml}^{-1}$                               | 1 coliphage $100 \text{ ml}^{-1}$                                 |
| Arches Drain        | 0.00656                             | 3,500 coliform $100 \text{ ml}^{-1}$                               | 1 coliphage $100 \text{ ml}^{-1}$                                 |
| Bather load (dunes) | –                                   | $1.6\text{E} + 7 - 1.1\text{E} + 8$ coliform $\text{h}^{-1}$       | $6.1\text{E} + 5 - 4.2\text{E} + 6$ coliphage $\text{h}^{-1}$     |
| Boat load           | –                                   | $9.88\text{E} + 10 - 3.95\text{E} + 11$ coliform $\text{day}^{-1}$ | $3.09\text{E} + 9 - 1.2\text{E} + 10$ coliphage $\text{day}^{-1}$ |
| Distributed load    | –                                   | 20 coliforms $\text{m}^{-2} \text{s}$                              |   |

location. Based on the ambient monitoring programme carried out specifically for this investigation, 1 to 12 dry season coliphage observations were available for each of the creeks and drains. The majority of these data were reported to be below detectable limits ( $< 1 \text{ pfu } 100 \text{ ml}^{-1}$ ). The maximum values were chosen for boundary concentrations to be health protective. If no data were above the detection limit for a particular inflow, the boundary was set at the detection limit.

### Calibration and simulation results

Large changes in coliform concentrations were computed between pre-dawn and peak sunlight hours; thus it is inappropriate to compare computed results over 24 hours with the OCHCA data that were collected during daylight hours. For a more reasonable comparison, median and average values were computed at each site for the model results between 7 a.m. and 3 p.m. only. Comparison of computed and observed values for the daytime hours only is reasonable for this study because, from site-specific data, it is known that the vast majority of body contact recreation occurs between the hours of 8 a.m. and 6 p.m. At most locations, the median computed values were in good agreement with the observed data.

Loading for the coliphage simulations was configured using a conservative interpretation of the limited

monitoring data, and then in-bay concentrations were compared with observed data. Two to five dry season coliphage observations were available at each station. Samples were taken between August 1999 and May 2000. Rigorous calibration of the coliphage model was not possible because of the limited dataset; however, the model did produce in-bay concentrations consistent with the observed data where the predicted coliphage concentrations at the sampling sites were less than 1 coliphage  $100 \text{ ml}^{-1}$ .

### Water quality model uncertainty analysis

To characterize the uncertainty in the water quality loading analysis, a numerical routine was developed to determine the contribution from each load to the total fecal coliform and coliphage concentrations at 25 specified locations throughout the bay. The results of this analysis were used to analyse the uncertainty of the model results based on the probability distributions of input data and uncertainty in die-off rates. Individual load simulations were performed for each constituent using: (1) die-off rates from the respective calibration/configuration simulations; (2) no light dependent die-off; and (3) no die-off. Throughout the bay the distributed load was found to be the largest contributor to fecal coliform concentrations using the calibrated die-off rates. With no die-off and the distributed

load eliminated, San Diego Creek became the largest contributor to fecal coliform concentrations. For the coliphage simulations, the vessel loading was the most important load, regardless of the die-off rate.

## Health risk modelling

### Model for disease transmission

A model for disease transmission (Figure 4) was developed previously to describe the epidemiological status of individuals within a population, and how that status varies over time (Eisenberg *et al.* 1996, 1998, 2004; EOA Inc. & UC Berkeley 1999; Soller *et al.* 2003). Four state variables in the disease transmission model (S, C, D and P) are used to track the number of people that are in each of the epidemiological states at any point in time (Figure 4). Rate parameters are used to determine the movement of the population from one state to another. The rate parameters include  $\beta$  (rate of acquiring infection),  $\sigma$  (rate of recovery from infectious states) and  $\gamma$  (rate of decline in immunity). Subscripts are used with rate parameters to distinguish the rates at which individuals move between states. For example,  $\beta_{SC}$  describes the rate at which individuals acquire infection and move from state S to state C. Similarly,  $\sigma_{DP}$  is the rate at which infectious and symptomatic individuals recover from a disease state (state D) and move to a state with temporary protection from infection (state P). Rate parameters are determined through literature review directly or are

functions of model parameters determined from the literature.

For this investigation, two routes of transmission are considered: primary transmission by background exposure and/or recreational contact in Newport Bay, and secondary transmission, which includes person-to-person transmission. Assuming that each of the transmission processes described above is independent (Hethcote 1976), the change in the fraction of the population in any state from one time period to the next is modelled as a first order differential equation. For example, the relative change in state S from one time period to the next due to primary infection is:

$$dS_1/dt = -\beta_{SC1} S - \beta_{SD1} S + \gamma P \quad (1)$$

(note that the numeric subscripts indicate that the route of transmission is (1) primary or (2) secondary).

Similarly, the relative change in state S from one time period to the next due to secondary (person to person) infections is directly related to the number of individuals who are in states S, C and D during that time period:

$$dS_2/dt = -(\beta_{SC2} + \beta_{SD2}) S(D + C) \quad (2)$$

The overall change in the number of susceptible individuals from one time period to the next is equal to  $dS_1/dt + dS_2/dt$ .

Additional model parameters used to compute rate parameters included: the probability of a symptomatic response  $P_{sym}$ ; the probability of infection for a specified

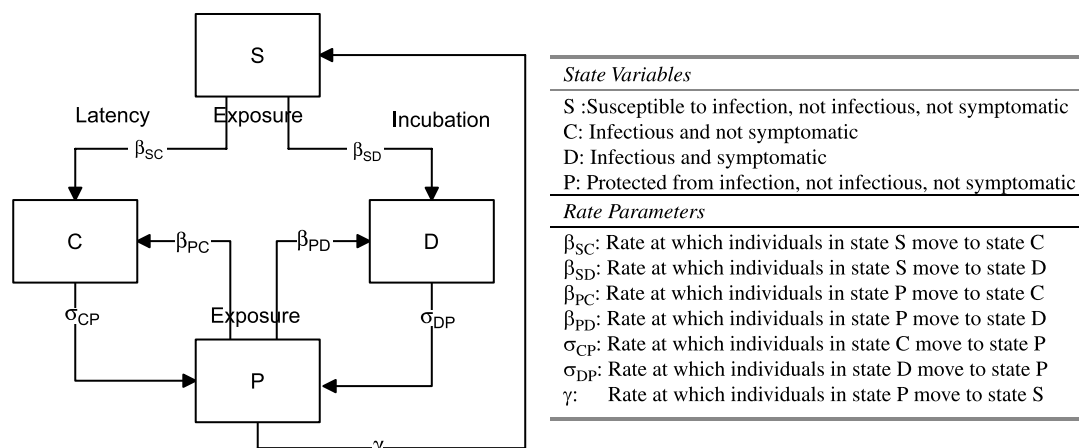


Figure 4 | Disease transmission model conceptual diagram.



dose,  $P_{\text{dose}}$ ; the beta Poisson dose response parameters,  $DR_{\alpha}$  and  $DR_{\beta}$ ; the probability of coming into contact with an infectious individual,  $P_{\text{contact}}$ ; the incubation period,  $\tau_i$ ; the latency period,  $\tau_l$ ; and an immunity factor used to account for partial protection due to previous exposures,  $\epsilon$ . Appropriate values for these parameters were determined through literature review (Table 2).

### Representative model pathogen

Rotavirus was selected for this study as the representative virus for enteric virus infections and disease, based on available dose-response and clinical data, and on its relative public health importance (Kapikian *et al.* 1980; Gurwith *et al.* 1981; Black *et al.* 1984; Champsaur *et al.* 1984; Mead *et al.* 1999). Rotavirus is the most infective virus for which published dose response studies are available (Haas *et al.* 1999), and also exhibits several critical epidemiological properties associated with other enteric viruses, such as the potential for person to person transmission of infection and partial protection from reinfection.

### Link between water quality, exposure and disease transmission model

The water quality model outputs hourly concentrations of the coliphage at each of the recreational sites. These water quality data are then used along with Newport Bay recreational use patterns to generate a profile of virus exposure for input to the disease transmission model. Specifically, the linkage between the water quality model and the disease transmission model is a dose (number) of the model pathogens to which recreators are exposed. That dose is a function of the expected number of people recreating at each site in Newport Bay (based on the use level of each site), the hour in the day, the day of the week, the duration of recreation events, the proportion of beach patrons who recreate in the water, the water quality at each site in the bay, and the volume of water ingested during recreational activities during each hourly time step. Exposure to the model pathogen from recreational activities was assumed to occur for 10 hours each day (representing 8 a.m. to 6 p.m.), whereas exposure to the pathogen from

secondary transmission was assumed to occur for 24 hours per day (EOA Inc. *et al.* 2001).

### Characterization of risk

The initial step in the risk characterization was to estimate a background level of pathogen exposure (assuming no exposure from recreational activities in Newport Bay) that would result in a mean prevalence level that was consistent with rotavirus prevalence levels derived from the incidence data reported in the literature (Rodríguez *et al.* 1987; Koopman *et al.* 1989; Mead *et al.* 1999). Average point prevalence incorporates both the number of cases and the duration of disease, resulting in a measure of disease intensity. Assuming steady state conditions and constant disease duration, average point prevalence can be compared with incidence (the number of new cases during a specified time period) by the following approximation (Kleinbaum *et al.* 1982):

Average daily prevalence  $\approx$  incidence \* duration of disease

A series of Monte Carlo simulations were run with randomly selected model parameter values (Table 2) and a given background pathogen exposure level (dose). Different dose values were used until the simulations produced mean prevalence values consistent with the literature. The result of this calibration was that a dose of 0.015 viruses resulted in simulations with a mean disease (infectious and symptomatic) prevalence of 0.3%. The identified background level of exposure was used in all subsequent simulations (see next paragraph) to ensure that the proportion of the population in each epidemiological state in the disease transmission model was a realistic estimation of the community's epidemiological status prior to additional exposure through recreational activity.

Exposure to enteric viruses in the Newport Bay watershed was then simulated for (1) background and (2) recreational exposures. For each simulation, a set of values was selected randomly for each of the model variables (Table 2). The background prevalence was checked to ensure it was consistent with prevalence levels derived from the literature. If the prevalence fell outside of the literature-based range, the parameter set was discarded and another

Table 2 | Model parameter summary

| Model parameters   | Parameter   | Units             | Range         | Source  |
|--|---|-------------------|---------------|---|
| Dose response parameters                                     | $DR\alpha$  | Unitless          | 0.125–0.5     | Ward <i>et al.</i> (1986), Regli <i>et al.</i> (1991)                       |
|  | $DR\beta$   | Unitless          | 0.21–0.84     | Ward <i>et al.</i> (1986), Regli <i>et al.</i> (1991)                       |
| Probability of symptomatic response                          | $P_{sym}$   | Unitless          | 0.1–0.45      | Tufvesson <i>et al.</i> (1977), Wenman <i>et al.</i> (1979), Haffjee (1995) |
| Previous exposure factor                                     | $\epsilon$  | Unitless          | 0.1–0.9       | Best professional judgement   |
| Reciprocal of incubation                                     | $1/\tau_I$  | day <sup>-1</sup> | 0.33–1.0      | Shepherd <i>et al.</i> (1975), Flewett & Woode (1978)                       |
| Reciprocal of latency  | $1/\tau_L$  | day <sup>-1</sup> | 0.145–0.533   | Gomez-Barreto <i>et al.</i> (1986), Ward <i>et al.</i> (1986)               |
| Rate diseased move to post infection state                   | $\sigma_{dp}$   | day <sup>-1</sup> | 0.09–0.5      | Gomez-Barreto <i>et al.</i> (1986), Gurwith <i>et al.</i> (1981)            |
| Rate carriers move to post infection state                   | $\sigma_{cp}$   | day <sup>-1</sup> | 0.05–0.125    | Ward <i>et al.</i> (1986), Ansari <i>et al.</i> (1991)                      |
| Rate of susceptible re-establishment                         | $\gamma$  | day <sup>-1</sup> | 0.0009–0.0027 | Bernstein <i>et al.</i> (1991), Ward & Bernstein (1994)                     |
| <b>Rate parameters dependent on model parameters</b>         |   |                   |               |   |
|  | $\beta_{SC} = (P_{dose} + P_{contact}) * (1 - P_{sym}) * \tau_L$            |                   |               |   |
|  | $\beta_{SD} = (P_{dose} + P_{contact}) * (P_{sym}) * \tau_I$                |                   |               |   |
|  | $\beta_{PC} = (P_{dose} + P_{contact}) * \epsilon * (1 - P_{sym}) * \tau_L$ |                   |               |   |
|  | $\beta_{SD} = (P_{dose} + P_{contact}) * \epsilon * (P_{sym}) * \tau_I$     |                   |               |   |
| <b>Intermediate variables used to compute rate variables</b> |   |                   |               |   |
|  | $P_{dose} = 1 - (1 + dose/DR\beta)^{-DR\alpha}$                             |                   |               |   |
|  | $P_{contact} = 1.38 * \sigma/N$   |                   |               |   |
|  | $N =$ Population size   |                   |               |   |

set was selected randomly. If background prevalence fell within the literature-based range, the simulation was rerun with the same model variable values for background exposure plus exposure through recreational activities. This process was repeated until 1,000 feasible simulations were completed. The model was implemented using Matlab 5.3 and Simulink 3.0 (Mathworks Inc. 1998).

Output from the disease transmission model is the number of people in each of the states as well as the average daily prevalence for the simulation, defined as the average proportion of the population that is symptomatic (in state D) during the whole simulation period. To estimate the attributable risk posed by body contact recreation in Newport Bay, the number of individuals entering the diseased state due to background exposure during each simulation was subtracted from those entering the diseased state from background plus recreational exposure. This difference accounts for the number of individuals entering the diseased state that is attributable to recreational exposure including subsequent secondary transmission of disease.

The water quality criteria for REC-1 exposure are derived directly or indirectly from the EPA's best estimate of the illness rates associated with the fecal coliform criteria (US EPA 1986), which are 8 illnesses per 1,000 swimmers for fresh water beaches and 19 illnesses per 1,000 swimmers at marine beaches. Therefore it was necessary to compare the output from the disease transmission model to the levels of risk considered tolerable by EPA (probability of disease per swim event). For this purpose the following approximation was employed:

$$\text{Risk of symptomatic illness per swim event} = \Delta P * \sigma_{dp} / N_s \quad (3)$$

where:

$\Delta P$  = the number of symptomatic illnesses attributable to REC-1 exposure

$\sigma_{dp}$  = the rate at which diseased individuals move to post infection state

$N_s$  = the average number of swimmers per day

## RESULTS

### Summary of water quality monitoring

Between 1990 and 2000, OCHCA analysed a total of approximately 26,625 analyses for total coliform, fecal coliform, *E. coli* and enterococcus for 35 Newport Bay sites. From a review of these data, the following general observations can be made (EOA Inc. et al. 2001):

- Most sites (27/35) had log-mean fecal coliform values for the wet and dry season that were less than the 200 MPN 100 ml<sup>-1</sup> water quality objective.
- Most sites (24/35) had wet season fecal coliform 90th percentile values above the 400 MPN 100 ml<sup>-1</sup> water quality objective.
- Approximately one half (17/35) of the sites had dry season fecal coliform 90th percentile values above the 400 MPN 100 ml<sup>-1</sup> water quality objective.
- For 1999 and 2000, 12 sites had fecal coliform 90th percentile values above 400 MPN 100 ml<sup>-1</sup>. Four of those sites were located in areas where recreational activities are prohibited by local ordinance, five sites are near areas that are used infrequently for recreational activity, and three are near areas which receive substantial recreational use.

### Beneficial use assessment results

The average number of individuals recreating at each of the Newport Bay beach sites between 11 a.m. and 3 p.m. from June 1999 to May 2000 is summarized in Table 3 based on the results of the beneficial use assessment. Based on the data presented in Table 3: (1) recreation intensity is substantially higher during the months of May to September than October to April; and (2) the vast majority of the recreational activities in the bay occurs at the high and extra high use level sites.

### Risk characterization: Disease transmission modelling

The number of individuals in each of the epidemiological states at the end of 1,000 simulations was found to be very similar for background and background plus body contact recreation exposure conditions (Figure 5). From the results of

**Table 3** | Average number of recreators during peak time of day

| Season                       | Type of day | Use level |        |      |                         |
|------------------------------|-------------|-----------|--------|------|-------------------------|
|                              |             | Low       | Medium | High | Extra high <sup>1</sup> |
| High (May through September) | Weekday     | 2.3       | 14.3   | 25.6 | 257                     |
|                              | Weekend     | 3.8       | 18.3   | 24.3 | 410                     |
|                              | Holiday     | 7.0       | 18.0   | 59.0 | 1300                    |
| Low (October through April)  | Weekday     | 1.0       | 3.0    | 3.7  | 8.3                     |
|                              | Weekend     | 0.3       | 2.4    | 4.1  | 6.3                     |
|                              | Holiday     | 0.8       | 2.2    | 0.7  | 18.2                    |
|                              | # Sites     | 10        | 10     | 5    | 1                       |

<sup>1</sup>For the purposes of the use assessment, the number of recreationists at the four Newport Dunes Beaches are reported together

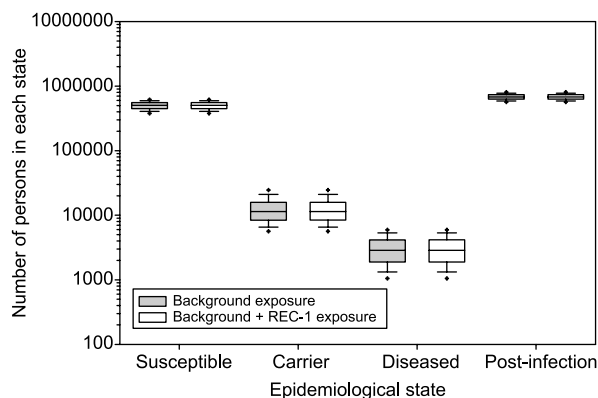
this analysis, it was found that approximately 99.9% of the population in the diseased state at any time is due to background exposure compared with approximately 0.1% due to recreational activities. In these simulations, all members of the population are subject to background exposure, whereas only those who choose to recreate in the bay are subject to the incremental recreational exposure.

Converting the population based attributable risk value to individual based equivalents indicates that the entire predicted distribution of population-based disease is below the EPA's acceptable illness rate (median risk per swim

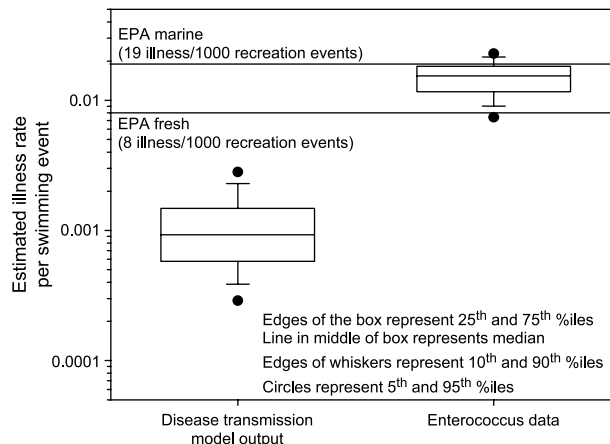
event) for both marine and fresh waters (Figure 6). The median illness rate per swim event predicted by the disease transmission model for recreation in Newport Bay during the dry season was ~0.9 illnesses per 1,000 recreation events.

**Risk characterization: Enterococcus data**

To provide an independent method of checking the disease transmission modelling results, a separate analytical approach was employed to evaluate enterococcus data collected in the most heavily used recreation site in Newport Bay. The approach involved applying a static



**Figure 5** | Disease transmission model output for background and background plus REC-1 exposure.



**Figure 6** | Comparison of disease transmission model output with enterococcus-based risk estimates.

based risk assessment approach (Soller *et al.* 2004) utilizing the EPA's enterococcus concentration: illness response function (US EPA 1986). The water quality results of 246 enterococcus samples were available from recreation sites for 1999 and 2000. The 48% samples that were below the detection limit of 10 MPN 100 ml<sup>-1</sup> were assumed to be present at the detection limit.

The enterococcus data were fit to a lognormal distribution using the method of maximum likelihood. Based on this distribution and the EPA's equation relating enterococci density in marine water to illness rate (US EPA 1986), Monte Carlo simulations were used to estimate the expected distribution of disease attributable to REC-1 activities.

Comparing the results of these simulations with the output from the disease transmission modelling simulations (Figure 6), the levels of disease predicted by both the enterococcus data and the disease transmission model are below the EPA's accepted marine levels. Furthermore, the levels of disease attributable to body contact recreation estimated by the disease transmission model are approximately an order of magnitude lower than those estimated using enterococcus data.

### Sensitivity and uncertainty analysis

Sensitivity and uncertainty analyses were carried out on the disease transmission modelling results to determine if the output was highly sensitive to any of the input parameters that were known to be highly uncertain and/or variable, and to characterize the sensitivity of the results in terms of implications to risk management. Three main source loading issues were investigated and are discussed below.

The concentration levels of viruses in Newport Bay were increased above those previously computed during the water quality modelling by factors of 10, 100 and 10,000. Simulations were run using the new concentrations along with the same parameter sets as described previously. The results of the analysis suggest that, even if the estimated concentrations of enteric viruses in Newport Bay were 10,000 times higher than were estimated in this investigation, the predicted median number of illnesses per recreation event (13/1,000) would be within the level of risk considered acceptable by the EPA for recreational activities in marine waters.

The loading contribution from vessels was increased by a factor of 100 over that specified previously, and that from recreators by a factor of 200. The vessel waste loading estimate is based on a number of factors including an estimated 10<sup>8</sup> infective virus particles per gram of feces shed by an infectious individual and an estimated 10% of the vessels discharging waste to the bay (EOA Inc. *et al.* 2001). The increase in loading from vessels investigated in this sensitivity analysis would be consistent with increasing the estimate of infective virus particles per gram of feces to 10<sup>10</sup> infective virus particles per gram (Flewett and Woode 1978) or including 100% of the boats in the bay discharging waste. The increase in loading from bathers corresponds to the 99th percentile of the distribution describing the estimated loading from bathers (EOA Inc. *et al.* 2001).

Based on the results of the analysis, the estimated risk to an individual per swimming event would still be within the levels considered acceptable by the EPA even if the levels of viruses contributed to Newport Bay via vessel waste loading were 100 times higher than estimated (7 illnesses per 1,000 recreation events), or if bather loading were 200 times higher than estimated (5 illnesses per 1,000 recreation events).

### Comparative assessment and planning level costs

Six alternative management options (Table 4) were developed and analysed via simulation. Each management option involves a reduction or elimination of a specific source of pathogens in the bay. The first four options focus on point source loadings to the Upper Bay such as storm drains and creeks. The last two options focus on bathers and boaters as sources of pathogen loading.

The comparative assessment indicated that under option 1 (which is currently in place) the median estimated illness rate is reduced from ~0.9 to 0.6 illnesses per 1,000 recreation events. Implementation of options 2, 3, 4 or 6 would result in an estimated incremental reduction of 0.1 illness per 1,000 recreation events or less. Under option 5, the median estimated illness rate decreases from approximately 0.6 per illnesses per 1,000 recreation events to approximately 0.3 per 1,000.

Planning level costs (Table 4) associated with options 1, 2, 3 and 4 involve estimated costs to treat or divert bay inflows while planning level costs associated with options 5

**Table 4** | Planning level costs for management options

| Option | Summary  | Estimated annual cost (\$) |
|--------|--|----------------------------|
| 1      | Back Bay storm drain diverted  | 15,000 <sup>1</sup>        |
| 2      | Option 1 plus water treatment for San Diego Creek                      | 2,359,000                  |
| 3      | Option 1 plus water treatment for Santa Ana Delhi                      | 1,405,000                  |
| 4      | Option 1 plus Upper Bay storm drains diverted to sanitary sewer        | 219,000 <sup>2</sup>       |
| 5      | Option 1 plus loading from vessels decreased to 10% of estimated level | 25,000–70,000 <sup>3</sup> |
| 6      | Option 1 plus loading from bathers decreased to 50% of estimated level | 25,000–50,000 <sup>3</sup> |

<sup>1</sup>Currently in place<sup>2</sup>Assumes drain diversion to sanitary sewer rather than treatment option<sup>3</sup>Assumes community outreach campaign

and 6 involve estimated costs for implementing new public outreach programmes (EOA Inc. *et al.* 2001). When the results of the simulations are considered along with planning level costs, the findings were as follows:

- (1) Eliminating potential dry weather viral loading from San Diego Creek, Santa Ana Delhi, or Big Canyon Wash to Upper Newport Bay (options 2, 3 and 4, respectively) would be more expensive than other options and would not substantially reduce the existing risk from recreation.
- (2) Reducing pathogen loading from vessels (option 5) may be more cost effective than the other alternatives considered.

Assessing the potential effectiveness of option 6 was difficult because the risks to recreators in the specific areas where the viral reductions were assumed to occur were not independently simulated.

## DISCUSSION

The primary advantage of a simulation-based methodology is the ability to evaluate the potential benefits associated with a

series of management options. It should, however, be understood that, during any recreation event, an individual may be exposed to a number of different pathogens derived from various different sources. Although it is not practical to estimate the cumulative risk from recreating via a simulation study such as this one, it is nevertheless feasible to frame an investigation in a manner such that practical risk management decisions may be considered (Soller *et al.* 2003). It is with this perspective that this investigation was designed and implemented.

## Limitations

Compiling and synthesizing a wide array of disparate data to assess the human health risk from exposure to recreational water is a complex task. To facilitate this assessment, a number of methodological assumptions were required. The limitations of the analyses presented herein generally stem from those assumptions. Two of the most important assumptions were: (1) the epidemiological status of the population could be approximated reasonably with the relatively simple structure of the disease transmission model; and (2) the model virus conservatively represented the pathogens of public health concern for recreation in Newport Bay.

With respect to the model structure, the health outcome modelled in this investigation was gastroenteritis. There are a number of other more serious disease outcomes that are also associated with exposure to enteric viruses. Those outcomes were not modelled in this investigation. Thus, characterizing the risk associated only with gastroenteritis probably underestimates the true cumulative risk to public health. Characterizing other endpoints more serious than gastroenteritis was beyond the scope of this investigation; however the likelihood for such health outcomes is important, and should be considered during the risk management process.

Microbial risk assessment is inherently agent specific and it is not practical to carry out separate assessments for all pathogens that may be present in an environment. Indicator organisms have been used for years because of the inability to take a universally defensible agent-by-agent approach. The same problem carries over to risk assessment. Therefore a variant of that theme was chosen



by modelling an outcome, viral gastroenteritis, and synthesizing a model organism that captured the salient features of viruses commonly known to be in water and known to cause gastroenteritis. The resulting analysis must be interpreted within this indicator context and extrapolation to other organisms, even viruses with dissimilar properties, must be done with caution.

In this investigation, risk estimates for recreational activities in Newport Bay were derived in two different ways: using the disease transmission modelling approach and using a static risk assessment approach with the US EPA's relation between enterococcus and recreation-related illness. The disease transmission modelling estimates were lower than the enterococcus-based results by approximately an order of magnitude. One explanation for the higher estimates of risk associated with enterococcus is that those risk estimates were driven by the detection limit. If a more rigorous approach were used to estimate the distribution of enterococcus concentrations, the risk estimate would probably be lower than that presented. A second possibility is that the effect of immunity in the disease transmission model may not be realistic for a mix of pathogens as may be found in recreational waters. If this is the case the disease transmission model risk estimates may underestimate the risk to recreators in Newport Bay. As a point of reference, it is noteworthy that the results of the disease transmission modelling are consistent with the World Health Organization (WHO) derived estimated risk values (less than 1 per 100) for recreational waters with enterococcus concentrations with 95th percentile values less than 40 per 100 ml (Kay *et al.* 2004). Given these three different methods of estimating risk from recreation in Newport Bay, the risk estimates derived herein seem reasonable. It is again urged that the results presented herein be interpreted with caution particularly given the range of pathogens that may be present in a recreational water body at any time.

### Risk management

Based on the findings of this investigation, it is reasonable to conclude that exceedances of the REC-1 fecal coliform water quality objectives most commonly occur during the time of the year when REC-1 use is low and/or in areas of the bay where the REC-1 use is low or prohibited by local ordinance.

Although the overall predicted population-level risk of illness from REC-1 activities is below that considered to be tolerable by US EPA for Newport Bay, the temporal and spatial variations in water quality undoubtedly can result in specific conditions with risk levels exceeding that tolerable level. Given these complexities and the difficulties associated with interpreting the public health implications of fecal indicator data, a more rigorous and comprehensive approach to evaluating the impairment of the REC-1 beneficial use in Newport Bay should be considered rather than relying solely on evaluating exceedances of bacterial based water quality standards. Such an approach would be consistent with (1) the basic principles of public health engineering regarding the use of sanitary surveys to identify and control potential sources to the maximum extent practicable; and (2) a health based monitoring approach for recreational waters outlined by the WHO (World Health Organization 1999).

If sufficient site-specific data or knowledge are available, it may also be reasonable to consider the issue of use impairment with a less comprehensive approach than that described herein. For example, data from a sanitary survey, relevant water quality data and/or specific knowledge about microbiological sources may allow a more qualitative approach to be employed. One example of how a qualitative approach may be used to facilitate risk management related to beneficial use impairment is presented in Table 5. In Table 5 water quality monitoring and exposure data are integrated to derive a relative level of public health concern associated with recreational activities. Management actions could be taken based on the prioritized level of public health concern.

Based on the results of this and previous investigations, it is clear that microbiological risk assessment investigations can provide valuable insight for risk management and regulatory decision-making. Given the technical results from an investigation such as that presented herein, risk managers should carefully evaluate the results along with all of the associated uncertainties within a broader regulatory context. Although the risk management process must account for many disparate considerations, to the extent feasible that process should take advantage of the best available scientific information including quantitative risk assessment methodologies to inform public policy. Quantitative risk assessment methods, as described in this study, offer an explicit description of available data, a clear

**Table 5** | Qualitative evaluation of relative public health concern for recreational waters

| Indicator concentration                   | Exposure |          |          |          |            |
|---|----------|----------|----------|----------|------------|
|   | None     | Low      | Medium   | High     | Extra high |
| Level of public health concern dry season |          |          |          |          |            |
| Low                                       | Very low | Very low | Very low | Low      | Medium     |
| Medium                                    | Very low | Low      | Low      | Medium   | High       |
| High                                      | Low      | Medium   | Medium   | High     | High       |
| Level of public health concern wet season |          |          |          |          |            |
| Low                                       | Very low | Very low | Very low | Very low | Low        |
| Medium                                    | Very low | Low      | Low      | Low      | Medium     |
| High                                      | Low      | Low      | Low      | Medium   | High       |

Note: The relative level of Public Health Concern shown in these tables is based on the combination of exposure level, indicator concentration, basic principles of sanitary engineering, and professional judgement. Concern at a given site changes seasonally, with the relative use that the site receives

statement of assumptions, a careful description of the analytical methods utilized to integrate the available data, and a clear statement about uncertainties relative to interpretation of the assessment results. Microbial risk assessment conducted within the context described in this paper offers valuable insight to risk managers.

## CONCLUSIONS

The principal findings from this investigation are that:

- (1) exceedances of the fecal coliform water quality objectives for the REC-1 use in Newport Bay are temporally sporadic, geographically limited and most commonly occur during the time of the year when REC-1 use is low and/or in areas of the bay where the REC-1 use is low or prohibited by local ordinance;
- (2) in areas where REC-1 activities occur, the estimated population-level risk of illness from REC-1 use in Newport Bay is generally below levels considered tolerable by the US EPA;
- (3) the reduction of the controllable sources of pollution that were investigated did not appreciably reduce the risk of illness from REC-1 activities.

The quantitative methodology employed and the qualitative approach introduced both allow risk managers to consider the impairment of the REC-1 beneficial use through a public health perspective, and could be used in other watersheds.

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