A SOURCE TO SEA METHODOLOGY FOR A COASTAL DISCHARGE

Benjamin R Knight[1], Dave Johnson[2], Derek Goring[3], Peter McComb[2], Paul Barter[1], Remy Zynfogel[2]
1. Corresponding author: ben.knight@cawthron.org.nz, Cawthron Institute, Nelson, NZ
2. MetOcean Solutions Ltd., …
3. Mulgor Consulting Ltd., …

ABSTRACT

A recent cross-disciplinary study has been undertaken by MetOcean Solutions Ltd. in partnership with the Cawthron Institute and Mulgor Consulting Ltd. to assess the performance of a coastal outfall in Hawke Bay. This paper describes the methods employed to estimate in-pipe, near and far field dynamics of suspended waste water material in the marine environment. The methods employed use a combination of existing and new "kiwi made" techniques which combine to provide a state-of-the-art assessment for the outfall, including multi-year simulations to quantify ENSO influences.

KEYWORDS

Hydrodynamic modelling, point source, bacteria, virus, pollution, outfall, wastewater, particle tracking

1 INTRODUCTION

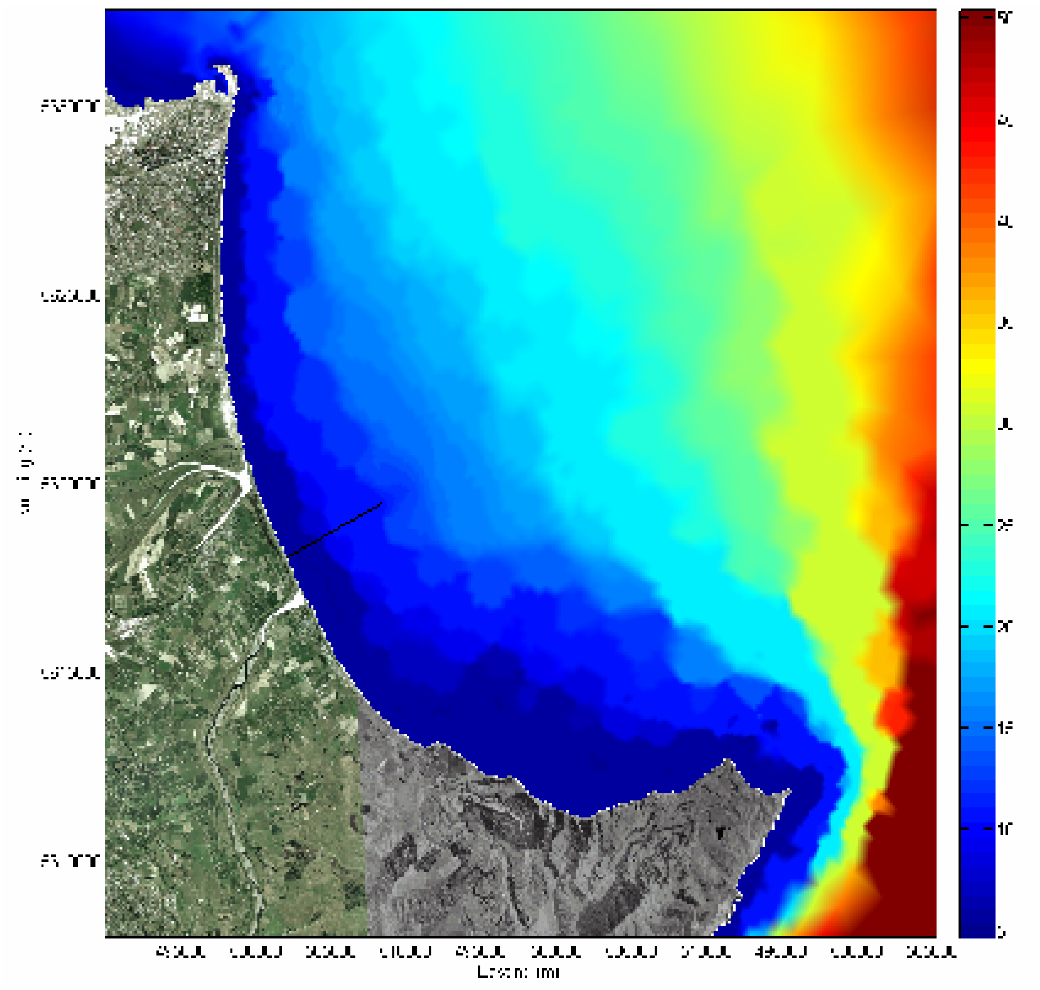
The production of organic wastes from urban and industrial sources continues to grow in line with the growth of urban centres and the increased industrial activities (e.g. food processing). Coastal oceanic discharges can offer a sustainable low energy solution to the disposal of these organic wastes. Provided these wastes are within the assimilation capacity of the receiving environment, marine ecosystems may dilute, consume and assimilate organic waste products without the need for external energy sources. However, the effects on the local users of the area surrounding an outfall also need to be considered, with people using coastal areas for food and recreation. Hence the social sustainability of an outfall is dependent on ensuring users of coastal water that interact with the outfall are not impacted by anything that may be considered pollutant.

In considering the impact of a discharge on a marine receiving environment, many properties of the outfall and the environment need to be considered. For the outfall, the magnitude and types of pollutants in the wastewater are important when considering its location and design. Typically the major pollutants of concern for domestic sewage are sediments, nutrients and pathogenic organisms (bacteria and viruses), although other unwanted compounds may also be present at low levels. Nutrients can enrich already eutrophied regions stimulating the growth of heterotrophic and autotrophic organisms which may lead to discolouration of the water, low dissolved oxygen concentrations and the stimulation of nuisance or harmful algal blooms. Pathogenic organisms are also of concern as they may affect the health of people using the water for recreation or food gathering.

As a consequence of these risks, guidelines produced by the Ministry for the Environment (MfE 2003) exist to provide guidance around what are acceptable levels in the water column. The design and positioning of the outfall should ideally allow the wastewater to be diluted in to a highly dispersive environment to ensure that discharged wastewaters meet these standards.

In 2009, Hasting District Council (HDC) contracted MetOcean Solutions Ltd. (MSL) with subcontracting assistance from the Cawthron Institute and Mulgor Consulting Ltd., to undertake an assessment of the Clive coastal wastewater outfall. The outfall is located in Hawke Bay and discharges up to 2.75 km offshore through 52 diffuser ports in an average depth of 13 meters (Fig. 1). The assessment was for a renewal of consent to discharge both treated domestic sewage and industrial wastewater (predominantly from meat and fruit processing plants) from the outfall. Suspended sediments and compliance with bacterial and viral guidelines were investigated in the study. For the purposes of this study, we focus on the methods employed to model bacteria and viruses around the outfall. Due to the pending consent process, this paper will focus on the modelling methods, the calibration process and our experience from this work rather than the results.

Figure 1: Location of the East Clive Wastewater Outfall (represented by the black line in the centre of the map), showing the depth relative to mean sea level in meters.



2 BACKGROUND

Our aim for our study was to provide HDC with estimated compliance to bacterial standards for their existing Clive waste water outfall under an estimated future flow scenario. This was undertaken to assess whether the existing outfall was able to meet coastal plan guidelines under a possible scenario that assumed future growth in discharge volume.

Central to our assessment was a review of activities in the region. This noted several recreational activities within the vicinity of the site, including: swimming, fishing, surfing and shellfish gathering. As all these activities involve contact with the water, the waters near the outfall are classified as contact recreation (class CR) under the Hawke Bay Regional Council regional coastal plan (HBRC 1999). The relevant guidelines for CR waters stated in the plan (HBRC 1999, Rule 11.4.2) for enterococci and faecal bacteria are as follows:

The water quality standards to be complied with, after reasonable mixing with the receiving water and disregarding any natural perturbations, by discharges to coastal waters classified as Class CR(HB) are:

(c) The median bacterial content of the waters over a bathing season, as determined by at least 25 samples taken from any site over any 4 month period, shall not exceed 35 enterococci per 100ml, and no sample shall contain more than 104 enterococci per 100ml;

(d) The median faecal coliform content of the water, as determined by at least 25 samples taken from any site over any 4 month period, shall not exceed 200 cfu per 100ml, and 95% of samples shall not exceed 2000 cfu per 100 ml;

Typically enterococci are used as a basis for saltwater sampling programs due to their higher survivability in salt water, indeed this ability forms the basis of the latest MfE marine recreational guidelines (MfE 2003). However, water quality sampling results show that a greater number of enterococci samples were below the detection limit (<1 cfu/100mL) compared to faecal coliform samples (25 vs. 23 samples) for the Clive outfall. A review of available research on light and temperature die-off relationships for enterococci was also not as extensive as for faecal coliforms, therefore the modelling of faecal coliform bacteria has been the focus of this study.

HBRC coastal plan rules (1999) are similar to the MfE (2003) guidelines which state that:

"These guidelines cannot be used to determine water quality criteria for wastewater discharges because there is the potential for the relationship between indicators and pathogens to be altered by the treatment process".

And go on to say that:

"while it is correct to infer that water exceeding guideline values poses an unacceptable health risk, the converse is not necessarily true".

These statements recognise that there is a possibility that guideline indicator bacteria concentrations may underestimate pathogenic organisms in treated wastewater. However, there is a notable absence of available guidelines specifically for treated wastewater, so drawing comparisons to the coastal plan rules is one of the most efficient ways of determining if receiving water poses an unacceptable health risk.

In New Zealand, no standards or guidelines exist with which to determine an appropriate concentration for viruses in the water column. However guidelines and directives are available in other countries for which acceptable concentrations are quoted. For the purposes of this study, we have chosen to use the European Economic Directive "Quality of Bathing Water" (76/160/EEC) which requires that cultivatable viruses be absent in 10 Litres of water (1 PFU/10 Litres) as referenced by Griffin et al. (2003). This standard is consistent with another study (Sattar, 1978) referenced by the guidelines for Canadian recreational water quality (Shaw et al. 1992). More stringent conditions (i.e. 1 PFU/ 40 Litres) are also referenced by the Canadian guidelines (e.g. Melnick, 1976), but these refer to infectious viruses which may vary greatly in concentration depending on the occurrence in the community. Hence the chosen limit of 1 PFU/10 Litres represents a credible attempt to provide an acceptable guideline value for comparison with the modelling study results.

Our aim for this project was to generate spatially explicit maps for 2 year-long periods encompassing two different phases of the southern oscillation index (SOI) under existing and elevated flow scenarios to determine the spatial extent of bacterial and viral concentrations in the waters surrounding the outfall. Concentrations were then compared to the standards stated above and will be used to inform a pending consent process. This paper attempts to synthesise our methods for presentation to other professionals in the hope that this work may stimulate discussion and collaboration with other projects requiring spatially explicit estimates of wastes discharged from coastal outfalls.

3 METHODS, CALIBRATION AND DISCUSSION

The modelling methods employed in this study follow a source to sea approach, whereby the study was separated into three main tasks:

1. In-pipe flow response of the outfall which was undertaken using the CORHYD modelling package.

2. Near-field modelling of the initial dilution and spread of the wastewater plume using the CORMIX modelling package.

3. Far-field modelling of suspended sediments and pathogens using the particle tracking software PartTracker in combination with transport estimates from the unstructured hydrodynamic model SELFE.

The results of each stage were passed on to successive stages through a loose coupling process, whereby the results of each stage of the model were first validated against collected data, and then passed to the next process following a source to sea approach. The goal of these efforts was to predict the compliance of the outfall relative to MFE guideline standards over two contrasting weather periods, an El Niño and a La Niña period corresponding to negative and positive phases of the SOI respectively.

3.1 IN-PIPE FLOW RESPONSE

The outfall pipeline was modelled using the CorHyd package. CorHyd is a steady flow pipeline model that has been configured for simulating outfall pipelines with diffusers. The purpose of this modelling exercise within the study scope was to provide the design velocity values for each of the diffuser ports. These velocities are critical to ensure accurate near field mixing and were used to validate the CorMix model estimates of port velocities (Section 3.2).

The pipeline consists of a 1.15 m diameter concrete feeder pipe 2.9 km in length and two 152 m long diffuser pipes of 1.067 m, then 0.915 m in diameter. In the diffuser pipes, there are a total of 51 ports of 0.155 m diameter discharging vertically and one 0.3 m diameter port at the end of the pipeline discharging horizontally.

3.1.1 CALIBRATION

Recorded pump flows and manifold pressures were obtained from HDC and were used for validating the model as shown in Figure 2. The smooth curves have been derived by low-pass filtering using orthogonal wavelet decomposition. The smooth curves correspond to timescales of 3 hours and longer.

The lower panel of Figure 2 shows the tide level (green curve); derived from tidal constituents from a sea level recorder that was deployed on the outer bank of the Napier Breakwater from Feb-2003 to Feb-2004. A cross correlation analysis indicated the tidal signal in the pressure record lags the ocean tide by 28 minutes. The outfall is 13 km south of the breakwater, but the tides are propagating up the coast, so we would expect the high tide over the outfall to occur a minute or two before high tide at the breakwater. Therefore, there is an approximate delay of 29 minutes between the occurrence of high tide at the outfall and the effect of high tide being felt at the pumps. Shifting the tide level at Napier Breakwater by 28 minutes and subtracting that from the smoothed pump pressure gives the net head, which is shown in Figure 3. This indicates there is a peak flow at the middle of each day and a corresponding peak pressure.

Figure 2: Typical period of pump flows (upper) and manifold pressures (lower) for the East Clive Outfall.

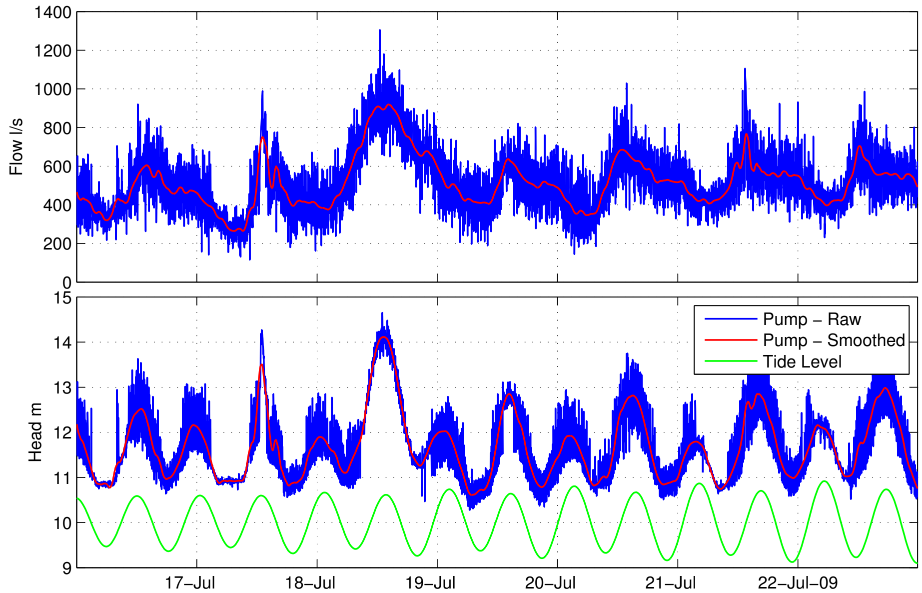
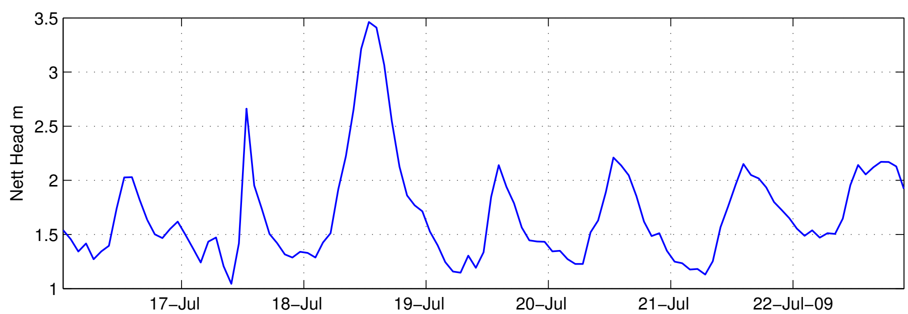
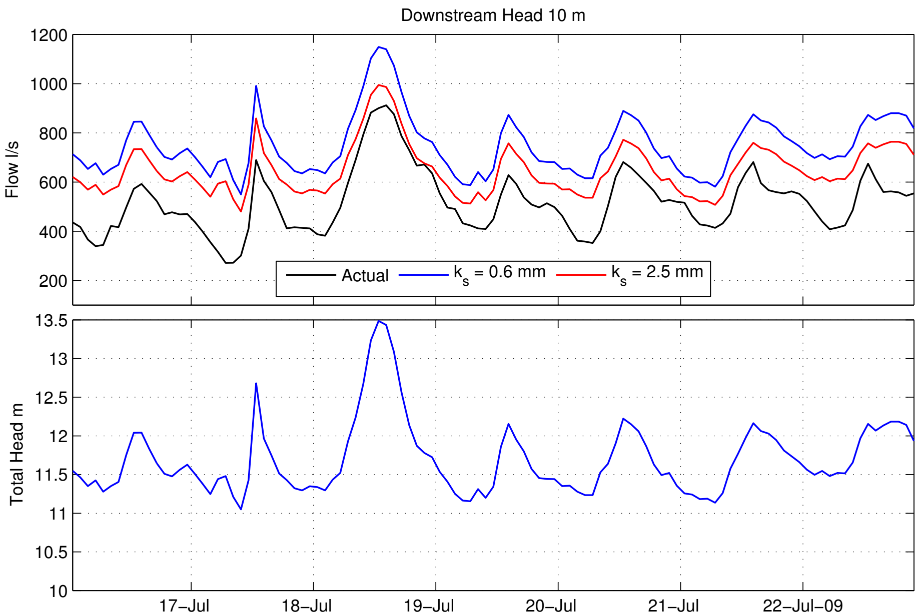


Figure 3: Net head after smoothing and removal of the tide.



Using the net head from Figure 3, the CorHyd model can be used to calculate the flow in the pipeline. The results are presented in Figure 4 for two options of the Nikuradse roughness for the feeder pipeline. These model results showed much higher flows than actually occur. In an effort to improve the estimates of flow from the model, the parameters were manipulated so as to reduce the calculated flows to the level of the measured flows. However, no amount of adjustment to the friction coefficient in the feeder pipe or the parameters related to the diffuser pipe (such as number of ports open) would reduce the flow sufficiently. Therefore, it was concluded that there must be an additional loss in head between the manifold where the pressures are measured and the pipeline. In order to calibrate the in-pipe hydraulics it was necessary to apply an additional head loss 0.2 m.

Figure 4: Actual flows compared to modelled flows for various additional head losses.



Using the calibrated CorHyd model for the East Clive Outfall, the jet velocity at each diffuser port was calculated for varying wastewater flows scenarios incorporating mean and peak wet and dry weather flows. The means of the calculated port velocities were used in subsequent near-field plume modelling using the CorMix model (Section 3.2).

3.2 NEAR-FIELD PLUME MODELLING

To characterise the plume behaviour in the near-field region, the CorMix-GI model[1] was used to predict the dispersion and dilution of the effluent plume. A full description of the CorMix model and its underlying assumptions are given provided by Jirka et al. (1996). Although the primary use of the model is for pre-construction design purposes under varying flow and diffuser configurations, it can also be used to verify field data and extrapolate near-field mixing (i.e. initial dilutions) under varying flow regimes for existing outfalls.

Given the complexity of hydrodynamic mixing and the range of variables that interact at an outfall diffuser, the model makes several assumptions and is not intended to be a substitute for *in situ* monitoring of the receiving environment. This monitoring was undertaken in a separate dye study which was used to validate the use of the model for use in the study.

A notable simplification of the CorMix model was that an average diffuser jet velocity was applied, when in reality the port velocities varied along length of the diffuser section, as shown in Section 3.1. There was also a 300 mm diameter horizontal port at the very end of the outfall, through which approximately 10% of the effluent volume exits which was not able to be included in the model. Examination of the effluent pumping pressures revealed an 8.6 minute oscillation, indicating the possibility of significant temporal variability of the effluent discharge regime, and the possibility of initial dilutions occurring within the diffuser section of the pipeline itself. The CorMix modelling was not able to resolve these dynamics and represents the idealised plume dynamics under spatially uniform and steady-state conditions (i.e. constant and equal flows from each diffuser port). Despite these simplifications comparison of CorMix results to a separate dye study showed the modelled dilutions were underestimated (i.e. the model predicted less dilution than observed). For the purposes of the

[1] Produced by the USA Environmental Protection Authority.

study this was considered to offer an additional level of conservatism and so further calibration was not undertaken on the model.

The CorMix model was used to predict the hydrodynamic mixing processes under four design flow conditions average and peak dry weather flows (ADWF, PDWF) and average and peak wet weather flows (AWWF and PWWF). Prior to running the model, CorMix was cross-checked with the outputs from the CorHyd model (Section 3.1). The average diffuser port jet velocity in the CorHyd model was compared against the jet velocity predictions in CorMix and found to be comparable (i.e. within 0.05 $m.s^{-1}$) for all four flow scenarios (Table 1).

Table 1: Summary results of the CorMix modelling of the diffuser, showing flow scenarios, applied jet velocities and near-field region (NFR) data from the model.

Outfall flow regime		**m^3/day (m^3/s)**	**CorHyd jet velocity (m/s)**	**CorMix jet velocity (m/s)**	**Ambient current (cm/s)**	**NFR distance (m)**	**Dilution at NFR X:1**	**Elapsed time (s)**
Average dry weather flow	**ADWF**	51,000 (0.5903)	0.65	0.63	0	-	-	-
					5	5.8	92.7	77
					10	80.1	578.8	801
					15	75.5	833.4	503
					20	74.1	1095.8	371
Peak dry weather flow (Feb -Apr)	**PDWF**	75,000 (0.8681)	0.95	0.92	0	-	-	-
					5	4.7	61.1	66
					10	84.0	403.6	840
					15	76.4	572.0	510
					20	74.6	749.1	373
Average wet weather flow	**AWWF**	70,000 (0.8102)	0.89	0.86	0	-	-	-
					5	4.8	64.5	67
					10	83.3	430.4	833
					15	76.2	611.7	508
					20	74.5	801.7	373
Peak wet weather flow	**PWWF**	120,000 (1.3889)	1.52	1.47	0	-	-	-
					5	4.1	43.2	60
					10	91.9	263.3	919
					15	77.8	362.7	519
					20	75.5	472.2	378

For each of the design effluent flows, the model was run for four sets of ambient currents: 5, 10, 15 and 20 cm s^{-1}, representing the typical range of depth-averaged current flows at the diffuser locations. Based on the measured depth-averaged data (Section 3), the 10 cms^{-1} level is approximately the median depth-averaged current speed, while 15 and 20 $cm.s^{-1}$ is the 86th and 97th percentile non-exceedence levels respectively. Conditions with depth-averaged current speeds of 5 $cm.s^{-1}$ or less occurred for some 22% of the time.

Results of the CorMix modelling under the range of ambient flow conditions for the design effluent flow states are presented in Table 1. These results include: i) the near field region (NFR) distance, which is the distance from the diffuser to the surface plume centroid, ii) the estimated dilution of the surface plume at the centroid, and iii) the travel time from diffuser port to surface plume centroid.

Notably, a comparison of the modelled dilutions with the measured values under the same flow regime (i.e. 0.2 $m.s^{-1}$ ambient currents and approx 50,000 m3/day effluent flow) showed good agreement, but was complicated by the high ambient currents on the day. The predicted CorMix dilutions under these higher than average currents (*i.e.* 20 cm/sec) was in excess of 1000:1 which was the limit of resolution for the dye study.

Nevertheless, the data collected showed that measured dilutions below 1000:1 were not common which supported the CorMix calcuations.

3.2.1 COUPLING OF NEAR TO FAR FIELD FLOWS

The particle tracking model (described in Section 3.3.2) was directly coupled to the CorMix near field diffusion results by generating a plume prediction file. This was calculated based horizontal and vertical dimensions of the plume at every output time step from CorMix, including the average concentrations at transects across the plume. The plume was fitted to Gaussian concentration profile, so that 99% of the total load was allocated to within the plume boundaries predicted by the CorMix model (e.g. Fig. 5. A lookup of plume templates was produced for the range of wastewater (ADWF, PDWF, AWWF, PWWF) and ambient flow regimes (5, 10, 15 and 20 cm s^{-1}).

At each 15-minute time step in the particle tracking model run, the wastewater flow rate and ambient oceanic flow at the diffusers are used to select and appropriate plume from the lookup templates. The current speed was obtained directly from the local 3D hydrodynamic model described in Section 3.3.1.

The selected plume was orientated to align with the current direction (relative to the diffuser) for the subsequent release of particles. It was not possible to model a pure slack water scenario with the CorMix model, and given flows less than 5cm/s occurred up to 22% of the time it was necessary to model a realistic slack water "boil". This was achieved using the 5 cm.s^{-1} plume scenarios for each flow regime. This was generated by offsetting the plume back by the distance moved due to the current (i.e. 4.5 m over 15 minutes for the 5 cm.s^{-1} current). The plume was then mirrored about the outfall pipe (the x-axis) to simulate the spread of the plume in both directions (Fig. 6) . With the exception of the slack water scenario (for currents less than 2.5cm/s), the selected plume was orientated to align with the ambient current direction. The particles were released using a weighted randomized position method, so that the relative concentration of particles matches the concentration given in the plume data predicted from CorMix.

Figure 5: An example of a 15 minute snapshot of the CorMix plume output used to determine particle release in a 20 cm.s-1 current (in the positive x direction) and during peak dry weather flow conditions.

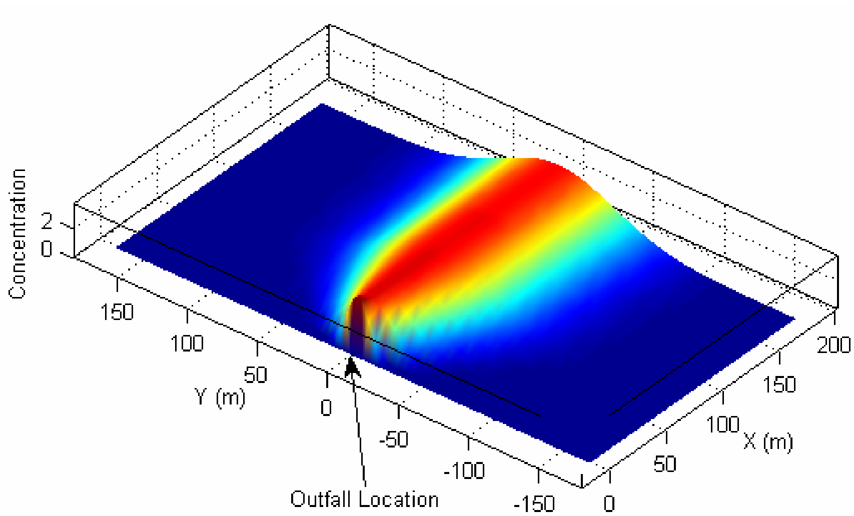
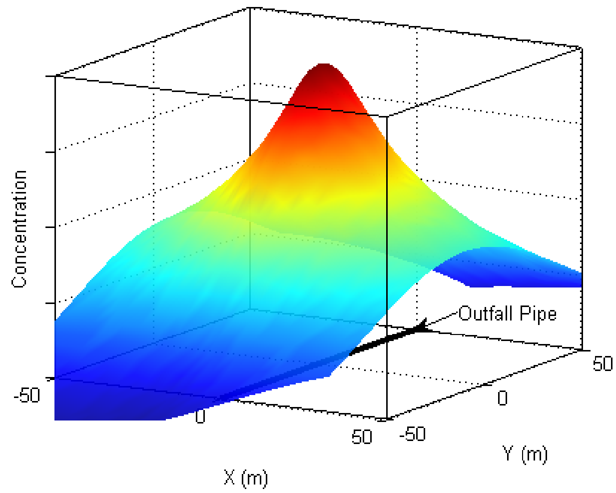


Figure 6: Concentration of a slack water "boil" generated for a PWWF scenario. The boil spreads in both directions away from the line of the outfall pipe.



In order to convert particle concentrations to microbial concentrations, a bacterial/viral load was associated with each released particle. This load was calculated so that at each 15-minute period when particles were released in the model, the total load matched the load over the same period from the outfall. The total load leaving the outfall over the particle tracking release period (15 minutes) was calculated by multiplying the effluent concentrations by the appropriate flow scenario scaled to the release period. These loads are evenly distributed to particles in the model, so that the modelled particle release concentrations match the predictions of the near-field modelled plumes.

The study used measured bacterial concentrations, however virus concentrations were not available for the outfall, so an average literature value of 50 PFU/l (where PFU refers to "plaque forming units") was used (Metcalf et al. 1995). We note that variations in virus concentrations are possible due to the occurrence of community epidemics.

3.3 FAR-FIELD PLUME MODELLING

The transport and dispersion of microbial and fine particulate matter from the wastewater effluent has been simulated with far-field particle tracking model coupled to near-field plumes from the CORMIX model presented in Section 3.2.

Central to the modelling of the far-field dispersion and transport of microbial and fine particulate matter, was the underlying transport dynamics of the region. As it is not feasible to measure transport at all depths and times over the entire Hawke Bay region, we utilised hydrodynamic models calibrated to collected current data from a 2 month period in 2009 (7 April to 16 June 2009) and passive drifter data to estimate current patterns in the region for two periods, an El Niño and a La Niña period. This section provides background on the methods we employed to construct the hydrodynamic models and a description of the particle tracking model methodology used to estimate far-field transport in the region.

After creating the transport data, Lagrangian particle tracking utilises this information to predict particle paths of suspended material from the outfall. Two types of material are modelled, neutrally-buoyant environmentally sensitive biological particles (bacteria and virus), and sinking sediment particles. For the biological particles, environmentally driven die-off relationships are applied to the particles so that their mortality dynamics were appropriately modelled.

3.3.1 HYDRODYNAMIC MODELLING

The study has employed two different hydrodynamic models to reproduce the complex flow regime at the outfall. At a coarse New Zealand scale, a 2D (depth-averaged) model was used to define the tidal and wind-driven currents at relatively coarse resolution (approximately 5 km). The data from this model was used to provide the boundary condition to a higher resolution 3D unstructured model that includes the wider Hawke Bay region including the outfall location. A description of the numerical models and details of their implementation is provided in the following sections, along with validation against the 2009 current profiles measured near the outfall diffuser.

Regional scale 2D hydrodynamics

The MSL implementation of Princeton Ocean Model (POM) was used to hindcast the regional tidal and wind-driven currents at the boundary of the Hawke Bay local-scale model domain. POM is a primitive equation ocean model that numerically solves for oceanic current motions. This model has been used for numerous scientific applications studying oceanic and shelf circulation; details of POM implementations are described by Mellor (2004). For the hindcast simulations, MSL-POM was used in a vertically-integrated two-dimensional mode, and implemented over a New Zealand-scale domain applying at a resolution of approximately 4.5 km by 5.4 km.

Surface forcing, both the 10 m winds and atmospheric pressure were input into the hydrodynamic model. The surface pressure is from the NCEP global reanalysis data product and surface winds are from the MSL Blended Sea Winds data product. Wind velocity components and atmospheric pressure were interpolated linearly in both space and time onto the model grid. The TPXO7.1 global inverse tidal solution (Egbert and Erofeeva, 2002) was used to prescribe the tidal elevation and current velocity at the boundaries of the New Zealand grid.

The MSL-POM model has been validated at various coastal and open-ocean locations around New Zealand (i.e. Otago, Taranaki, Western Cook Strait and Bay of Plenty).

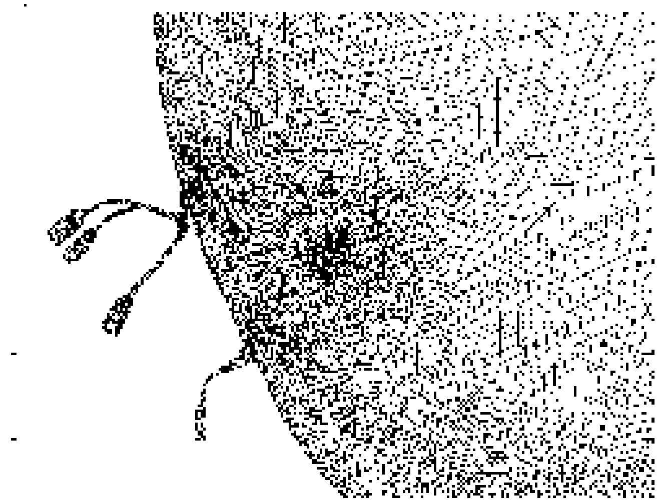
Local scale 3D hydrodynamics

A local-scale hydrodynamic model (SELFE) was also applied in order to determine the fine-scale transport information in the region of the outfall. SELFE is a prognostic finite-element unstructured-grid model designed to simulate 3D baroclinic circulation. The barotropic mode equations employ a semi-implicit finite-element Eulerian-Lagrangian algorithm to solve the shallow-water equations, forced by relevant physical processes (atmospheric, oceanic and fluvial forcing). SELFE uses either pure terrain-following sigma, or S-layer coordinates in the vertical, or a hybrid system using both S and Z-layers as required and uses sophisticated vertical turbulent closure models. A detailed description of the SELFE model formulation, governing equations and numerics can be found in Zhang and Baptista (2008) .

Collected current data from a location near the outfall showed periods of strong vertical current shear, implying a high degree of stratification. For this reason SELFE was run in a fully 3D baroclinic mode, including all thermohaline processes except precipitation and evaporation.

A triangluar finite element mesh grid was constructed for the local-scale model to cover all of Hawke Bay. The mesh was refined in shallow regions, around complex coastline features and in the vicinity of the outfall (Fig. 7). The mesh size ranges from 2.7 km^2 to 1100 m^2 near the coastline. Bathymetric data was obtained from a blend of survey data and digitised nautical charts and interpolated to the mesh nodes. The mesh was further refined to 800 m2 near the outfall diffuser. The vertical discretisation used 10 sigma levels with 30, 0.7 and 10 as the hc, θb and θf constants used in Song and Haidvogel's (1994) S-coordinate system. This ensured that the surface layers had a constant thickness regardless of depth unlike a standard sigma parameterised grid whereby the depth of surface cells is a function of the water column depth.

Figure 7: *Close-up view of the finite element grid used in SELFE model around the East Clive outfall.*



The SELFE grid was nested within MSL-POM regional (New Zealand) model, with surface elevation and current velocity prescribed at all open boundaries. The combined tidal and wind driven depth-averaged flow was specified. The depth variation of velocity at the boundary was approximated with a logarithmic profile using roughness length consistent with the model bottom friction parameter. The model velocity fields are 'cold' started from rest with a ramping period of 2 days, during which the forcing and boundary conditions are gradually applied.

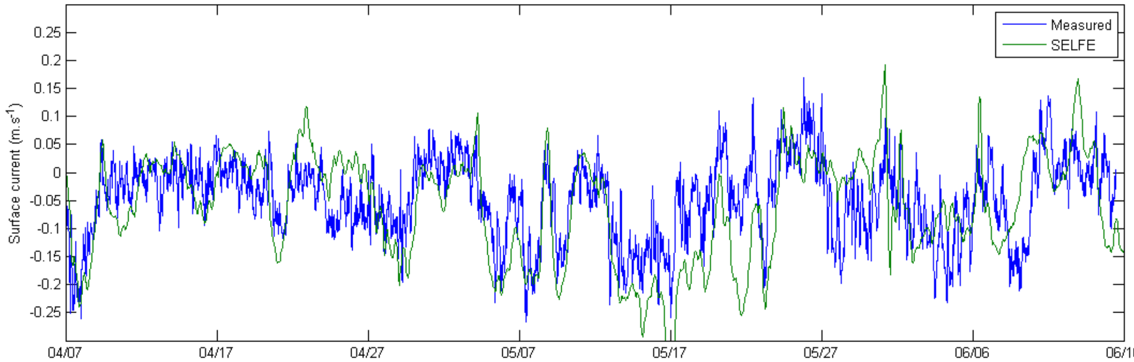
Three-dimensional temperature and salinity boundary conditions were interpolated from the Bluelink ReANalysis (BRAN) 2.1 data (Schiller et al., 2008). BRAN is a multi-year integration of an ocean model that assimilates observations, including along-track sea-level anomalies (from altimeters), in situ temperature and salinity (from ARGO drifters) and sea surface temperature (from satellite observations). Initial conditions for temperature and salinity were interpolated from the BRAN data.

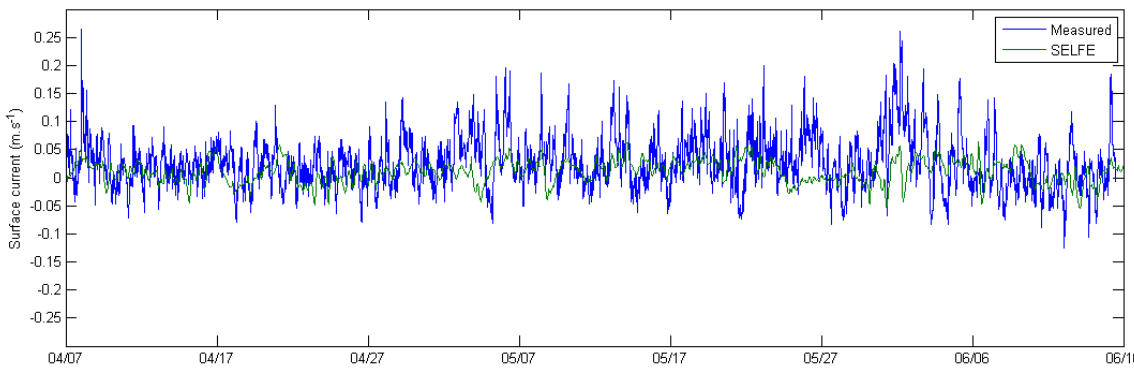
Wind and meteorological forcing of local model

The surface shear stress due to wind forcing required a spatially and temporally varying wind field. It was found that the flow within the model has a very high degree of sensitivity to the wind field. Furthermore there is a large degree of sheltering, in particular between Napier and Cape Kidnappers; which was observed shore-based winds were compared to offshore locations. It was therefore necessary to utilise a relatively sophisticated diagnostic wind model, CALMET (Scire et al., 2000), to construct an appropriate wind field across the model domain.

CALMET was used to interpolate and adjust data from available land stations around Hawke Bay with the offshore blended Seawinds data (Zhang, 2006). In addition CALMET makes some adjustment for topographic sheltering and steering that occur around the complex topography of the Hawke Bay and for the frictional effect of the land surface. Accurately reproducing the wind-field in the region was a major technical challenge of the project, but careful construction of the windfield ensured the hydrodynamic model results were able to accurately reproduce recorded currents at the site (Fig.8).

Figure 8: Timeseries comparison of modelled and measured near surface velocity for validation period. Alongshore currents are presented in the upper plot, cross shore currents are presented in the lower plot. Up coast and offshore are the positive vectors.





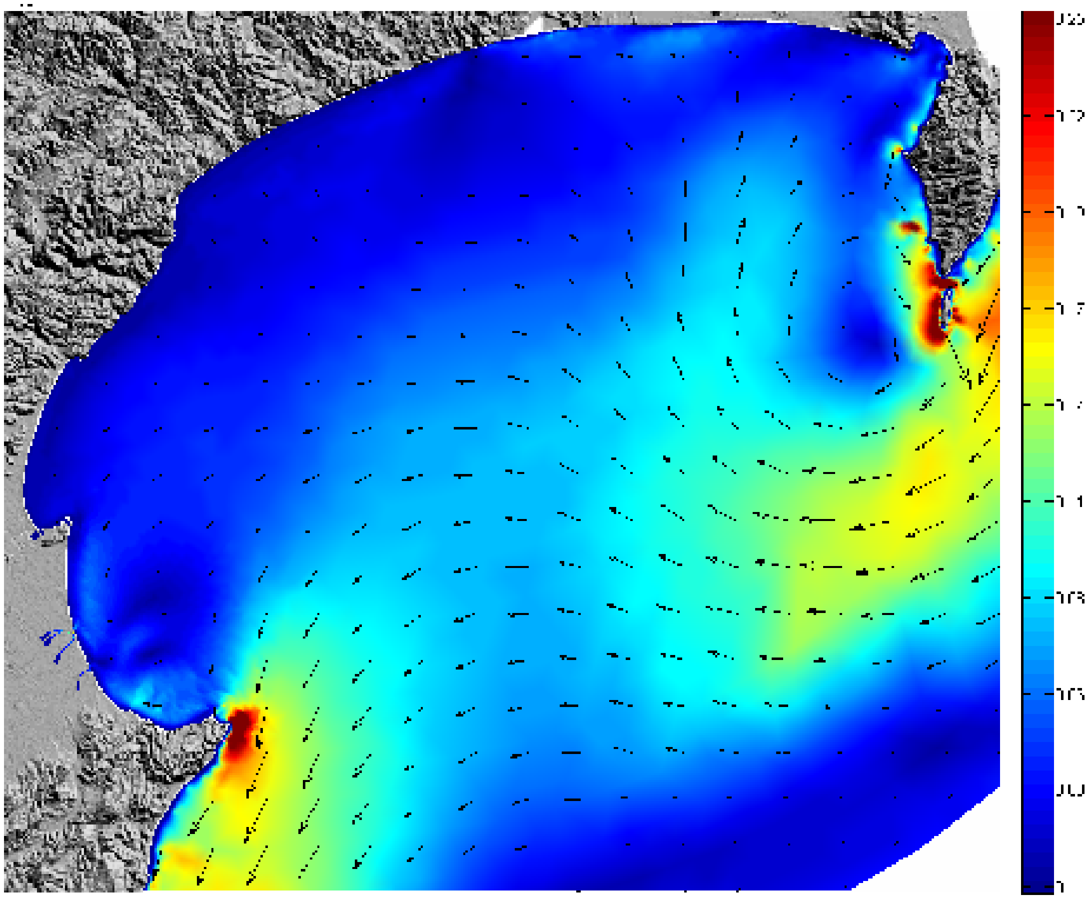
The surface atmospheric pressure, near-surface temperature and humidity, and down welling short and long wave radiation were specified at the model surface. All these data were interpolated in time and space from NCEP/NCAR Reanalysis 2 data (Kanamitsu et al., 2002). The NCEP R2 is a global analysis/forecast system used to perform data assimilation using historical data.

River inflows were included in the model. The lower sections of the rivers were included with the hydrodynamics in those section fully simulated. The temperature of the river water was specified on a climatic monthly basis from averages of the available measured temperature records. Flow rate data and temperature measurements were provided by HBRC. Full timeseries were not available for all rivers, and these rates were inferred from available coincident relative measurements.

Ultimately, all these drivers combine to give a realistic spatially explicit model of the physical properties and currents of the coastal water in Hawke Bay. An example of a depth averaged current field is displayed in figure 9 which illustrates the spatial variation in currents in Hawke Bay at a point in time.

Comment [brk1]:

Figure 9: An example of modelled depth-averaged currents (m/s) over Hawkes Bay



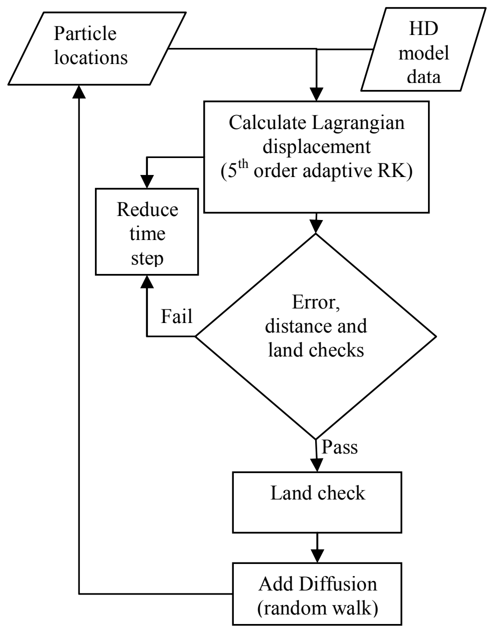
3.3.2 PARTICLE TRACKING AND BIOPHYSICAL DYNAMICS

Lagrangian particle tracking methods were used to predict the transport and dispersion of suspended material (bacteria, viruses and sediment) from the outfall. It was used in preference to other methods (e.g. Eulerian methods) due to advantages associated with the computational efficiency of rerunning multiple particle releases, modelling environmental interactions and reducing numerical mixing effects. Particle tracking has been undertaken utilising a software tool, PartTracker, jointly developed by MetOcean Solutions Ltd and the Cawthron Institute. The technique has been validated and is described in the paper by Knight et al. (2009).

PartTracker calculates the Lagrangian paths of released particles over a given time step by numerical integration within a time-varying velocity field. The particles can represent a finite quantity (or “load”) of material, for example bacteria or sediment. PartTracker is novel in that it uses a 4th/5th order adaptive Runge-Kutta scheme to ensure the accuracy of the integration calculation. The integration calculation uses velocity estimates that are linearly interpolated in time and space from discrete time “snapshots” of the flow fields from the hydrodynamic model output (Section 3.3.1). The numerical scheme then calculates an error estimate using the difference between the 5th order and embedded 4th order Runge-Kutta solutions (see Press et al. 1992). If the calculated error estimate for a given particle is greater than a predefined value (1 mm is used in this study) then the time step is further reduced until the required accuracy is achieved.

If a particle passes the error check, it is also tested to ensure that the distance moved is not greater than a predefined value (50 m is used in this study). This ensures that particle movements do not jump over velocity data in areas of high flow. As with the error check, if this distance is exceeded, then the time step is lowered accordingly. This ensures that the particles accurately reproduce streamlines within the current fields. This process is illustrated in the following schematic of the model (Fig. 10).

Figure 10: Schematic showing the processes undertaken by PartTracker for a given time period. Note that after each calculation the numerical accuracy of the displacement estimate is checked. A proportion of the particles may fail, requiring shorter time steps to meet the desired accuracy.



After the particle path has been determined, a further check is also made to see if particles have become beached by interacting with land. For the purposes of this study, the model is parameterised so that particles hitting the shoreline remain there until the currents allow the particles to move away from the shore. This simulates the association of pathogens with organic material, for which this is feasible behaviour. For sinking particles (i.e. the fine particulate matter in the effluent), a 'sticky' seabed is parameterised, whereby particles that reach the seabed are not re-suspended.

After calculation of the Lagrangian displacement is completed for all particles, an additional random displacement is added to simulate diffusion processes. The random component is determined from a classical diffusion equation (García-Martínez & Flores-Tovar, 1999; Lonin, 1999) as:

$$\begin{aligned} T_x &= R(6D_x\Delta t)^{1/2} \\ T_y &= R(6D_y\Delta t)^{1/2} \\ T_z &= R(6D_z\Delta t)^{1/2} \end{aligned} \qquad (1)$$

where Tx and Ty are the random turbulent components. R is a uniformly distributed random number and Δt is the model time step. D(x,y,z) is the diffusivity in the horizontal an vertical directions respectively, the value of which is estimated from the eddy diffusivity calculated by the hydrodynamic model.

Application of biological properties to particles

In order to ensure that realistic estimates of indicator bacterial concentrations can be derived, it was also necessary to model the die-off of bacterial loads associated with modelled particles. The die-off rates for the bacterial and viral loads were based on literature relationships for the decay constant (K) or T90 times (the time for 90% of bacteria/viruses to die) where environmental relationships were not available.

The light and temperature die-off relationships are based on the approach of Wilkinson (2001), which uses an average of several studies to determine appropriate die-off relationships for light and temperature and assumes that they are additive and independent of each other. Consequently, the total die-off rate (K_{tot}) is equal to:

$$K_{tot} = K_T + K_I \qquad (2)$$

Where K_T is the temperature dependent die-off and K_I is the irradiance dependent die-off rate. The proportion of bacteria (R) surviving after a given time period (dt) can be calculated as:

$$R=e^{-K_{tot} \cdot dt} \qquad (3)$$

And the T90 (in days) is related to K_{tot} by:

$$T90 = -\ln(0.1)/K_{tot} \qquad (4)$$

Light dependent die-off for faecal coliform bacteria is calculated using the average of several studies (Bellair et al., 1977; Evison, 1988; Pommepuy,1992), so that K_I is:

$$K_I = 0.371\ I_z\ 0.518 \qquad (5)$$

Where I_z is the irradiance (in W/m^2) at depth z (in m). Iz is related to the surface irradiance (I_0) by:

$$I_0 = I_z\, e^{-\alpha z} \qquad (6)$$

Where α is the attenuation coefficient, defined as 0.06 m^{-1} for the purposes of this study, based on an observed average value for the attenuation of 440 nm light for NZ coastal waters (Davies-Colley, 1992). Hourly surface irradiance data for the appropriate period is obtained from Napier airport using the NIWA CliFlo database.

The model mean of studies by: McFeters & Stuart (1972), Mitchell & Starzyk (1975), Flint (1987), Evison (1988), Qin et al. (1991) and Auer & Niehaus (1993) have been used to determine a temperature related die off rate (day^{-1}) for faecal coliform bacteria of:

$$K_T=10^{(0.11\ +\ 0.015\ T)} \qquad (7)$$

where T is the temperature. Water temperature data is obtained from the hydrodynamic model.

A light and temperature inactivation (die-off) relationship was not available for viruses, so we used an estimate of a T90 inactivation time. A representative inactivation time for pathogenic human viruses was difficult to determine, as a wide range of values have been reported in the literature. A review by Griffin et al. (2003) reports T90 times in the range of 1 day for polio and parvovirus (Wait & Sobsey 2001) to 57 days for infectious viruses (Wetz et al. 2004), with higher survivals noted for cooler waters. The review also states that associations with suspended sediments may allow greater survival, with one study reporting the isolation of human enteric viruses 17 months after disposal of sewage sludge (Goyal et al. 1984). Given the wide range of lifetimes observed for viruses in effluent, a moderately conservative (mid-range) value of 30 days was chosen as the T90 for the study.

Notably, the die-off for viruses and bacteria commences from when the plume is initialised within the far-field model. Thus, the conservative assumption is made in that the microbial concentrations entering the wastewater pipe are not subject to die-off until the surface plume is established. Based on dye study results this time was estimated at approximately 1.5 hours under dry weather flow conditions.

When the results of this environmentally sensitive modelled die-off were compared to regular monitoring data good agreement was observed. As a comparison, a conservative constant T90 time of 48 hours was also applied to the particles, however this lead to bacterial concentrations well in excess of observations from five monitoring sites located near the outfall. This result highlighted the importance of incorporating environmentally driven die-off in bacterial modelling without which predicted compliance would have been substantially underestimated by the model. It is possible that the constant 30 day T90 time for viruses had similar issues, but unfortunately the data was not available to test this.

Concentration calculations

One limitation of using particle tracking models is that results are returned as particle locations and their associated bacteria, sediment or virus load and not a concentration. As the guideline standards are expressed as concentrations it was necessary to convert the particle densities back to concentrations in a realistic fashion. Conversion is not a trivial task; consider the example of the basic box method, whereby concentrations are estimated by counting the total load of particles and dividing by the volume of the box. By varying the size of the box it is possible to change the value of the concentrations.

In order to ensure accurate and objective conversion from particle loads to concentrations for comparison with guideline standards we applied a novel method for assigning length-scales of areas. This was undertaken following the kernel method of Vitali et al. (2006). The method uses a variable bandwidth to reconstruct the concentration at each spatial location in the grid. The use of a variable bandwidth (kernel size) attempts to represent true variability of spatial concentration, while minimising statistical variability that inevitably occurs away from the source due to a necessarily finite number of particles. A small kernel is used in regions of high numbers of particles, where it is statistically appropriate to infer relatively small scale changes in concentration. A larger kernel in areas of low density prevents unrealistically high densities around the precise (but partially random) locations of a few isolated particles.

The areal density, D is computed as:

$$D(x,y,t)=\sum_{i=1}^{n}\frac{m_i}{\lambda_x(x,y,t)\lambda_y(x,y,t)}K\left(\left|\frac{x_i-x}{\lambda_x}\right|\right)K\left(\left|\frac{y_i-y}{\lambda_y}\right|\right) \qquad (8)$$

where n is the total number of particles, λx, λy are the kernel bandwidth in the x and y directions and K is the kernel function. The loading, mi, for each particle depends on the quantity being calculated, and may represent (for example) a bacterial load count.

Following Vitali et al. (2006), a Epanechnikov kernel function is used:

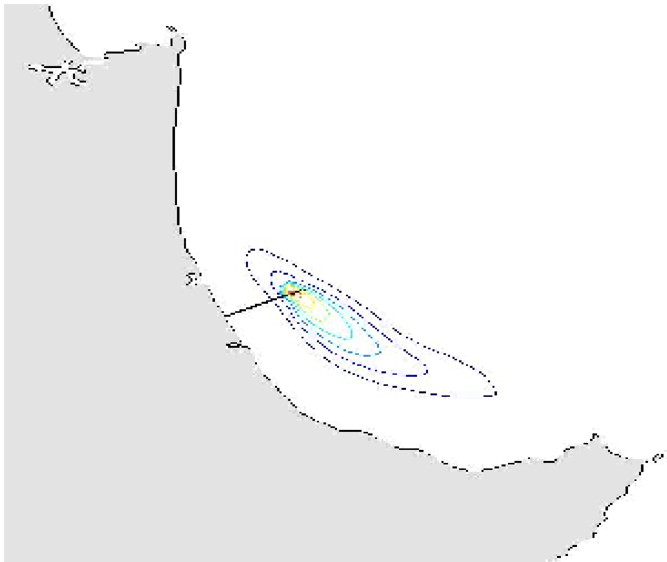
$$K(q)=\begin{cases}0.75\left(1-q^2\right), & |q|\le 1\\ 0 & |q|>1\end{cases} \qquad (9)$$

and a receptors based method (a modification of their RL3) is used to define the bandwidths. The bandwidths are defined as twice the standard deviation of the projected distance in the x or y direction of any particles in the neighbourhood of a grid point. The neighbourhood is defined as the region enclosing the 1/20th closest particles. The aspect ratio (e.g. λx/λy) of the bandwidths are limited to be no greater than 5:1 to prevent unrealistically elongated kernels, with the smaller value increased.

Areal densities (per m2) are calculated in predefined slabs by only considering particles within specified depth limits; densities are then converted to concentrations (per m3) by dividing by the slab thickness. For a well mixed water column with neutrally buoyant particles, the concentration is representative of the complete depth range. In the case of the bacteria and viruses a surface layer of one meter thickness was used due to the relatively low vertical dispersion of neutrally buoyant particles associated with the buoyant brackish water plume. By using this technique we were confident that hourly particle loads and positions are accurately converted to surface water concentrations in an objective manner.

After calculating the water-borne concentrations of bacterial and virus concentrations we were then able to undertake comparison to the contact recreation standards described in the coastal plan.. An example of the type of annual percentage compliance is presented in the following figure (Fig 11).

Figure 11: An example of compliance map for the Clive Outfall, showing decreasing levels of compliance with proximity to the outfall. Scale



4 CONCLUSIONS

Our source to sea approach to estimating spatial compliance for a coastal outfall enables planners, stakeholders and regulators to visualise both near and far-field effects of bacterial and viral wastes associated with the outfall plume. We have used 2 year-long periods so that the majority of conditions experienced at the site will be included in our analysis. The coupling of a proven near-field mixing model (CorMix) to a particle tracking model driven by a high-resolution 3D hydrodynamic model enables important initial fine scale mixing processes to be resolved from the outfall ports whilst including the interaction effects with background bacterial and viral concentrations.

Although the general methods we have employed are present in other studies (e.g. Conolly et al. 1999; Signell et al. 2000), the techniques presented here make use of custom “kiwi-made” solutions (e.g. PartTracker and the application of environmentally sensitive die-off of bacteria). We recognise that one major obstacle to the wide application of the techniques presented here is the availability of validated 3D high resolution coastal hydrodynamic models. Freely available data provided by coarse-scale models such as the CSIRO BlueLink datasets allow oceanic scale data to drive regional coastal models in a realistic fashion, making the construction of regional models more accessible. However, the effort involved in the construction of validated regional high resolution models should not be underestimated. In the study presented here, local scale winds proved an important driver of coastal circulation in the region and a significant investment of time has been needed to resolve these winds. Indeed this may not be an isolated case and in order to resolve currents (particularly surface currents) realistically may require realistic fine-scale wind drivers for other coastal areas. Nevertheless, the increasing availability of fine-scale meteorological forecast data may make this less of an issue in the future.

The initial investment in construction of these local scale models can allow them to be recycled to address other issues associated with marine transport within their constructed domains. There are increasing demands to provide spatially explicit estimates of effects for developments that discharge wastes or pollutants into the coastal environment. Examples of such developments are: sediment run-off from road building projects, nutrients from fin fish farms or diluted diffuse effluent from land intensification, all of which may add to existing coastal pressures cumulatively. As a result it is likely that these models will become increasing useful tools for displaying likely environmental effects to stakeholders in order to inform an educated debate on the pros and cons of coastal developments and mapping possible mitigation pathways. As with any study of this nature there are always areas that may be improved. We hope that the publication of our approach will aid discussions towards the improvements that will increase the availability of spatially explicit information for other coastal development projects.

ACKNOWLEDGEMENTS

The contribution to the study from Bob McWilliams (Wastewater Treatment Manager, HDC) is gratefully acknowledged. We also wish to thank Nick Houston (HDC) for project management support and to Neil Daykin from the Hawke's Bay Regional Council for various data and background environmental information.

Data provided from international meteorological and oceanographic data has been pivotal in the construction of hydrodynamic models. We wish to acknowledge the provision of data of NCEP Reanalysis Derived data provided by the NOAA/OAR/ESRL PSD, Boulder, Colorado, USA, from their Web site at http://www.esrl.noaa.gov/psd/. The Australian Government, through the Commonwealth Bureau of Meteorology, Royal Australian Navy and CSIRO are all thanked for the provision of BlueLink data used as a basis for the boundary conditions used in the regional model.

REFERENCES

Auer, M.T. and Niehaus, S.L. (1993). 'Modeling fecal coliform bacterial. I: Field and laboratory determination of loss kinetics.' *Water Research* 27, 4, 693-701.

Bellair J.T., Parr-Smith G.A., Wallis I.G. (1977) 'Significance of diurnal variations in fecal coliform die-off rates in the design of ocean outfalls.' *Water Pollution Control Federation* 49, 9, 2022-2030.

Connolly J.P., Blumberg A.F., and Quadrini J.D. (1999) 'Modeling fate of pathogenic organisms in coastal waters of Oahu, Hawaii' *Journal of environmental engineering*, 125, 398-406.

Davies-Colley R.J. (1992) 'Yellow substance in coastal and marine waters round the South Island, New Zealand.' *New Zealand Journal of Marine and Freshwater Research* 26, 311-322.

Egbert, G. and Erofeeva, L. (2002) 'Efficient inverse modelling of barotropic ocean tides.' *Journal of Atmospheric and Oceanic Technology*, 19, N2

Evison, L.M. (1988). 'Comparative studies on the survival of indicator organisms and pathogens in fresh and sea water.' *Water Science and Technology*, 20, 11/12, 309-315.

Flint, K.P. (1987). 'The long-term survival of Escherichia coli in river water.' *Journal of Applied Bacteriology*. 63, 3, 261-270.

García-Martínez, R. and Flores-Tovar, H. (1999). 'Computer modelling of oil spill trajectories with a high accuracy method.' *Spill Science and Technology Bulletin* 5, 5, 6323–330.

Goyal, S.M., Adams, W.N., O'Malley, M.L. and Lear, D.W. (1984). 'Human pathogenic viruses at sewage sludge disposal sites in the Middle Atlantic region.' *Applied and Environmental Microbiology* 48, 4, 758-763

Griffin, D.W., Donaldson, K.A., Paul, J.H. and Rose J.B. (2003). 'Pathogenic Human Viruses in Coastal Waters.' *Clinical Microbiology Reviews* 16, 1, 129-143.

HBRC (1999) 'Regional Coastal Plan, June 1999.', pp 221. ISBN 1-877174-16-5

Jirka, G.H., Doneker, R.L. and Hinton, S.W. (1996). 'User's Manual for CorMix: A hydrodynamic mixing zone model and decision support system for pollutant discharges into surface waters.'

Kanamitsu, M., Ebisuzaki, W., Woollen, J., Yang, S.-K., Hnilo, J.J., Fiorino, M. and Potter, G.L. (2002). 'NCEP-DEO AMIP-II Reanalysis (R-2)' *Bulletin of the Atmospheric and Metererolgical Society*, 1631-1643.

Knight, B.R., Zyngfogel, R. and Forrest, B. (2009). 'PartTracker – a fate analysis tool for marine particles.' In Proceedings of Australasian Coasts and Ports Conference 2009. Dawe I, Ed. New Zealand Coastal Society, Wellington. pp 8.

Lonin, S. A. (1999). 'Lagrangian model for oil spill diffusion at sea.' *Spill Science and Technology*, 5 ,5-6, 331–336.

McFeters, G.A. and Stuart, D.G. (1972). 'Survival of coliform bacteria in natural waters: field and laboratory studies with membrane-filter chambers.' *Applied and Environmental Microbiology* 24, 5, 805.

Mellor, G.L. (2004) Users guide for "A three-dimensional, primitive equation, numerical ocean model". Princeton University, Princeton, NJ. (http://www.aos.princeton.edu/WWPUBLIC/htdocs.pom/)

Melnick, J.L. (1976). Viruses in water: an introduction. In Viruses in Water. G. Berg (ed.). American Public Health Association, Washington, D.C. 3-10.

Metcalf, T.G., Melnick, J.L. and Estes, M.K. (1995). 'Environmental virology: from detection of virus in sewage and water by isolation to identification by molecular biology-a trip of over 50 years.' *Annual Reviews in Microbiology* 49, 1, 461-487.

MfE (2008) 'Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas' Ministry for the Environment, pp 159. ISBN: 0-478-24091-0

Mitchell D. and Starzyk, M. (1975). 'Survival of Salmonella and other indicator microoganisms.' *Canadian Journal of Microbiology* 21.

Pommepuy M., Guillaud J.F., Dupray E., Derrien A., Le Guyader F., Cormier M. (1992) 'Enteric bacteria survival factors.' *Water Science and Technology*. 25, 12.

Press, W.H., Flannery, B.P., Teukolsky, S.A. and Vetterling, W.T. (1992). 'Adaptive Stepsize Control for Runge-Kutta.' Chapter 16.2 In: Numerical Recipes in FORTRAN 77: The Art of Scientific Computing, Cambridge University Press, ISBN 0-521-43064-X.

Qin D., Bliss, P.J., Barnes, D and FitzGerald, P.A. (1991). 'Bacterial (total coliform) die-off in maturation ponds.' *Water Science and Technology*, 23, 7, 1525-1534.

Sattar, S.A. (1978). 'Viral Pollution of the Ottawa River and its Possible Impact on the Quality of Potable and Recreational Waters in the Ottawa Area - Phase I.' Final Report for the Ontario Ministry of the Environment Research Study, Contract No. 77-044-11. pp 103

Schiller, A., Oke, P.R., Brassington, G.B., Entel, M., Fiedler, R., Griffin, D.A. and Mansbridge, J. (2008). 'Eddy-resolving ocean circulation in the Asian-Australian region inferred from an ocean reanalysis effort.' *Progress in Oceanography* doi:10.1016/j.pocean.2008.01.003.

Scire, J.S., Robe, F.R., Fernau, M.E. and Yamartino, R,J. (2000) 'A User's Guide for the CALMET Meteorological Model (Version 5).' Earth Tech, Inc., 196 Baker, Avenue, Concord, MA 01741.

Signell R.P., Jenter H.L., and Blumberg A.F., (2000) 'Predicting the physical effects of relocating Boston's sewage outfall.*' Estuarine, Coastal and Shelf Science*, 50, 59-71.

Shaw, J., Peck, S., Sekla, L., Allen, M., Sumarah, R., Leggart, E., Grouin, D., Terry, D., Taylor, M., Tobin, R. and Robertson, W. (1992). 'Guidelines for Canadian Recreational Water Quality.' Authority of the Minister of National Health and Welfare. pp 100.

Song, Y. and Haidvogel, D. (1994). 'A semi-implicit ocean circulation model using a generalized topography-following coordinate system.' *Journal of Computational Physics*, 115, 228–244.

Vitali, L., Monforti, F., Bellasio, R., Bianconi, R., Sachero, V., Mosca, S. and Zanini, G. (2006). 'Validation of a Lagrangian dispersion model implementing different kernel methods for density reconstruction.' *Atmospheric Environment*, 40, 8020-8033.

Wait, D.A. and Sobsey, M.D. (2001). 'Comparative survival of enteric viruses and bacteria in Atlantic Ocean seawater.' *Water Science Technology*, 43, 139-142.

Wetz, J.J., Lipp, E.K., Griffin, D.W., Lukasik, J., Wait, D., Sobsey, M.D., Scott, T.M. and Rose, J.B. (2004). 'Presence, infectivity, and stability of enteric viruses in seawater: relationship to marine water quality in the Florida Keys.' *Marine Pollution Bulletin*, 48, 7-8, 698-704.

Wilkinson R 2001. 'Modelling Faecal Coliform Dynamics In Streams And Rivers.' University of Wales, Aberystwyth, United Kingdom.

Zhang, H.M., Bates, J.J. and Reynolds, R.W. (2006). 'Assessment of composite global sampling: Sea surface wind speed.' *Geophysical Research Letters*, 33, L17714, doi:10.1029/2006GL027086.

Zhang, Y.-L. and Baptista, A.M. (2008). 'SELFE: A semi-implicit Eulerian-Lagrangian finite-element model for cross-scale ocean circulation.' *Ocean Modelling*, 21, 3-4, 71-96.