# Raglan Wastewater Treatment Plant - Water Quality Assessment

Prepared for Waikato District Council
Prepared by Beca Limited

**31 October 2019**


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## Contents



## Appendices

## Revision History

| Revision Nº | Prepared By | Description | Date |
|---|---|---|---|
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## Document Acceptance

| Action | Name | Signed | Date |
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| Prepared by | **Gemma Wadworth** | | 31/10/2019 |
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| on behalf of | Beca Limited | | |

# 1 Introduction

Waikato District Council (WDC) currently operate the Raglan Wastewater Treatment Plant (WWTP) under an existing discharge consent which is due to expire on the 14th February 2020. Section 124 of the Resource Management Act (RMA) allows an applicant to continue operating under an existing discharge consent whilst applying for a new discharge consent. WDC seek to utilise section 124(2) of the RMA to lodge an application for a relatively short-term resource consent at least three months prior to consent expiry.

WDC are currently developing a long-term resource wastewater management preferred solution and an associated application for the necessary resource consents will be sought imminently. The short-term resource consent will allow the long-term preferred option to be investigated and completed adequately.

WDC has engaged Beca Ltd (Beca) to undertake a water quality effects assessment to support the short-term resource consent application and Assessment of Environmental Effects (AEE).

This Water Quality Assessment seeks to assess the effects of the existing discharge regime from the WWTP on the coastal water quality of the Whāingaroa Harbour.

## 1.1 Scope and Objectives

The scope and objectives of the water quality assessment are to:

- Describe the water quality of the existing environment;
- Assess the characteristics of the existing discharge with respect to key water quality parameters, timing, duration and flow rate;
- Summarise hydrodynamic modelling undertaken to determine the zone of reasonable mixing and predicted dilutions of the discharge in the receiving environment;
- Predict concentrations of key water quality parameters in the receiving environment as a result of dilution; and
- Compare the predicted concentrations of key water quality parameters against relevant guidelines to assess whether any adverse effects are likely to occur on human health and the environment.

This assessment references the standards and terms, and address the assessment criteria, relevant to water quality effects under rule 16.3.8 of the Waikato Regional Coastal Plan (WRCP). These are:

**Standards and Terms**

- The discharge, after initial mixing, shall not result in:
  - The production of conspicuous oil or grease films, scums or foams, or floatable or suspended materials; or
  - Any conspicuous change in the colours or visual clarity; or
  - Any emission of objectionable odour

**Assessment Criteria**

In assessing any application for sewage discharges, regard shall be had to:

- The Decision-Making Criteria and Considerations which are set out in Appendix II of the WRCP, and which are relevant to this activity;
- The extent to which water quality has been maintained or enhanced; and
- The extent to which the discharge will or is likely to have any adverse effects on human health; and
- The extent to which, after initial mixing, the discharge (either by itself, or in a combination with other discharges) will or is likely to result in any adverse effects on aquatic flora or fauna.

Relevant considerations set out in Appendix II of the WRCP include:

- Whether or not the quality of discharge will meet the standards required, after initial or reasonable mixing, for contact recreation purposes as stated in the third schedule of the RMA.
- Whether or not the discharge contains nutrients which will cause undesirable biological growth;
- The extent to which the discharge, after initial and reasonable mixing, results in:
  - The production of any conspicuous oil or grease films, scums or foams, or floatable or suspended materials;
  - Any conspicuous change in the colour or visual clarity;
  - Any emission of objectionable odour;
  - Any significant adverse effects on aquatic life

The assessment also considers Section 107 of the Resource Management Act 1991 (RMA) which states that a discharge is not allowed if, after reasonable mixing, the discharge (either by itself or in a combination with the same, similar or other contaminants or water), is likely to give rise to certain effects in the receiving environment. These effects are the same as the final four points outlined above.

## 2 Description of the Existing Wastewater Treatment Plant

The WWTP is located to the south-west of the Raglan community on Wainui Road. Wastewater is received at the inlet works (screen), from where wastewater is piped to anaerobic ponds 1 and 2, then aerated ponds A and D, and on to ponds B and C as represented in Figure 1 below. The aerobic ponds have an aeration system and aquamats installed. The aquamats provide additional surface area for biological activity. The pond treated wastewater currently discharges into a day pond for storage prior to discharge on the outgoing tide. If the holding capacity of the day pond is exceeded, it will overflow to the roadside (storage) pond. From the day pond treated wastewater is pumped via an inline UV disinfection system to the entrance of the Whāingaroa Harbour.

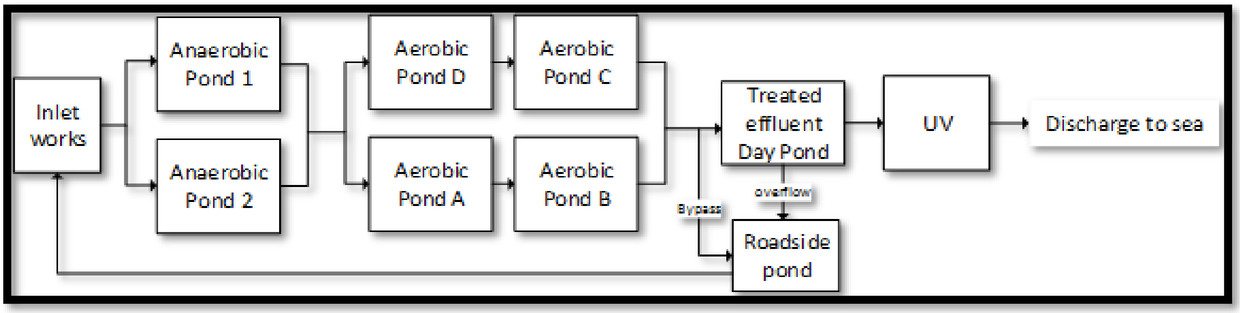


Figure 1: Existing process at Raglan Wastewater Treatment Plant

# 3 Description of the Receiving Environment

## 3.1 Contaminant Inputs and Residence times in the Harbour

The Whāingaroa Harbour has a large catchment area of 445 $km^2$ with seven major river catchments and smaller streams flowing into the Harbour. These include the Ohautira, Opoturu, Waingaro, Tawatahi and Waitetuna Rivers (Fisher, 2014). Due to the large-scale of the catchment, surrounding catchment land uses have an impact on the Harbour water quality whereby contaminants generated from land uses can be transported via rivers to discharge into the Harbour.

The contribution of contaminants from catchment diffuse sources into the Harbour is expected to be greater than that from the existing WWTP discharge and has been previously studied using hydrodynamic modelling (Greer, 2015) ('the modelling study'). The modelling study sought to develop an understanding of inputs to the harbour from the WWTP using hydrodynamic modelling in conjunction with a 13-river catchment model to enable some comparison of inputs from the WWTP compared to contaminant inputs from inflowing rivers.

The modelling study concluded that rivers entering the Harbour have a large-scale influence on Harbour water quality relative to the WWTP discharge with regards to Faecal Coliform (FC) concentrations. It was noted that the model did however carry some level on uncertainty with regards to predicted source FC concentrations.

Both diffuse contaminant sources associated with catchment land uses and the WWTP discharge influence a range of water quality parameter concentrations including pathogens, nutrients, and suspended solids (Greer, 2015). Pathogens, including a wide range of microbes are the primary concern with regard to human health given the recreational nature of the Harbour and use of the Harbour for collection of kaimoana.

Residence times have an impact on the fate of contaminants discharged to the Harbour (eCoast, 2016). Through hydrodynamic modelling (eCoast, 2016), residence times of 35 to 45 days in the Harbour have been observed (in the upper estuary during medium river flow conditions). In particular, the Waituna and Waingaro Rivers drain approximately 60% of the Harbour Catchment area and flow into the Harbour head where residence times can be up to 45 days during low river flows (eCoast, 2016). Conversely, lower residence times are predicted to occur at the mouth of the Harbour (eCoast, 2016), where the WWTP discharge is located.

Variables other than river flows also contribute to residence times spatially within the Harbour. Tidal influences are such that residence times of contaminants at the mouth of the Harbour, in the vicinity of the WWTP discharge, can be <1 day (based on interpretation of tracer experimentation where a threshold of 20% reduction in initial tracer concentration was applied, when the tracer was released at high tide (eCoast, 2016)). Under the same tidal conditions, areas of the Harbour influenced by river discharges (predominantly in the upper Harbour) maintain relatively long residence times and consequent tracer/contaminant retention (eCoast, 2016). This provides further evidence that the Harbour water quality is more influenced by contaminants transported by river flows than by the WWTP outfall discharge. Overall, the modelling undertaken by eCoast found that in drowned river valley estuaries such as the Whāingaroa Harbour, residence times exhibit an increasing gradient from the mouth to the head of the estuary (Figure 2).

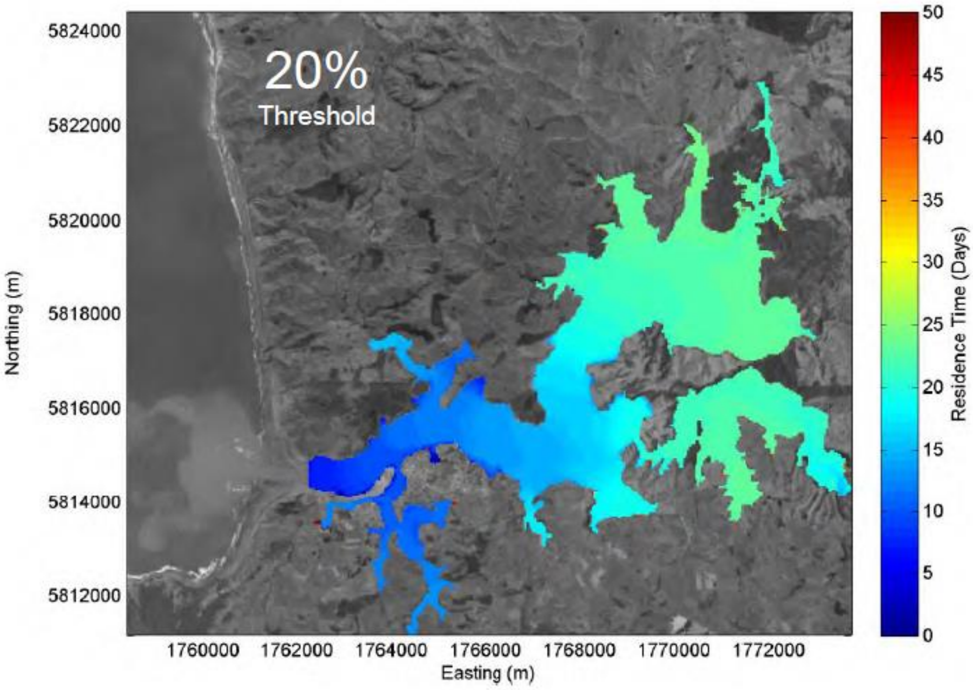


**Figure 3.9: Residence time for Whaingaroa Harbour calculated using medium flow. The tracer was released at high tide during mid-range tides. The threshold at a cell was considered flushed was taken to be 20% of the original tracer concentration.**

Figure 2: Excerpt from eCoast, 2016 showing gradient of residence times in the Harbour by which 20% of the original concentration of a tracer released at high tide had been reached.

## 3.2 Existing Water Quality

Maintaining a high level of water quality in the Harbour is important due to its uses for recreation including swimming, water sports, along with collection of kaimoana. Faecal indicator species are used to monitor microbial water quality in the harbour including E.coli, Enterococci and Faecal Coliforms.

The general microbiological water quality of the harbour was assessed by NIWA using data from the routine monitoring undertaken in the Harbour by Waikato Regional Council (WRC)[1] (Appendix A). NIWA reported that concentrations of faecal indicator species tend to consistently decrease from the upper Harbour toward the mouth of the Harbour suggesting substantial dilution and attenuation of indicator species from inflow toward the Harbour mouth. The 95th percentile concentration of Enterococci calculated from data for January 2017 to June 2019 at the Raglan Mouth was 15 Enterococci /100 mL while at two upper Harbour monitoring locations 95th percentile concentrations for the same period were 186 and 412 Enterococci /100 mL, respectively.

NIWA also assessed data for two recreational sites in the Harbour where the number of samples collected, and sampling frequency met the criteria for comparison to the Ministry for the Environment New Zealand recreational quality guidelines (2003). NIWA reported that in 12 of the 16 recreational seasons assessed, gastrointestinal illness risk was likely to be less than 1% based on the faecal indicator date, indicating high recreational water quality. In three of the seasons, the gastrointestinal illness risks were likely to have

[1] *Microbiological wate quality assessment for Raglan Harbourr*, NIWA, October 2019.

exceeded 5%, with a greater than 10% illness risk likely in one recreational year (reflecting a single sample that contained an unusually high Enterococci concentration).

Waikato Regional Council also monitor other key biological and chemical water quality parameters such as:

- Salinity
- pH
- Dissolved Oxygen (DO)
- Chlorophyll A (Chl-A)
- Nitrite + Nitrate Nitrogen (NNN)
- Ammonia ($NH_4$)
- Total Nitrogen (TN)
- Dissolved Reactive Phosphorous (DRP)
- Total Phosphorous (TP)
- Total Suspended Solids (TSS)
- Water clarity (by Secchi Disk (SD))

Six sites are actively monitored by WRC (since October 2017), their locations are shown in Figure 3.


Figure 3: Approximate locations of active WRC marine water quality monitoring sites in the Raglan Harbour.

Median concentrations of selected parameters monitored by WRC are presented below in Table 1. Higher median nutrient and TSS concentrations appear to occur near the head of the harbour compared with concentrations reported for locations closer to the mouth. This may be attributed to spatial variation in residence times as discussed above and the influence of freshwater sources.

Table 1: Median concentrations of selected parameters monitored by WRC

| Parameter | Opoturu | Waingaro | Waitetuna | Raglan Mid Harbour | Raglan Mouth | Wainui |
|---|---|---|---|---|---|---|
| Chl-A (mg/m³) | 7.9 | 10 | 12 | 7.9 | 6.4 | 7.2 |
| DO (g/m³) | 7.6 | 7.7 | 7.7 | 7.7 | 7.8 | 7.7 |
| DO (%Sat) | 98.8 | 99.7 | 98.4 | 100 | 101 | 101 |
| DRP (g/m³-P) | 0.0079 | 0.01 | 0.012 | 0.0079 | 0.0064 | 0.0072 |
| $NH_4$ (g/m³-N) | 0.028 | 0.021 | 0.022 | 0.023 | 0.017 | 0.028 |
| NNN (g/m³-N) | 0.028 | 0.018 | 0.04 | 0.017 | 0.012 | 0.011 |
| TN (g/m³-N) | 0.20 | 0.25 | 0.30 | 0.18 | 0.16 | 0.17 |
| TP (g/m³-P) | 0.018 | 0.028 | 0.030 | 0.022 | 0.018 | 0.017 |
| TSS (g/m³) | 9.9 | 18.5 | 25.5 | 14.6 | 11.9 | 13.8 |
| SD (m) | 1.3 | 0.86 | 0.63 | 1.0 | 1.4 | 1.2 |

WRC report various guidelines and standards used to assess estuarine water quality for ecological health.[2] The reported median dissolved oxygen (% saturation) and ammonia values for all monitoring sites in Table 7 are within the 'Excellent' WRC guideline category of >90% and <0.1 g/m³, respectively. The median total phosphorous concentrations for all sites are either at or below the upper 'satisfactory' criteria value of 0.03 g/m³. However, the median NNN at four sites in the Harbour are indicative of elevated nutrients, being within the 'unsatisfactory' categories of >0.015 g/m³. (assuming NNN is in the form of nitrate). Median NNN concentrations at two sites near the mouth of the Harbour and discharge outfall are lower and within the 'satisfactory' category for nitrate. Lower NNN concentrations near the mouth of the Harbour compared to those closer to upper reaches of Harbour arms could be due to a lesser influence from freshwater inputs to the Harbour near the mouth.

Though water quality in the Harbour can be expected to fluctuate over short periods of time due to events such as heavy rainfall and seasonality, key indicators of water quality outlined above appear to have remained relatively stable since monthly monitoring commenced in 2017 at each of the active WRC Harbour water quality monitoring locations. In summary, the above data shows that the water quality in the harbour mouth is generally good, and representative of open coastal water. However, water quality declines towards the estuary (to the west of the harbour mouth) due to land-based influences, particularly during times of rainfall.

[2] https://www.waikatoregion.govt.nz/environment/environmental-information/environmental-indicators/coasts/estuarine-water-quality-report/estuarine-water-quality-techinfo/

# 4 Discharge Characteristics

### 4.1.1 Location of the Discharge

Treated wastewater from the existing WWTP discharges from an outfall at the mouth of the Whāingaroa Harbour near Wainamu Road shown in Figure 4 below. The entrance to the Harbour is bordered by sandy beaches with Wainamu Beach to the south and Rangitoto Point to the north. The Harbour has a deep central channel (up to 20m deep), which has been carved out by strong tidal flows just inside of the harbour entrance. Within the Harbour, adjacent to the main township, the sediment within the main channel consists of shelly sand and shelly gravel. Moving up the Harbour, the channels become more shallow and narrow and the sediment becomes finer, consisting of sandy mud. Extensive intertidal and shallow subtidal sand and mud flats occur throughout the numerous tributary arms of the harbour.


Figure 4: Approximate location of the outfall (Google Earth 2019)

### 4.1.2 Existing Quality of the Discharge

Treated wastewater quality data from monthly grab samples of final treated wastewater between 2014 and 2018 was reviewed by Beca. Annual median and 90th Percentile concentrations of key water quality parameters were calculated for each year between 2014 and 2018. The averages of the respective annual median concentrations are presented in Table 2 and compared to the Resource Consent discharge limits where applicable.

Table 2: Summary of Treated Wastewater Quality 2014-18 (Pond Treated wastewater post UV)

| Parameter | Median Consent | Average Annual Median | 90th Percentile Consent | Average Annual 90th Percentile |
|---|---|---|---|---|
| Carbonaceous biochemical oxygen demand (cBOD5) (mg/L) | 10 | 6.5 | 20 | 12 |
| Total Suspended Solids (TSS) (mg/L) | 20 | 33 | 30 | 84 |
| Faecal Coliforms cfu/100mL | 14 | 5 | 43 | 28 |
| Enterococci no. /100mL | - | 7 | 35 | 17 |
| Ammoniacal Nitrogen ($NH_4$-N) (mg/L) | - | 7 | - | 14 |
| Nitrate Nitrogen ($NO_3$-N) (mg/L) | - | 10 | - | 18 |
| Total Nitrogen (TN) (mg/L) | - | 21 | - | 26 |

Overall the WWTP has been performing well over the past five years, however the pond system is producing peaks of TSS from time to time (associated with algal growth in the pond based treatment system), and as a result the WWTP discharge does not meet the associated consent limits for TSS. The WWTP treated wastewater $cBOD_5$ concentrations over the last five years have been reasonably consistent and below the consented discharge limits.

The average annual median Faecal Coliform concentration for the past 5 years of 5 cfu/100 mL is 34% of the consent limit while the average annual 90th Percentile concentration of 28 cfu/100 mL is 65% of the consent limit. The average annual 90th Percentile Enterococci concentration of 7 no./100 mL is just 49% of the consent limit.

Ammoniacal nitrogen, nitrate nitrogen and total nitrogen are not monitored as part of the existing resource consent compliance, however WDC has been collecting these data in addition to the existing consent requirements. These parameters are assessed with regard to water quality effects in Section 5 of this report.

### 4.1.3 Existing Discharge Regime

An analysis of the existing discharge regime has been undertaken by DHI Water & Environment Ltd (DHI) and presented in the report ‘Raglan Wastewater Treatment Plant Discharge Assessment’, dated October 2019[3] (**Appendix B**). The analysis used SCADA data from the WWTP for the period between January 2015 and May 2019. The analysis found that the median duration of the discharge is 2 hours 15 minutes while the median start time of the discharge is 15 minutes after high tide with an average flow rate of 0.058 $m^3/s$.

The existing resource consent requires the discharge to commence no earlier than 30 minutes prior to high tide. The DHI analyses of the distribution of the WWTP discharge timing compared to high tide shows that the discharge commences after high tide approximately 67% of the time. This demonstrates that WDC have been proactive with respect to maximising the amount of treated wastewater that is discharged after high tide that can be flushed more efficiently from the Harbour with the outgoing tide. The discharge timing has been optimised to commence after high water to provide a greater degree of dilution and minimise the possibility of treated wastewater flowing eastwards towards the Raglan township.

An excerpt from the DHI analysis showing the example timing of the Raglan WWTP discharge using SCADA data and Manu Bay tide gauge data is presented in Figure 5.

[3] *Raglan Wastewater Treatment Plant Discharge Assessment*, DHI Water & Environment Ltd, October 2019.

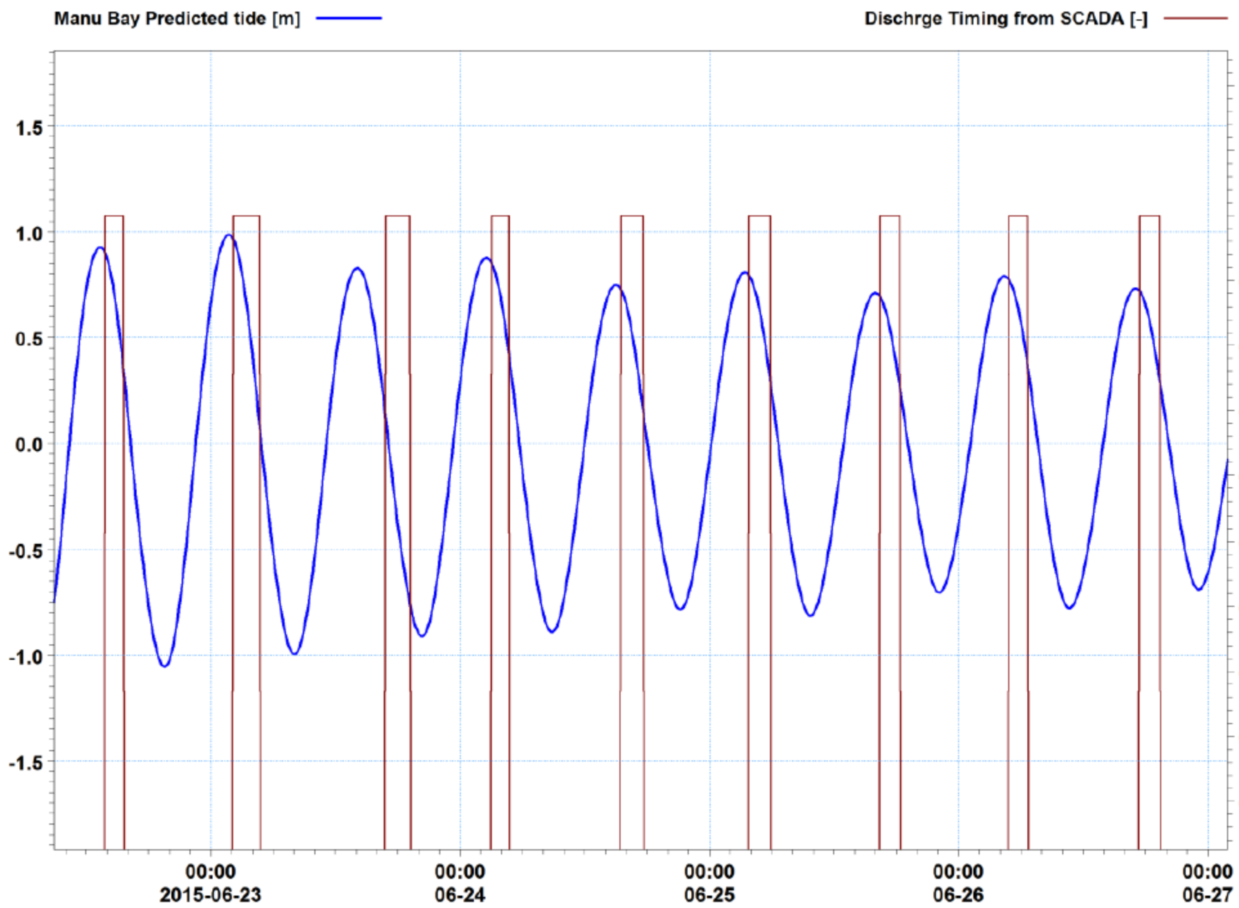


Figure 5: Example timing of the Raglan WWTP discharge using SCADA data and Manu Bay tide gauge data

# 5 Water Quality Assessment

Concentrations of contaminants in the discharge have been assessed with respective predicted dilutions at the zones of initial mixing and reasonable mixing. Potential impacts on the receiving environment as a result of the treated wastewater discharge are then assessed using relevant human health and ecological guidelines.

## 5.1 Zone of Reasonable Mixing and Predicted Dilutions

Both the RMA and WRCP require that certain effects outlined in Section 1.1 of this report do not occur beyond the zone of reasonable mixing. It is considered that adopted water quality guidelines apply after the zone of reasonable mixing. Additionally, the WRCP states that regard to should be had to certain effects after initial mixing.

Initial Mixing is defined under the WRCP as:

> "*The first phase of the mixing of a discharge with receiving water. In the case of sewage effluent being discharged to waters of the CMA, initial mixing refers to all mixing process that occur between the effluent leaving the discharge structure and reaching the surface of the receiving water*".

DHI also undertook near field dilution CORMIX modelling where near field dilutions were defined at the distance where there is strong initial mixing. Because of the relatively low discharge rate of the existing regime, the near field zone was found to be less than 10 m. Modelling was undertaken for different phases of the tide and found that lowest dilutions occur just after high tide when ambient currents are lowest. At this time the near field dilution is predicted to be 12.4-fold, however such conditions are only expected to occur for approximately 5% of the time. At other tide phases near field dilutions range from 14.5 – 42.4-fold.

The WRCP defines the Reasonable Mixing Zone as:

> "*The zone within which a discharge would dissipate into the existing waters. The zone will be defined on a case-by-case basis by consideration of location, size, shape, outfall design and in-zone quality*".

Decision making criteria under the regional coastal plan also considers the various section 107 effects.

The zone of reasonable mixing was modelled by DHI using CORMIX simulations, details of the modelling undertaken by DHI are appended in **Appendix B**. Based on the CORMIX simulations, DHI report a typical zone of reasonable mixing of 150 m from the discharge and predict that the minimum dilution achieved at this point is 70-fold under the current discharge regime.

## 5.2 Far Field Dilutions

DHI has also modelled far field dilutions of the existing discharge regime. The following figure shows the 5$^{th}$ percentile dilution (i.e. dilutions of greater than this occur for 95 percent of the time) achieved for the 2018 model simulations under the current discharge regime of 1,175 m$^3$/day. The flow rate and timing of the discharge is derived from the 2018 discharge monitoring data. Figure 6 is presented for a conservative tracer (which quantifies the degree of physical mixing).

Figure 6 shows that beyond 150m from the outfall the 5th percentile dilutions are greater than 1000-fold, beyond 700 m from the outfall the 5th percentile dilutions are greater than 2000-fold and that beyond 1200 m the 5th percentile dilutions are greater than 4000-fold. These levels of dilution are achieved due to a combination of the near-field dilution achieved and because the discharge only occurs for a portion of the outgoing tide resulting in significant levels of dilution between subsequent discharges.

Beyond around 1000 m from the outfall there is a significant degree of dilution with the 5th percentile dilution often exceeding 30000-fold.

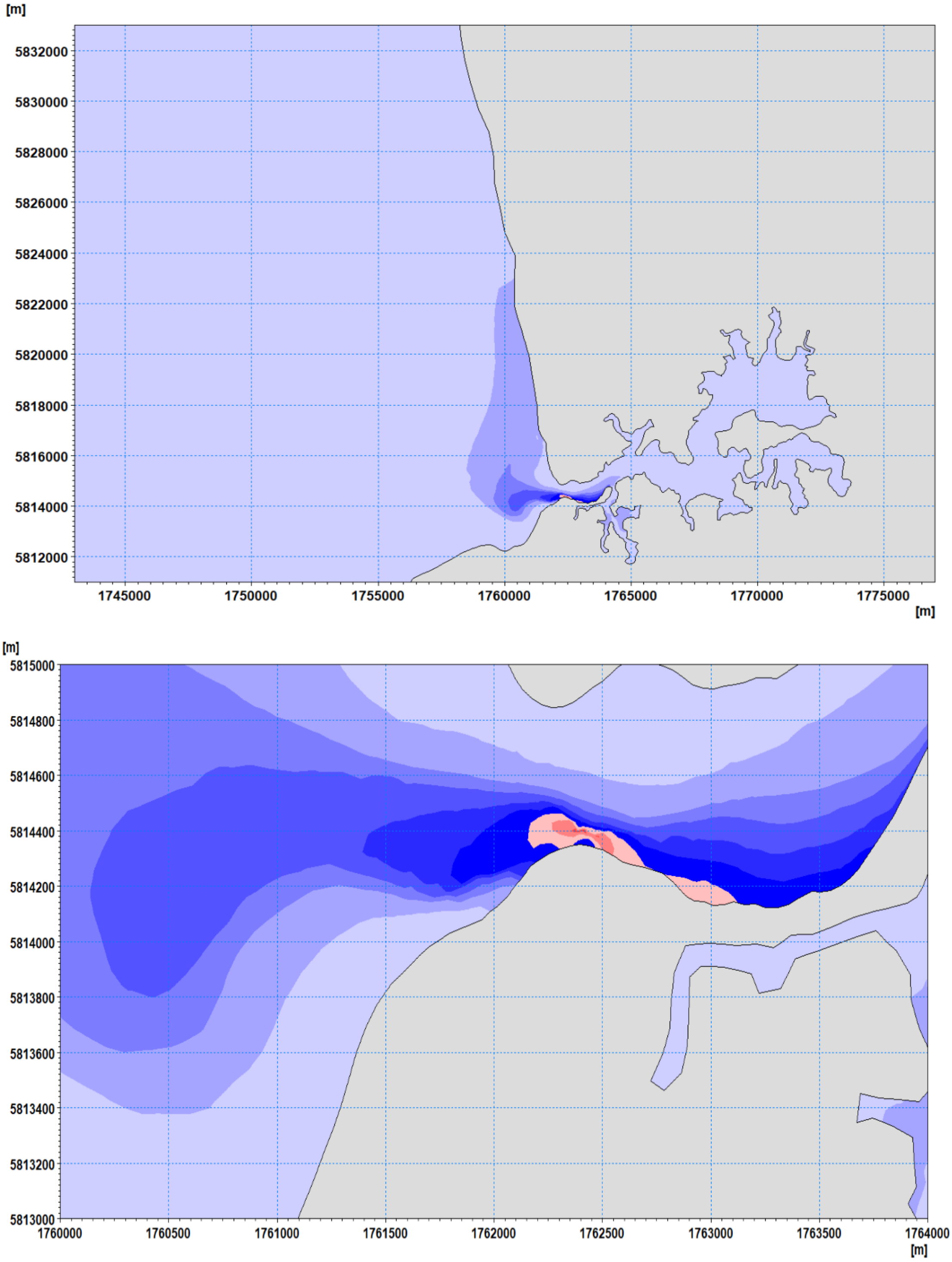


Figure 6: 5th Percentile Surface Layer Dilution for a Conservative Tracer Under the Current Discharge Regime and Timing

## 5.3 Predicted Concentrations

The predicted 70-fold dilution at the edge of the zone of reasonable mixing and the worst-case scenario near field mixing dilution have been applied to the average annual median concentrations for each of the water quality parameters presented in Table 2 (except for ammoniacal nitrogen where the average annual 90th percentile concentration has been used in dilution calculations) to determine likely concentrations in the receiving environment. Resulting concentrations after applying the dilution factors are shown in Table 3.

Predicted concentrations are compared to the guidelines and standards published by WRC which are used to assess estuarine water quality for ecological health, contact recreation and for shellfish gathering.[4] The WRC guidelines use three categories, ‘excellent’, ‘satisfactory’ and ‘unsatisfactory’. This assessment is used as an indicative assessment and does not take into account background concentrations of contaminants.

Predicted nutrient concentrations are also compared to ANZECC (2000) guidelines for fresh and marine water quality. Site specific guidelines have not been developed for New Zealand marine environments. The guidelines suggest that consideration can be given to the use of interim trigger values for slightly disturbed inshore ecosystems in south-east Australia. The WRC guidelines and standards are also based upon the south-east Australia ANZECC (2000) interim trigger values.

[4] https://www.waikatoregion.govt.nz/environment/environmental-information/environmental-indicators/coasts/estuarine-water-quality-report/estuarine-water-quality-techinfo/

Table 3: Predicted concentrations after zone of initial mixing and reasonable mixing compared to adopted guidelines.

| Parameter | Discharge Concentrations | | Concentration with 70-fold dilution (dilution after reasonable mixing) | Concentration with 12.4-fold dilution (worst case near field dilution) | ANZECC 2000 Guideline | WRC Estuarine Guideline Categories | | |
|---|---|---|---|---|---|---|---|---|
| | Average Annual Median | Average Annual 90th Percentile | | | | Excellent | Satisfactory | Unsatisfactory |
| Faecal Coliforms cfu/100mL | 5 | 28 | < 1 | 0.38 | - | <2[a] | 2 – 14[a] | >14[a] |
| Enterococci /100mL | 7 | 17 | < 1 | 0.54 | - | <28[b] | 28 – 280[b] | >280[b] |
| $NH_4$-N (mg/L) | 6.4 | 14 | 0.2* | 1.1* | 0.91 | <0.1[c] | 0.1 – 0.91[c] | >0.91[c] |
| $NO_3$-N (mg/L) | 10 | 18 | 0.15 | 0.83 | 0.005 | <0.005[d] | 0.005 – 0.015[d] | >0.015[d] |
| TN (mg/L) | 21 | 26 | 0.30 | 1.7 | 0.12 | - | - | - |
| $cBOD_5$ (mg/L) | 6.5 | 12 | 0.09 | 0.52 | - | - | - | - |
| TSS (mg/L) | 33 | 84 | 0.48 | 2.7 | - | - | - | - |

Notes: *Average annual median concentrations used for dilution calculations except for ammonia where the average annual 90th percentile concentration has been used; a = human health guideline for shellfish-gathering; b= human health guideline for contact recreation; c= guideline for ecological health (toxicity to fish); d = guideline for ecological health (nuisance plant growth).

## 5.4 Effects Assessment

### 5.4.1 Microbiological indicators

After both initial mixing and reasonable mixing both Faecal Coliform and Enterococci concentrations are predicted to be well within the ‘Excellent’ WRC categories for shellfish gathering and contact recreation, respectively.

NIWA has also undertaken a human health risk assessment for the discharge[5] (**Appendix C**). Quantitative Microbial Risk Assessment (QMRA) techniques were used to assess human health risks arising from possible exposure to pathogens at various contact recreation and shellfish gathering sites in the Harbour. NIWA concluded that high initial dilution of the discharged treated wastewater is achieved and as a consequence, infection and illness risks to both recreation water users and consumers of uncooked shellfish are generally low for all pathogens modelled, and at all sites where exposure to diluted treated wastewater may occur.

### 5.4.2 Toxicants

Ammonia can result in toxicity effects on aquatic life, particularly in fish. The predicted concentration after the zone of initial mixing (1.1 mg/L) is within the ‘unsatisfactory’ category and above the adopted ANZECC (2000) guideline. However, the concentration of 1.1 mg/L is only expected to occur for approximately 5% of the tide (i.e. about 45 minutes per tide) based on the conditions which a 12.4 fold dilution are predicted to occur under, and in relatively close proximity to the discharge. The next most conservative near field dilution is 15-fold (occurring near the end of the discharge window during neap tide), resulting in an ammoniacal-nitrogen concentration of 0.93 mg/L. For the remaining time (i.e. under the 6 other tidal phases modelled by DHI for near field mixing; **Appendix B**), dilutions result in predicted concentrations below the adopted ANZECC (2000) guideline.

The predicted ammoniacal-nitrogen concentration beyond the zone of reasonable mixing (0.2 mg/L) is within the ‘satisfactory’ WRC guideline category and below the adopted ANZECC (2000) guideline (0.91 mg/L)

These results indicate that adverse toxicity effects on aquatic life are unlikely to occur as a result of the discharge given at reasonable mixing the ammoniacal nitrogen concentration is predicted to be below the adopted ANZECC 2000 guideline and at initial mixing the ammoniacal concentration is only expected to exceed the guideline for only a short period of time during certain tidal conditions (under worst case 90%ile concentrations). . It is noted that the ANZECC guideline is not however specific to New Zealand marine waters an should be used with caution.

### 5.4.3 Nutrients

Nitrate concentrations after both the zone of initial mixing and reasonable mixing are above the adopted ANZECC guideline and are in the ‘unsatisfactory’ WRC guideline category (>0.015 mg/L). Total nitrogen is also above the adopted ANZECC guideline. Nitrate and total nitrogen are not considered toxicants and are therefore not expected to cause adverse effects to aquatic flora and fauna in the receiving environment. Predicted concentrations of total nitrogen are consistent with those reported for WRC monitoring sites that are discussed in Section 3.2. Further significant dilution is expected to occur beyond the zone of reasonable mixing, in the order of 30,000-fold beyond 1 km from the outfall as detailed in the DHI report (**Appendix C**), which will minimise the overall contribution of nitrate and total nitrogen from the WWTP in the wider Harbour. Additionally, the tidal nature of the Harbour and relatively short residence time at the Harbour mouth, as detailed in Section 3.1, inhibit the ability for nuisance plant growth to occur. Effects on benthic macroalgal

[5] *Human health risk assessment Raglan WWTP*, NIWA, October 2019.

growth are highly unlikely to occur due to the rapid tidal currents in the area and rapidly moving sand benthic environment in the area receiving treated wastewater.

### 5.4.4 Biochemical Oxygen Demand and Total Suspended Solids

WDC propose to amend the TSS discharge concentration allowed under the consent to 40 $g/m^3$ compared to the existing discharge limit of 20 $g/m^3$. The receiving environment is expected to have relatively high levels of suspended sediment resulting from tidal movements, wind and wave action. It is therefore expected that contributions of TSS (which are largely in the form of algae) from the WWTP discharge will be negligible as a result of the proposed discharge.

$cBOD_5$ is not a contaminant of concern in treated wastewater and concentrations measured in the discharge are below the relevant consent limits and are therefore considered unlikely to have an adverse effect on the water quality of the receiving environment. Dilution is rapid and any organic enrichment effects in receiving waters are highly unlikely.

### 5.4.5 Other potential effects of the discharge

No conspicuous oil or grease films, scums or foams, or changes in colour or visual clarity have been observed after initial mixing.

No objectionable odour is expected to occur after initial mixing or reasonable mixing as the discharge is treated wastewater that is aerated. As the wastewater is aerated, compounds such as hydrogen sulphide that would produce objectionable odours should not be present.

# 6 Summary

Predicted dilutions after initial and reasonable mixing have been applied to concentrations of contaminants in the Raglan WWTP discharge. Resulting concentrations from the dilutions have been predicted. The proposed discharge is the same as the current discharge regime (subject to optimisation of the discharge timing under condition 11 and the proposed amendment to condition 14 in relation to TSS) therefore water quality in the receiving environment is expected to be maintained, at the same level as under the existing discharge.

Based on the predicted microbiological indicator concentrations, the quality of the discharge will meet WRC contact recreation and shellfish collection standards after initial mixing and reasonable mixing. The QMRA assessment undertaken by NIWA also concludes that illness risks related to Adenovirus and Norvirus as a result of the discharge appear generally low.

Significant adverse effects on aquatic life are also unlikely to occur based on the predicted ammoniacal-nitrogen concentration being within the ‘satisfactory’ WRC guideline category after reasonable mixing, while the worst-case concentration at initial mixing may be within the ‘unsatisfactory’ WRC guideline category for around only 5% of the time (under worst case 90% concentrations in the treated wastewater). For the majority of the time after initial mixing, the predicted ammoniacal-nitrogen concentration is within the ‘satisfactory’ WRC guideline category.

Physical elements of the harbour, including low retention times near the mouth of the harbour where the outfall is located, are expected to negate nuisance biological growth effects and predicted concentrations after reasonable mixing are reflective of background concentrations observed in wider WRC water quality investigations.

After both initial and reasonable mixing, the discharge is not expected to result in:

- The production of conspicuous oil or grease films, scums or foams, or floatable or suspended materials; or
- Any conspicuous change in the colours or visual clarity; or
- Any emission of objectionable odour

Overall it is considered that beyond the zone of reasonable mixing of 150m, there are predicted to be negligible adverse effects on the water quality of the Whāingaroa Harbour.

# A

Appendix A – NIWA Microbiological water quality assessment

# Microbiological water quality assessment for Raglan Harbour

*Prepared for BECA*

*October 2019*

Prepared by:
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| Quality Assurance Statement | | |
|---|---|---|
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| | Formatting checked by: | Alison Bartley |
| | Approved for release by: | Michael Bruce |

# Contents



## Tables

## Figures

# Executive summary

Waikato District Council wishes to renew the consent to discharge treated wastewater from the Raglan wastewater treatment plant. Currently treated wastewater is discharged to an outfall located on the south shore of the harbour mouth. As part of the renewal process, WDC is considering several options, including continuation of the current discharge, discharge to an off-shore outfall, and disposal of treated wastewater to land. More advanced wastewater treatment options (to produce higher-quality effluent) are also being considered.

NIWA was engaged to assess the risk of illness to recreational water users associated with the discharge of treated wastewater. That assessment relies on a modelling approach and is the subject of a separate report. In this report, the general microbial quality of harbour water is assessed using data derived from routine monitoring; these data are also used to estimate the likely impact that treated wastewater has on harbour microbial water quality. Review of available data indicates:

## Wastewater treatment efficacy:

- Wastewater quality (2015-2018) is reasonably consistent, with Faecal Indicator Bacteria (FIB) concentrations typically less than 100 enterococci or faecal coliforms/100 mL.
- The treatment process (which includes UV irradiation) consistently reduces FIB concentrations by approximately five-log orders, from approximately 10,000,000 faecal coliforms/100 mL to 100 faecal coliforms/100 mL.

## General harbour microbiological quality

- Microbiological water quality in the upper harbour is largely determined by freshwater inflows, particularly by the two largest inflows.
- Concentrations of all FIB consistently decrease from the upper harbour toward the mouth, suggesting substantial dilution and attenuation of FIB from inflows toward the Harbour mouth.
- Generally low FIB concentrations near the Harbour mouth also suggest that the wastewater does not have a persistent impact on water quality near the Harbour mouth. An intensive water quality survey undertaken in the 1994/95 recreation season also suggested that the plume of diluted wastewater does not impact on the inner harbour in the vicinity of the Campground and the pedestrian bridge where recreational access and activity is concentrated.

## Recreational water quality

- More than 12 data exist at each of two recreational sites for a total of 16 recreation seasons over the period 1994/95 to 2017/18.
- In 12 of these seasons, the gastrointestinal illness risk was likely to have been less than 1% (the "no observable adverse effect level), indicating high recreational water quality.

- In three of these seasons, illness risks were likely to have exceeded 5%, with a greater than 10% illness risk likely in one recreational year (2016/17).
- In the 2016/17 season, this higher risk status reflects a single sample that contained an unusually high enterococci concentration.

# 1 Introduction

Raglan wastewater treatment plant (WWTP) is operated by Waikato District Council (WDC), who currently have a consent to discharge treated wastewater via an outfall pipe located at the mouth of the harbour. The influent is primarily domestic sewage with minor/minimal trade waste. WDC wishes to renew the consent to discharge treated wastewater to the Harbour. In recognition of the sensitivity of the local community (including impact on the mauri of the harbour), WDC is considering several options for discharge of the treated wastewater. These include:

- continued discharge at the existing site, either using the existing outfall pipe, or using a new outfall pipe (to allow discharge of the wastewater further offshore, nearer the centre of the channel, which would also allow the pipe to be buried)
- disposal of the total discharge volume to land by irrigation
- discharge of the total volume of wastewater via a new outfall located outside of the harbour, or
- a hybrid option, where land disposal would be favoured when soil moisture conditions permitted, with the existing or one of the new outfall options used when soil conditions are unfavourable for disposal to land.

As part of the consent renewal process, WDC is also considering increasing the level of wastewater treatment. The WWTP currently comprises traditional 'passive' waste stabilization ponds with UV irradiation of treated wastewater, with limited options for treatment to improve the quality of treated wastewater. Options for improving the wastewater treatment process identified to date include use of membrane filtration, chemical dosing and increased UV disinfection; upgrade of the pond system to a high rate pond (HRP) could also be considered – these perform more effectively and consistently than conventional pond systems with regard to microbial quality. Selection of the wastewater treatment option will be determined by several factors, including the location and nature of the future discharge. Another important factor that will direct the level of future wastewater treatment is the health risk associated with the discharge. Despite being well-managed, all treatment systems have the potential to be sources of pathogens – exposure to these pathogens carries the risk of infection and illness.

The harbour is a well-used recreational and highly-valued cultural resource. Two marae are located within the Raglan Harbour (Whaingāroa Harbour) catchment: Poihākena Marae (Ngāti Tahinga of Tainui) is located in Raglan on Wainui Road, Whaingāroa, and Waingaro Marae (Ngāti Maahanga and Ngāti Tamainupo) is located approximately 25 kilometres northeast of Raglan, on Waingaro Landing Road. The location of the wastewater treatment plant and current outfall in relation to the town and the two marae is indicated in Figure 1-1.

To assist with the process of evaluating wastewater treatment options, and to facilitate discussion with the community, NIWA was engaged to provide the following services:

1. Undertake a health risk assessment (described separately in (Hudson 2019)).
2. Consider microbial water quality data derived from routine monitoring programmes and the impact of these contaminants on harbour water quality – this report.


**Figure 1-1: Location of Raglan WWTP and wastewater outfall in Raglan Harbour.**

## The objectives of this report

Data derived from regional water quality monitoring programmes were not focused on the wastewater discharge – they provide information regarding other sources of faecal contaminants, the fate of these contaminants in the harbour, and the relative magnitude of FIB contributions from freshwater inflows vs. the wastewater discharge.

The primary objectives include summarising historical and current wastewater microbiological water quality, discharge characteristics and relevant compliance data. Relevant receiving environment water quality data (e.g., data derived from Waikato Regional Council State of Environment or recreational water quality monitoring programmes) were also to be reviewed. The information derived from this review was to create a harbour microbiological water quality 'context', which would (as far as the data would allow), enable the impact of the current and future discharge to be assessed. The detailed health risk assessment of the wastewater discharge was also considered within this context.

# 2 Materials and methods

## 2.1 Data used in the assessment

Microbiological water quality data in major inflows to Raglan Harbour, and at sites across Raglan Harbour were obtained from Waikato Regional Council (WRC). Details of available data and sites used in this assessment are identified in Table 2-1. The location of these sites is indicated in Figure 2-1. Wastewater water quality and discharge data are summarised in Table 2-2.

**Table 2-1: Water quality monitoring sites operated by Waikato Regional Council.** Sites in bold were particularly important for this water quality assessment.

| Site code | Site type | Site name |
|---|---|---|
| **1167_4** | **Waingaro River** | **Ruakiwi Rd Off SH22** |
| **1247_2** | **Waitetuna River** | **Te Uku-Waingaro Rd** |
| **857_1** | **Raglan Harbour** | **Motor Camp North Of Bridge** |
| 857_10 | Raglan Harbour | Inner Harbour North (Bottom) |
| 857_11 | Raglan Harbour | Inner Harbour South (Surface) |
| 857_12 | Raglan Harbour | Inner Harbour South (Bottom) |
| 857_13 | Raglan Harbour | Mid Harbour (Surface) |
| 857_14 | Raglan Harbour | Mid Harbour (Bottom) |
| 857_15 | Raglan Harbour | Outer Harbour (Surface) |
| 857_16 | Raglan Harbour | Outer Harbour (Bottom) |
| **857_2** | **Open coast** | **Ngarunui Beach** |
| 857_23 | Raglan Harbour | Raglan Wharf |
| 857_24 | Opotura River | River 1 |
| 857_25 | Raglan Harbour | Opotoru 1 |
| 857_26 | Raglan Harbour | Bridge 1 |
| 857_27 | Raglan Harbour | Wainui 1 |
| 857_28 | Raglan Harbour | Harbour 1 |
| 857_29 | Raglan Harbour | Causeway |
| **857_30** | **Raglan Harbour** | **Waingaro** |
| **857_31** | **Raglan Harbour** | **Waitetuna** |
| **857_32** | **Raglan Harbour** | **Raglan Mid Harbour** |
| **857_33** | **Raglan Harbour** | **Raglan Mouth** |
| **857_34** | **Raglan Harbour** | **Wainui** |
| 857_9 | Raglan Harbour | Inner Harbour North (Surface) |

**Table 2-2: Wastewater monitoring sites and data used for treatment plant assessment.** Provided by WDC.

| Wastewater monitoring sites | Period for which data were available | No. results | | |
|---|---|---|---|---|
| | | Faecal coliform conc. (n/100 mL) | Enterococci conc. (n/100 mL) | Discharge data (daily total volume, L) |
| Raglan WWTP wastewater inflow | January 2015 – February 2018 | 51 | | |
| Raglan WWTP post-UV wastewater outflow | January 2015 – February 2018 | 47 | 31 | |
| Discharge data | June 2016 - April 2019 | | | 1058 |

## 2.2 Methods used for this assessment

All data were manipulated using Microsoft Excel or Systat for Windows v13.2 and analysed using Systat for Windows v13.2.

Selected statistical data were selected for spatial display as maps using QGIS v4.3. Ninety-fifth percentile values were plotted at the location of the sample collection point as bar graphs, with the data on each map scaled relative to the largest value in each data set.


**Figure 2-1: Location of water quality monitoring sites operated by Waikato Regional Council.** Sites are located in major inflows to Raglan Harbour, across Raglan Harbour, and one site on the shore of the Tasman Sea. Site details are listed in Table 2-1.

# 3 Results and discussion

## 3.1 Selected wastewater treatment plant characteristics

Figure 3-1A provides a time series of WWTP discharge data, which shows that it is highly variable. The fifth and ninety-fifth percentile vales are 6.4 and 24.2 L/s respectively, and the median discharge is 10.3 L/s (summary statistics are included in Appendix B). The discharge of wastewater is timed to commence at highwater, so the timing of discharge over a day will vary according to the tidal stage. The wastewater discharge has a distinct seasonal cycle, with median flow rate increasing by approximately 50% between March and July annually (Figure 3-1B). The reasons for this seasonal behaviour are not immediately obvious, but it probably reflects infiltration of surface water to the sewers during the wet season.

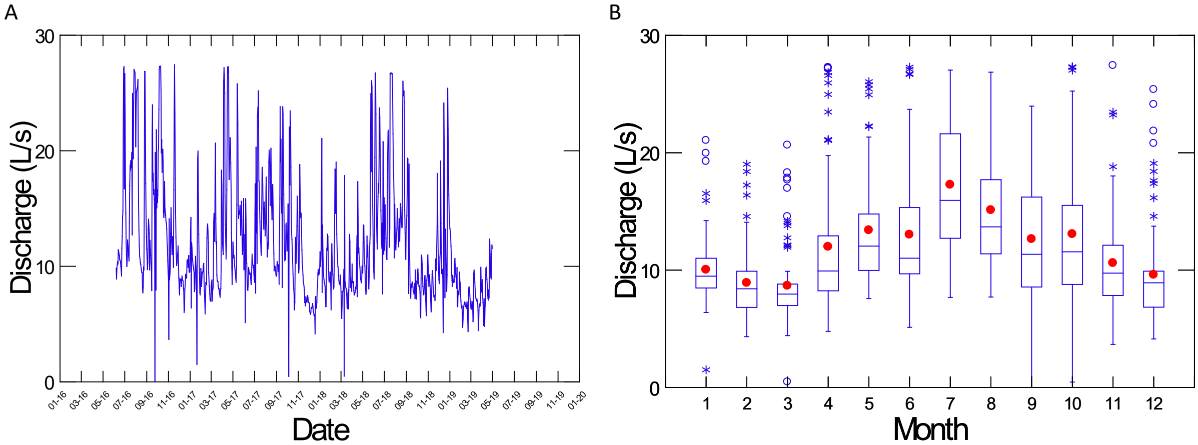



**Figure 3-1: Raglan WWTP daily average discharge data (A) and seasonal variation in daily discharge (B).** In the box and whisker plot the red dot indicates daily average discharge. An explanation of the symbols used in a box and whisker plot is included in Appendix A. The red dot indicates the monthly average value.

Wastewater microbiological quality is summarised in Figure 3-2 for faecal coliform and enterococci concentrations. Data are not available for enterococci in the inflow to the wastewater treatment plant. Although data are not available for *E. coli* either, approximately 90% of faecal coliforms are *E. coli*.[1] The faecal coliform data indicate an approximately five log reduction[2] in faecal coliform numbers through the WWTP, with indication of slight improvement in discharge quality over time.

[1] Personal communication, Dr Rob Davies-Colley, NIWA Hamilton.

[2] By convention treatment efficacy is described in terms of log numbers, where "one $\log_{10}$" (referred to as one log reduction) indicates ten-fold reduction in contaminant concentration, and five $\log_{10}$ indicates 100,000-fold reduction (five-log reduction).

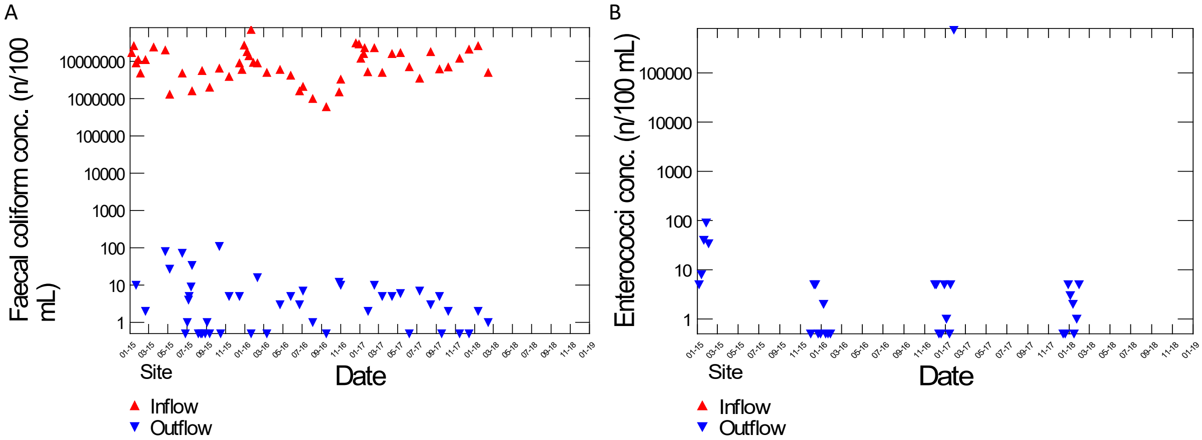



**Figure 3-2: Inflow and outflow concentrations for faecal coliforms (A) and enterococci (B).** Inflow enterococci concentrations were not available. The y-axis has $\log_{10}$ scale.

A similar order of reduction is indicated in terms of faecal coliform flux (Figure 3-3) – derived by multiplying concentration n/100 mL by discharge (L/s) (and adjusting for unit change). For both concentration and flux, available data indicate very consistent performance over time, which suggests a reasonably consistent discharge microbial water quality is likely. Although inflow data are not available for enterococci concentration, the outflow concentrations are also consistently low (ninety-fifth percentile <100 enterococci/100 mL). The relatively low numbers of Faecal Indicator Bacteria (FIB) reflect the UV treatment of the wastewater prior to discharge. The consistent, relatively small range of concentrations and flux of FIBs indicates that suspended sediment concentrations in the wastewater are also consistent. High suspended sediment concentrations reduce the efficacy of UV irradiation, because they attenuate the light penetration and effectively shade FIB and viruses. This does not appear to be a concern for this discharge.

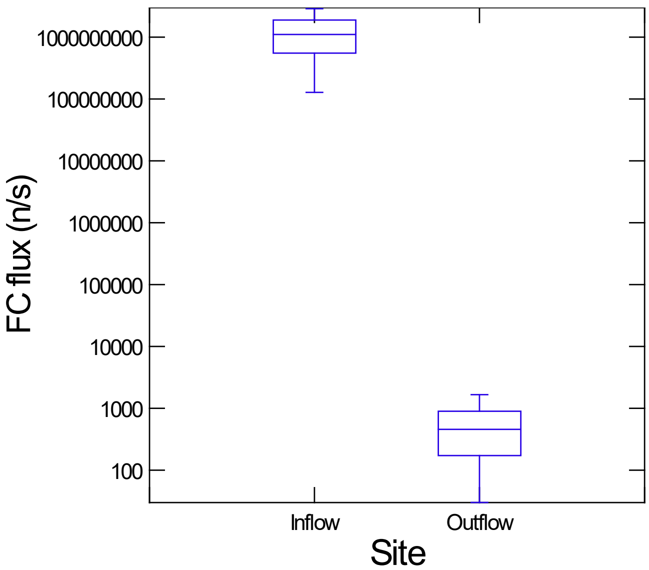



**Figure 3-3: Inflow and outflow flux for faecal coliforms.** Flow data were not available for all concentration data, and no inflow enterococci concentrations were available. The y-axis has $\log_{10}$ scale.

These data indicate that the microbiological water quality of the effluent is of a reasonably high and consistent standard. Although the concentrations of FIB such as faecal coliform and enterococci cannot be simply extrapolated to infer concentrations of pathogenic organisms such as viruses, these data suggest that efficient and consistent virus inactivation may be anticipated.

## 3.2 Receiving water quality data

Time series plots for FIB in samples collected from Raglan Harbour and major inflows to the harbour are summarised in Appendix D (*E. coli* - Figure D-1; enterococci - Figure D-2, and faecal coliforms - Figure D-3). Concentrations of all three FIB are highly variable over time. A feature of these figures is that the major tributary inflows to the harbour (sites 1167_4 and 1247_2) consistently have the highest concentrations of FIB. These results are similar to those reported previously by Greer et al. (2015), who related the proportion of freshwater (expressed as dilutions) to concentrations of faecal coliforms in the harbour to. Highest faecal coliform concentrations occurred in the arms of the harbour subject to tributary inflow, where salinities were lowest and dilutions were smallest.

When relating measured concentrations of FIB to human health, it is informative to consider ninety-fifth percentile concentration values, because these provide an indication of infrequent but more extreme microbiological conditions, when human health risks are likely to be greatest.

Eight sites across Raglan Harbour were selected for assessment, as well as one site on the Tasman Sea coast, outside of the Harbour. These sites were selected to provide a longitudinal series from the major freshwater inflows to Harbour mouth and adjacent shoreline.

- Figure 3-4 indicates 95$^{th}$ percentile enterococci concentrations at the sample location during the period Jan 2017-June 2019, and Figure 3-5summarises the complete data set for this variable for each site over this period.
- In Figure 3-6, enterococci concentrations for the period 2016-2019 are shown for these sites according to year.
- Similar figures for *E. coli* and faecal coliforms are provided in Appendix C.

These data confirm that microbiological water quality in the Harbour is largely determined by freshwater inflows, and that there appears to be little impact of Harbour water quality at the open ocean beach site. One exception is observed in 2017 (Figure 3-6), when elevated enterococci concentrations were observed at two sites in the lower Harbour, and on the open coast outside the harbour. The data available does not indicate whether the source was the wastewater discharge, or an elevated concentration arising from the small catchment discharging to the harbour at the Motorcamp. The hydrodynamic data used for the Quantitative Microbial Risk Assessment (Oldman 2019) suggests that wastewater is infrequently transported into the Harbour, and the performance data summarised in Figure 3-2 and Figure 3-3 indicates that during this period the WWTP was discharging treated wastewater with low FIB concentrations (typically <100 enterococci/100 mL). This suggests that the source of the elevated enterococci concentrations was probably a stream inflow.

Earlier review of Harbour water quality by Greer et al. (2015) included modelling of 13 inflow streams, as well as hydrodynamic modelling of the Harbour. Elevated FIB concentrations were related to greater proportions of freshwater in the harbour, suggesting that stream inflows were the dominant sources of faecal coliforms and enterococci. From their modelling they concluded that the wastewater load "is small relative to the riverine inputs and it is consistently not visible against the background river flows".

It would be incorrect to conclude from the results of Greer et al. (2015) that the wastewater discharge has negligible impact on harbour water quality. Although the freshwater inflows from streams and rivers have elevated FIB concentrations and loads, these are indicator organisms, derived primarily from non-human sources (Soller et al. 2010). This does not mean that they are harmless - these non-human (primarily livestock) sources may be associated with bacterial and protozoan pathogens of humans (e.g., *Campylobacter, Cryptosporidiium*), but not human viruses. The treated wastewater, in contrast, is derived from human wastes, and is likely to contain at least some viral contaminants likely to create the potential for human health risk, as well as (potentially) bacterial and protozoan pathogens. The work of Greer et al. (2015) does however indicate that wastewater effects are likely to be highly localised around the wastewater outfall, and, with ebb tide discharge, the contaminant plume is most likely to be consistently transported out of the harbour.

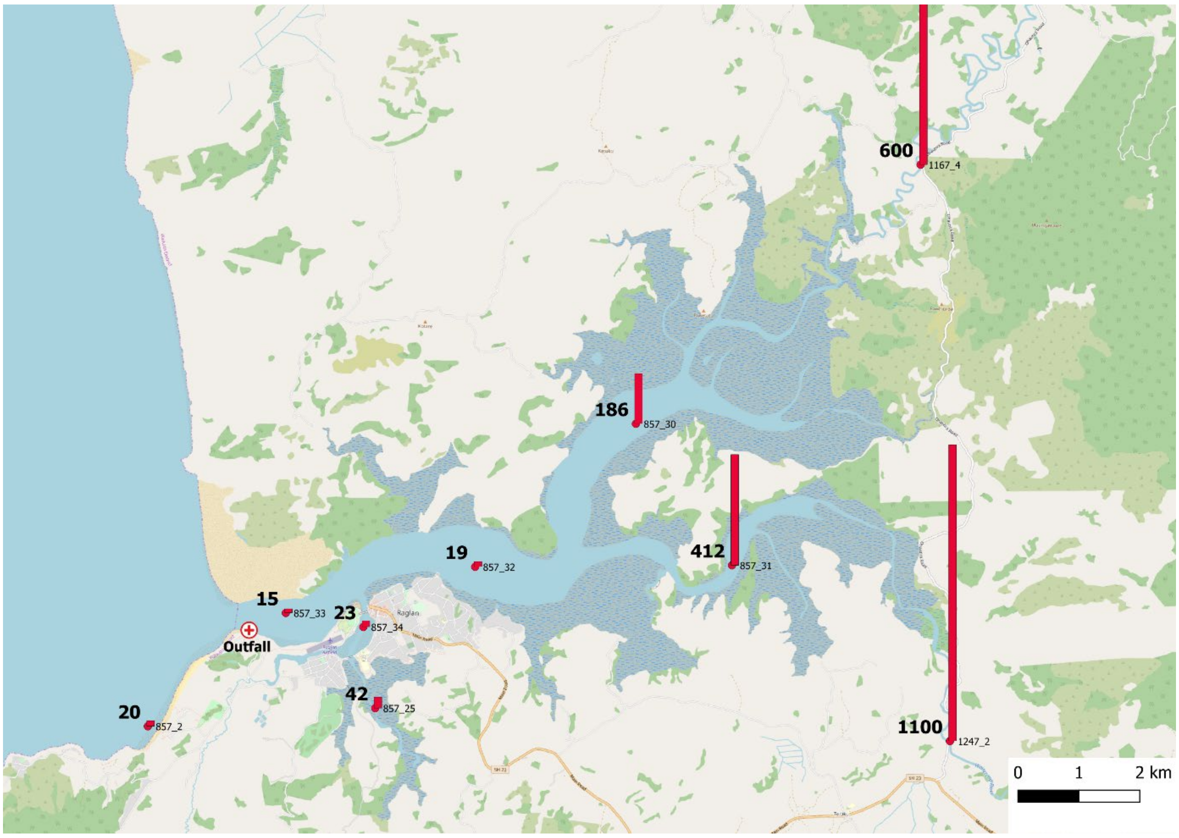



**Figure 3-4: Ninety-fifth percentile enterococci concentration calculated from data for January 2017 – June 2019.** Bars are scaled according to the highest value in the data set. The 95th percentile value is shown as the larger, bold number. The smaller number is the WRC site code. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1.

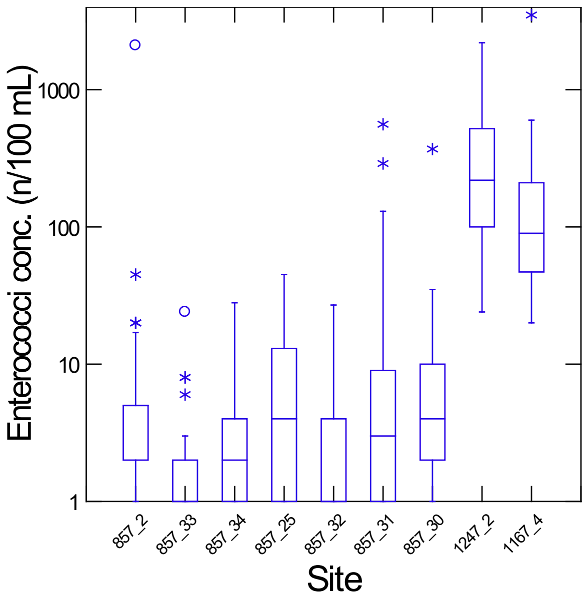
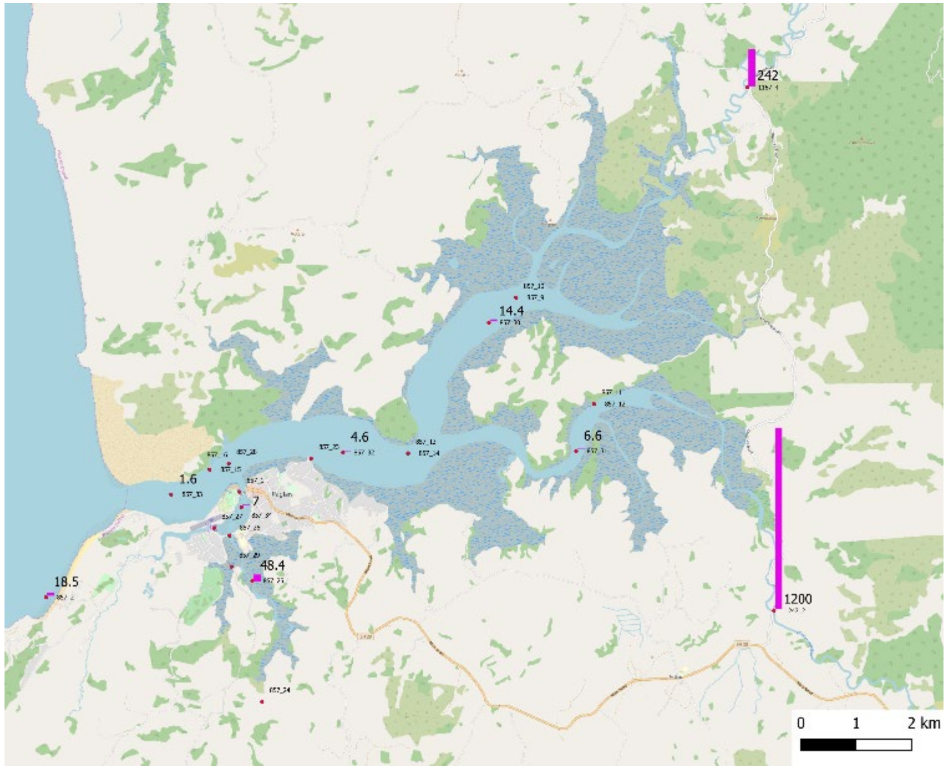



**Figure 3-5: Enterococci concentration data for the period January 2017- June 2019.** An explanation of the symbols used in a box and whisker plot is included in Appendix A. The y-axis has $\log_{10}$ scale. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1.

2019




2018

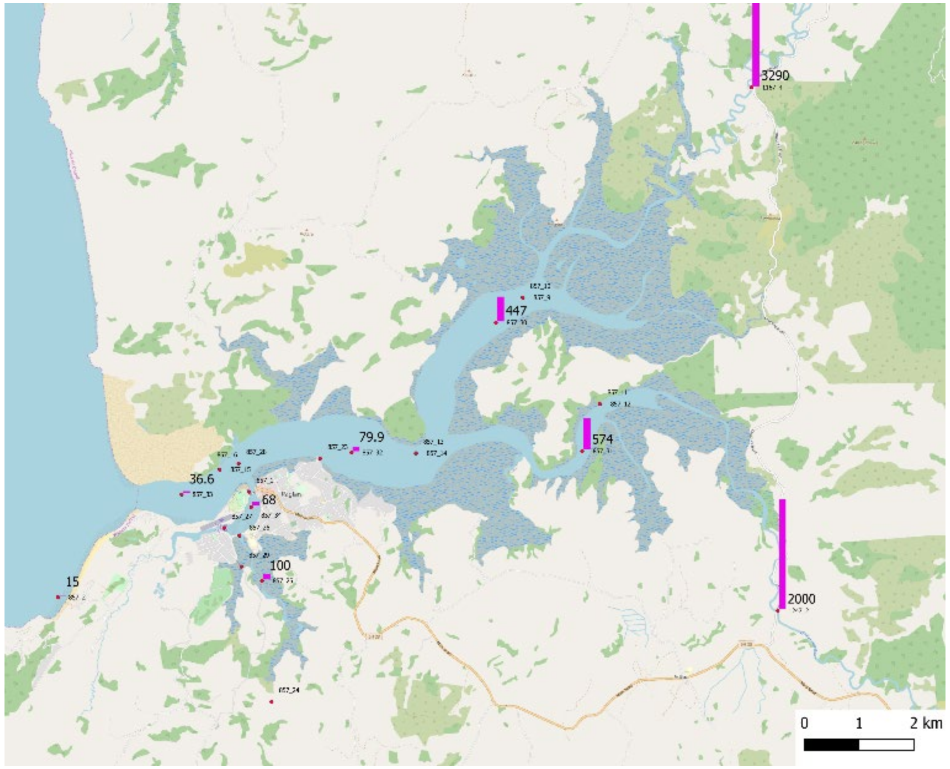
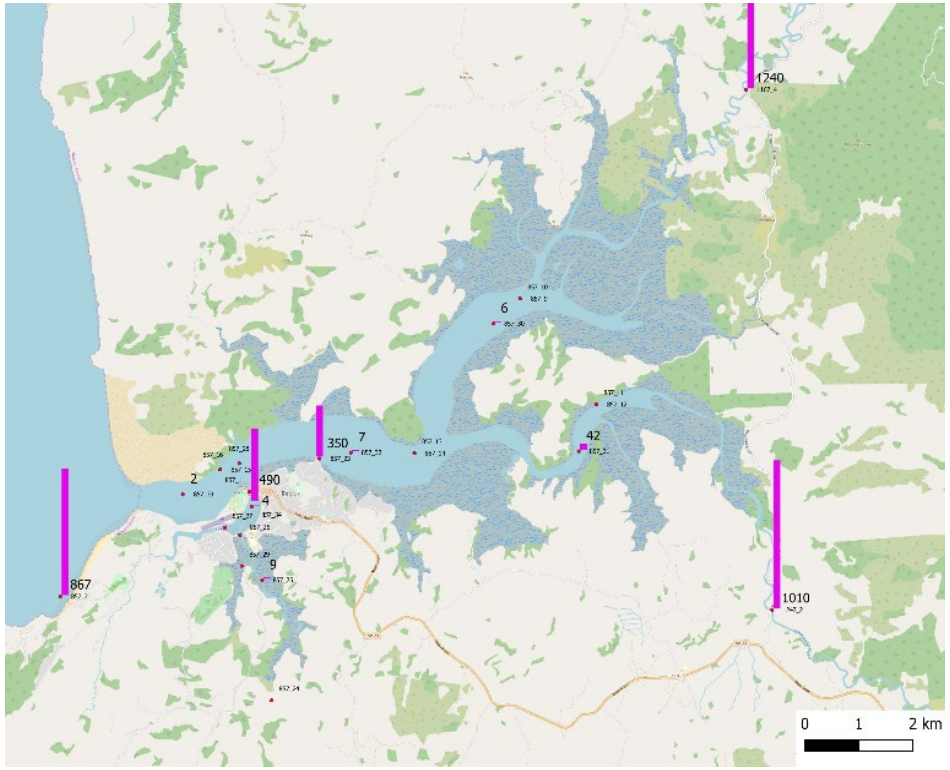



**Figure 3-6: Ninety-fifth percentile enterococci concentration calculated from data for January 2017- June 2019.** Bars are scaled according to the highest value in the data set. The 95th percentile value is shown as the larger, bold number. The smaller number is the WRC site code. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1.

2017




2016

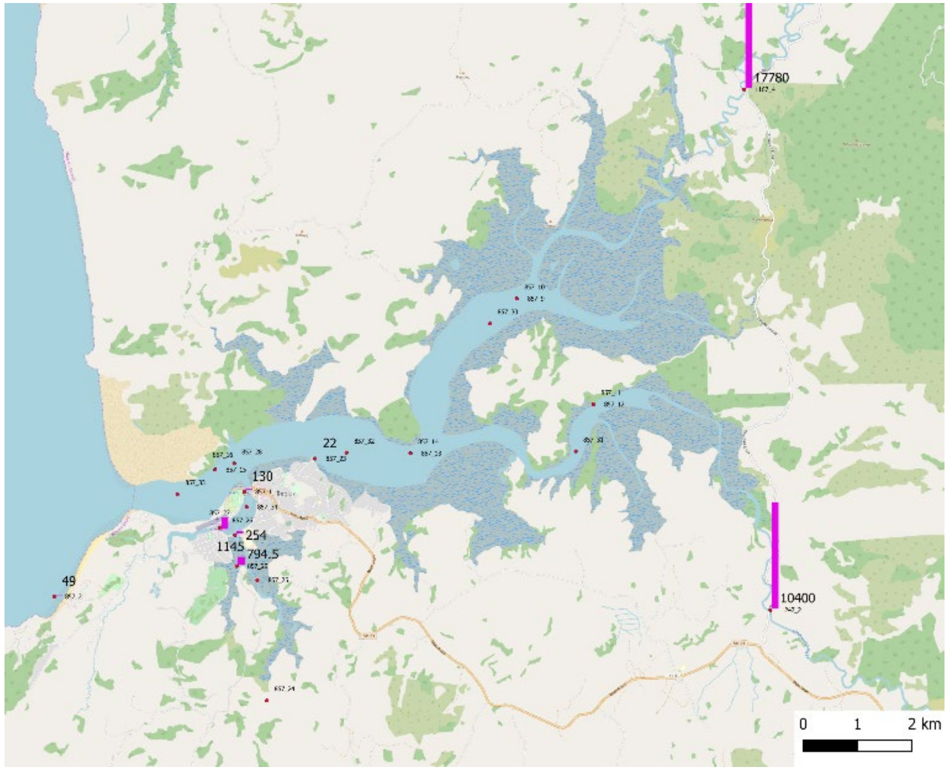



**Figure 3-6:** Ninety-fifth percentile enterococci concentration calculated from data for January 2017- June 2019. (Continued).

## 3.3 Recreational water quality

The New Zealand recreational water quality guidelines (MfE/MoH 2003) define protocols for sampling marine and freshwaters used for recreation. These include the FIB (enterococci are recommended for saline waters because of their greater persistence in sunlit coastal waters – contrary to freshwaters in which faecal coliform and *E. coli* are more persistent than enterococci (Nelson et al. 2018)), the sampling frequency (at least weekly), with preferably at least 20 samples collected annually over a recreation season (typically the period between November and March). Results from these samples are used to grade a beach in terms of suitability for use. Table H1 of the Guidelines provide thresholds for four categories of water, and these are related to illness risks. These thresholds are defined in terms of 95$^{th}$ percentile concentration values for each recreation season. The guidelines are summarised in Table 3-1 below:

**Table 3-1: Guideline values for microbiological marine water quality.** (MfE/MoH 2003).

| 95$^{th}$ percentile enterococci concentration (n/100 ml) | Basis of derivation | Estimated illness risk |
|---|---|---|
| ≤40 | Below "no observable adverse effects level" (NOAEL) | <1% gastrointestinal illness<br><0.3% acute febrile respiratory illness |
| 41-200 | Exceeds a "lowest observed adverse effects level" (LOAEL) | 1-5% gastrointestinal illness risk<br>0.3-1.9% acute febrile respiratory illness |
| 201-500 | Substantial elevation in probability of all adverse health outcomes for which dose-response data are available | 5-10% gastrointestinal illness<br>1.9-3.9% acute febrile respiratory illness |
| >500 | Significant risk of high levels of minor illness transmission | >10% gastrointestinal illness<br>>3.9% acute febrile respiratory illness |

Figure 3-7 summarises available data for two sites where samples have been collected for recreational grading – the Motor Camp North of Bridge (857_1) and Ngarunui Beach (857_2) sites. This figure shows the distribution of results at each site over each recreation season. The contamination levels tend to be more variable at the inner harbour site (857_1) than on the ocean beach (857_2).

Figure 3-8 summarises the recreation season 95$^{th}$ percentile values for both sites relative to the thresholds defined in Table 3-1. The 95$^{th}$ concentration values and numbers of samples per site are summarised in Table 3-2.

- In 12 of 16 recreation seasons where 12 or more data exist, 95$^{th}$ percentile concentrations were less than 40 enterococci /100 mL at both sites, indicating that gastrointestinal illness risk was "low" (likely to be less than 1%).
- In three of 16 recreation seasons, 95$^{th}$ percentile concentrations exceeded 200 enterococci/100 mL, indicating that gastrointestinal illness risk was likely to be "moderate" (greater than 5%).
- The high 95$^{th}$ percentile concentrations in the 2016/17 year result for samples collected on 7 April 2017. From the information available, we cannot identify the causes of highest microbial concentrations at either site, and we cannot attribute the elevated concentrations observed at both sites to a specific source.

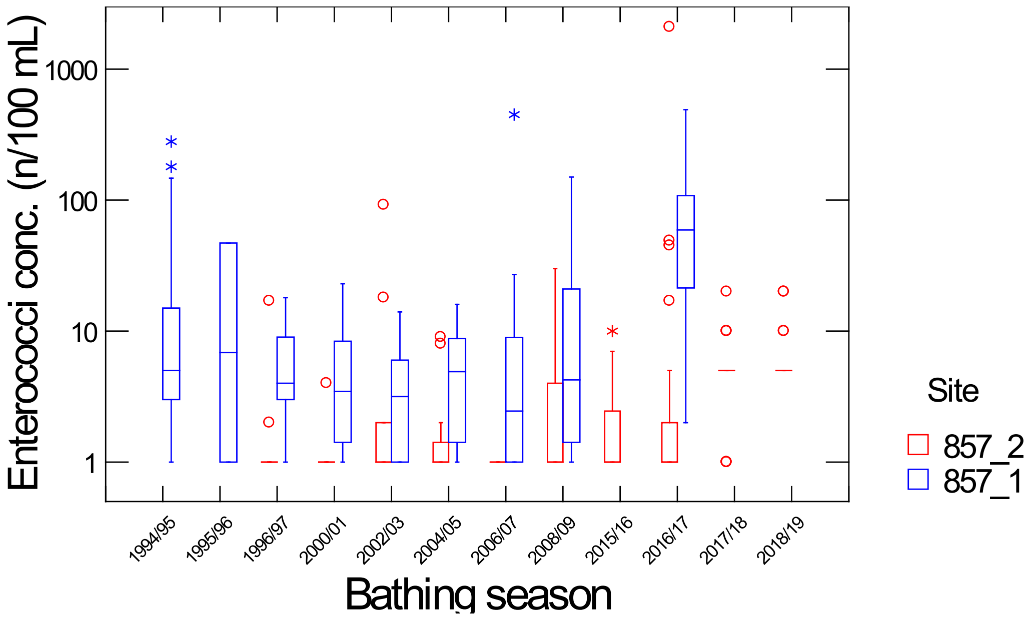



**Figure 3-7: Enterococci concentration data for the period 1994/95 – 2018/19.** An explanation of the symbols used in a box and whisker plot is included in Appendix A. The y-axis has $\log_{10}$ scale. Site 857_1 is at the "Motor Camp North of Bridge" site, and site 857_2 is the "Ngarunui Beach" site. Note the data are not continuous over the period 1994 – 2019. Absence of a box (e.g., 2017/18 and 2018/19) indicates the interquartile range was zero – with the exceptions of outliers, all other data is represented by the median line.

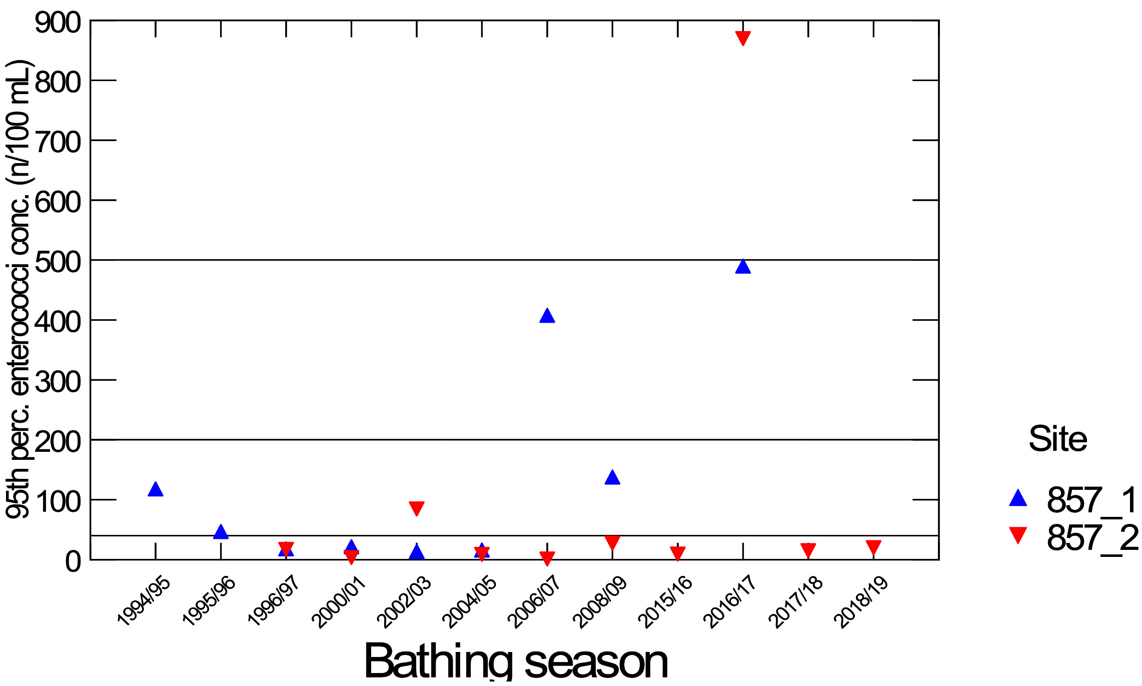



**Figure 3-8: Bathing season 95th percentile enterococci concentration data for the period 1994/95 – 2018/19.** Each symbol represents the seasonal value. Site 857_1 is at the "Motor Camp North of Bridge" site, and site 857_2 is the "Ngarunui Beach" site. Note the number of data per season vary over time (see Table 3-2). The broken horizontal lines are the threshold concentrations defined in Table 3-1 (40, 200, and 500 enterococci/100 mL).

**Table 3-2: 95th percentile enterococci concentration values and numbers of samples per bathing season .** (MfE/MoH 2003)(MfE/MoH 2003)(MfE/MoH 2003)(MfE/MoH 2003)(MfE/MoH 2003)(MfE/MoH 2003). Note the particularly high values at both sites in the 2016/17 recreational season (due to sampling results for the 7/4/16).

| Recreation season | Site 857_1 Motor Camp at Bridge | | Site 857_2 Ngarunui Beach | |
|---|---|---|---|---|
| | 95th percentile conc. (n/100 mL) | Count | 95th percentile conc. (n/100 mL) | Count |
| 1994/95 | 118 | 74 | | |
| 1995/96 | 47 | 3 | | |
| 1996/97 | 18 | 9 | 17 | 9 |
| 2000/01 | 22 | 12 | 4 | 12 |
| 2002/03 | 14 | 12 | 85 | 12 |
| 2004/05 | 16 | 12 | 9 | 12 |
| 2006/07 | 408 | 12 | 1 | 12 |
| 2008/09 | 138 | 12 | 27 | 12 |
| 2015/16 | | | 9 | 15 |
| 2016/17 | 490 | 8 | 869 | 22 |
| 2017/18 | | | 15 | 20 |
| 2018/19 | | | 20 | 20 |

During December 2016, samples were collected at hourly intervals from the “Campground at bridge” (857_1) site. These data are presented together with recreational bathing beach data in Figure 3-9. Data collected on 8/12/2016 are summarised in Figure 3-10. This figure indicates that as the tide receded (and water flowed from the embayment and catchment upstream of site 857_1), enterococci concentrations increased, suggesting that faecal contaminants originated from the catchment, not the harbour following contamination by wastewater from the wastewater discharge. This interpretation is supported by:

- typical consistent and effective wastewater treatment over this period, shown in Figure 3-2 and Figure 3-3, and
- consistently low enterococci concentrations observed at Ngarunui Beach (857_2) during December 2016, suggesting that the high performance of the wastewater treatment plant contributed to low enterococci concentrations observed at the beach.

Additional information (such as hydrodynamic modelling over this period or faecal source tracking) would be required for confirmation. This information is not available, so the conclusion that Harbour water quality is negligibly influenced by wastewater discharge remains speculative. However, the work of Greer et al. (2015) which included hydrodynamic modelling, suggests that the impact of the wastewater discharge is localise, and that catchment sources predominate.

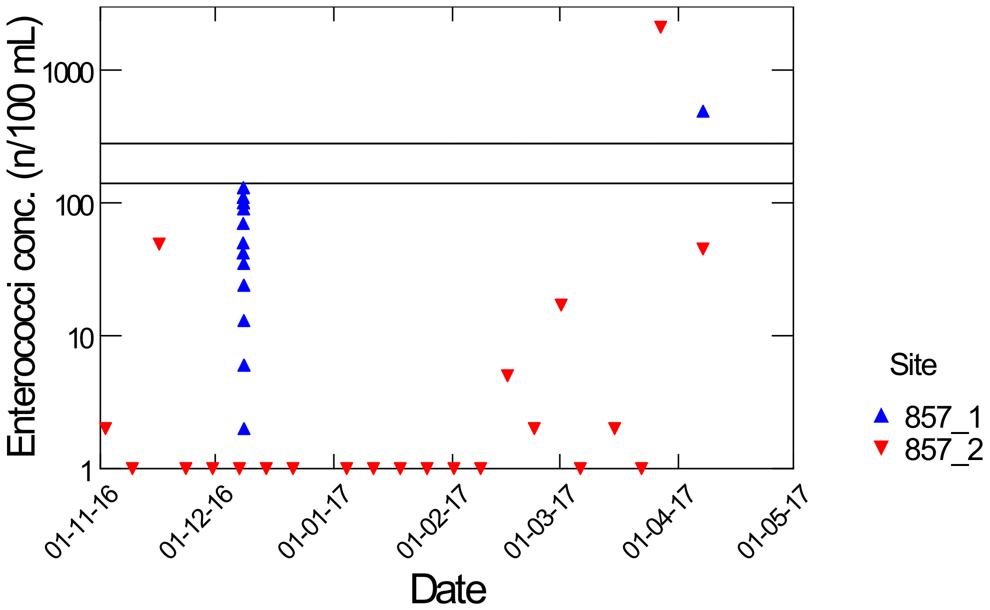



**Figure 3-9: Discrete enterococci concentration data for the 2016/17 bathing season.** Each symbol represents a discrete sample. Site 857_1 is at the "Motor Camp North of Bridge" site, and site 857_2 is the "Ngarunui Beach" site. Seven samples were collected from site 857_1 between 07:00 and 13:00 on 08/12/2016. The broken horizontal lines are the threshold concentrations defined in the MfE/MoH guidelines for single samples – alert= 140 enterococci/100 mL, and red = 280 enterococci/100 mL. Note y-axes have $\log_{10}$ scale.

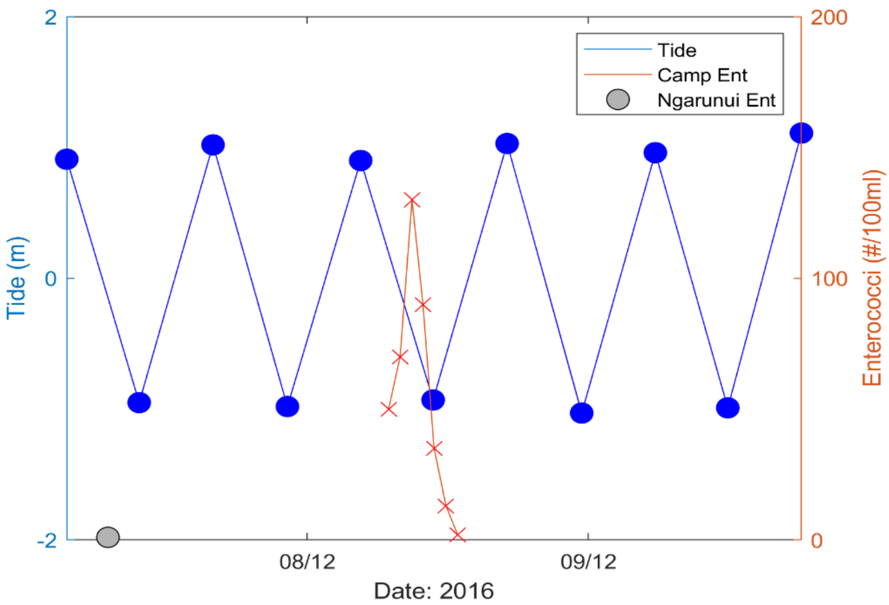



**Figure 3-10: Discrete enterococci concentration data collected on 8 December and 9 December 2016.** Seven samples were collected from Site 857_1 ("Motor Camp North of Bridge") between 07:00 and 13:00 on 08/12/2016, and one sample was collected from site 857_2 ("Ngarunui Beach") site. The blue line and symbols show the predicted tidal cycle.[3]

[3] https://www.niwa.co.nz/node/26820/results - data predicted for the Raglan Harbour Entrance site (37° 48' 11.0" S, 174° 50' 32.0" E).

Finally, during the 1994/95 recreation season, samples were collected at the Campground site (857_1) in replicate at 11:00 and 15:00 on 17 days (68 samples). A further 6 samples were collected at other times. The wastewater treatment did not include UV irradiation in 1994/95, so higher enterococci concentrations would be expected than currently. These data are summarised in Figure 3-11 as a time series plot (A), and as a box and whisker plot (B).

If elevated numbers of faecal contaminants from the WWTP were being transported into the inner harbour at times, it is likely that data collected at this site at this frequency would detect these incursions. The four-hour difference in sample time over the 17 sample dates would likely cover a range of tidal stages, and therefore discharge status. There was very little difference in the median value of the two data sets defined by sample time (data not shown), and the range of data were also almost identical.

Fewer than 15% of results were greater than 40 enterococci/100 mL, and the 95th percentile value of the full data set was 194 enterococci/100 mL (Table 3-2); the latter value indicates a gastrointestinal illness risk of 1-5% (Table 3-1). These results suggest that the plume of diluted wastewater had limited effect on the inner harbour even before UV treatment was installed.

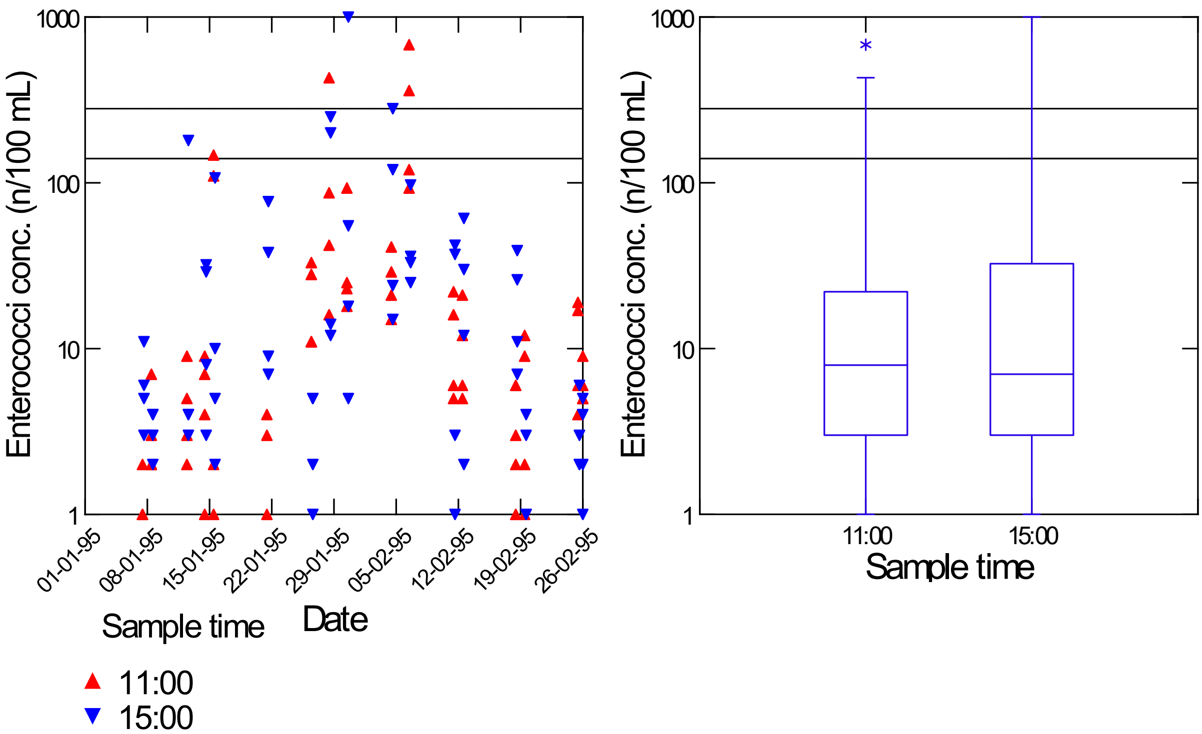



**Figure 3-11: Intensive enterococci monitoring conducted at the "Motor Camp North of Bridge" site during the 1994/95 recreation season.** Data are for before UV treatment was installed. A) Time series of 74 samples collected between 18 December 1994 and March 1995. B) Distribution of data by sample time. The broken horizontal lines are the threshold concentrations defined in the MfE/MoH guidelines for single samples – alert= 140 enterococci/100 mL, and red = 280 enterococci/100 mL. Note y-axes have $log_{10}$ scale.

# 4 Conclusions

Raglan Harbour is a relatively shallow harbour, with numerous embayments and two major tributary inflows. Review of available water quality data indicates:

### Wastewater treatment efficacy:

- Wastewater quality (2015-2018) is consistent and effective, with FIB concentrations typically less than 100 n/100 mL.
- The treatment process (which includes UV irradiation) consistently reduces FIB concentrations by approximately five-log orders.

### General harbour microbiological quality

- Microbiological water quality in the upper harbour is largely determined by catchment sources, particularly from the two largest tributaries.
- Concentrations of all FIB tend to consistently decrease from the upper harbour toward the mouth suggesting substantial dilution and attenuation of FIB from inflow toward the Harbour mouth.

### Recreational water quality

- More than 12 data exist for each of two recreational sites for 16 recreation seasons.
- In 12 of these 16 seasons, the gastrointestinal illness risk was likely to have been "low" (less than 1% - the "no observable adverse effect level), indicating high recreational water quality.
- In three of these 16 seasons, illness risks were "moderate" (likely to have exceeded 5%, with a greater than 10% illness risk likely in the 2016/17 recreational year).

Review of intensive water quality data available for the 1994/95 recreation season (before UV treatment commenced) suggests that the plume of diluted wastewater impacts negligibly on the harbour. This observation is consistent with predicted dispersion of wastewater undertaken by Greer et al. (2015), who suggested that the impact of the wastewater would be localised around the outfall, and the plume would be transported out of the harbour with the outgoing tide.

# 5 References


Greer, S.D., McIntosh, R., Harrison, S., Phillips, D., Mead, S. (2015) Understanding water quality in Raglan Harbour. *Australasian Coasts and Ports Conference 2015*, Auckland, 15-18 September, 2015.

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MfE/MoH (2003) Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas. *Report* No. MfE 474: 124.

Nelson, K.L., Boehm, A.B., Davies-Colley, R.J., Dodd, M.C., Kohn, T., Linden, K.G., Liu, Y., Maraccini, P.A., McNeill, K., Mitch, W.A., Nguyen, T.H., Parker, K.M., Rodriguez, R.A., Sassoubre, L.M., Silverman, A.I., Wigginton, K.R., Zepp, R.G. (2018) Sunlight-mediated inactivation of health-relevant microorganisms in water: a review of mechanisms and modeling approaches. *Environmental Science: Processes & Impacts*, 20(8): 1089-1122. 10.1039/C8EM00047F.

Oldman, J. (2019) Hydrodynamic modelling of Raglan wastewater discharge. Draft report prepared by DHI for Beca. *Report* No: 66.

Soller, J.A., Schoen, M.E., Bartrand, T., Ravenscroft, J.E., Ashbolt, N.J. (2010) Estimated human health risks from exposure to recreational waters impacted by human and non-human sources of faecal contamination. *Water Research*, 44(16): 4674-4691. http://dx.doi.org/10.1016/j.watres.2010.06.049.

# Appendix A Explanation of a box and whisker plot

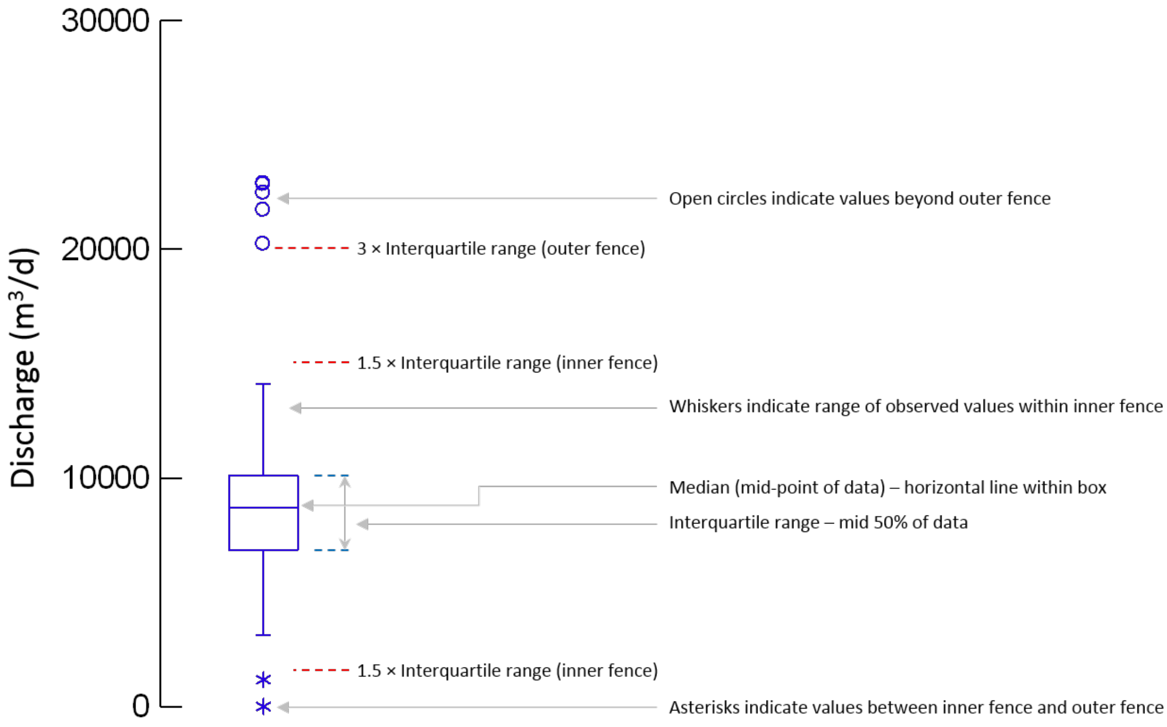


**Figure A-1: The symbology used in box and whisker plots generated by the Systat software package.**

# Appendix B Wastewater discharge characteristics – summary statistics

**Table B-1: Summary statistics for Raglan WWTP discharge.** Data for period June 2016 – April 2019.

| Statistic | Discharge (L/s) |
|---|---|
| N of Cases | 1058 |
| Minimum | 0 |
| Maximum | 27.4 |
| Median | 10.3 |
| Arithmetic Mean | 12.0 |
| Standard Deviation | 5.3 |
| Cleveland percentiles | |
| 1.00% | 4.8 |
| 5.00% | 6.4 |
| 10.00% | 7.0 |
| 20.00% | 8.0 |
| 25.00% | 8.4 |
| 30.00% | 8.7 |
| 40.00% | 9.5 |
| 50.00% | 10.3 |
| 60.00% | 11.5 |
| 70.00% | 13.3 |
| 75.00% | 14.2 |
| 80.00% | 15.4 |
| 90.00% | 19.9 |
| 95.00% | 24.2 |
| 99.00% | 27.1 |

**Table B-2: Summary statistics for enterococci concentration and flux Raglan WWTP discharge – outflow only.** Data for period January 2015 – February 2018.

| Statistic | Enterococci conc. (n/100 mL) | Enterococci flux (n/s) |
|---|---|---|
| N of Cases | 31 | 17 |
| Minimum | 0.5 | 30 |
| Maximum | 750000 | 79613042 |
| Arithmetic Mean | 24201.03 | 4683341 |
| Standard Deviation | 134702.59 | 19308942 |
| Cleveland percentiles | | |
| 1.00% | 0.5 | 30 |
| 5.00% | 0.5 | 32 |
| 10.00% | 0.5 | 37 |
| 20.00% | 0.5 | 40 |
| 25.00% | 0.5 | 41 |
| 30.00% | 0.5 | 44 |
| 40.00% | 0.95 | 82 |
| 50.00% | 2 | 176 |
| 60.00% | 5 | 390 |
| 70.00% | 5 | 454 |
| 75.00% | 5 | 471 |
| 80.00% | 5 | 475 |
| 90.00% | 36.4 | 530 |
| 95.00% | 87.5 | 51748667 |
| 99.00% | 750000 | 79613042 |

**Table B-3: Summary statistics for Raglan WWTP faecal coliform concentrations – inflow and outflow.** Data for period January 2015 – February 2018.

| Statistic | Inflow | Outflow |
|---|---|---|
| | Faecal coliform conc. (n/100 mL) | Faecal coliform conc. (n/100 mL) |
| N of Cases | 51 | 47 |
| Minimum | 600000 | 0.5 |
| Maximum | 71000000 | 110 |
| Arithmetic Mean | 12158824 | 10.2 |
| Standard Deviation | 11965955 | 21.8 |
| Cleveland percentiles | | |
| 1.00% | 604000 | 0.5 |
| 5.00% | 1310000 | 0.5 |
| 10.00% | 1600000 | 0.5 |
| 20.00% | 3780000 | 0.5 |
| 25.00% | 4800000 | 0.5 |
| 30.00% | 5000000 | 0.8 |
| 40.00% | 6000000 | 2 |
| 50.00% | 9000000 | 3 |
| 60.00% | 11100000 | 5 |
| 70.00% | 16200000 | 5.4 |
| 75.00% | 17750000 | 7 |
| 80.00% | 20300000 | 10 |
| 90.00% | 26000000 | 24.8 |
| 95.00% | 28900000 | 73.2 |
| 99.00% | 70600000 | 110 |

**Table B-4: Summary statistics for Raglan WWTP faecal coliform flux – inflow and outflow.** Data for period January 2015 – February 2018.

| Statistic | Inflow | Outflow |
|---|---|---|
| | FC flux (n/s) | FC flux (n/s) |
| N of Cases | 25 | 21 |
| Minimum | 127888758 | 30 |
| Maximum | 2880000000 | 1661 |
| Arithmetic Mean | 1260000000 | 561 |
| Standard Deviation | 876289692 | 504 |
| Cleveland percentiles | | |
| 1.00% | 127888758 | 30 |
| 5.00% | 170749122 | 39 |
| 10.00% | 198754537 | 61 |
| 20.00% | 301254941 | 115 |
| 25.00% | 490720089 | 159 |
| 30.00% | 766453877 | 176 |
| 40.00% | 929836632 | 246 |
| 50.00% | 1110000000 | 458 |
| 60.00% | 1360000000 | 524 |
| 70.00% | 1640000000 | 853 |
| 75.00% | 1900000000 | 928 |
| 80.00% | 2120000000 | 1026 |
| 90.00% | 2750000000 | 1351 |
| 95.00% | 2790000000 | 1622 |
| 99.00% | 2880000000 | 1661 |

# Appendix C Longitudinal harbour *E. coli* and faecal coliform concentration data

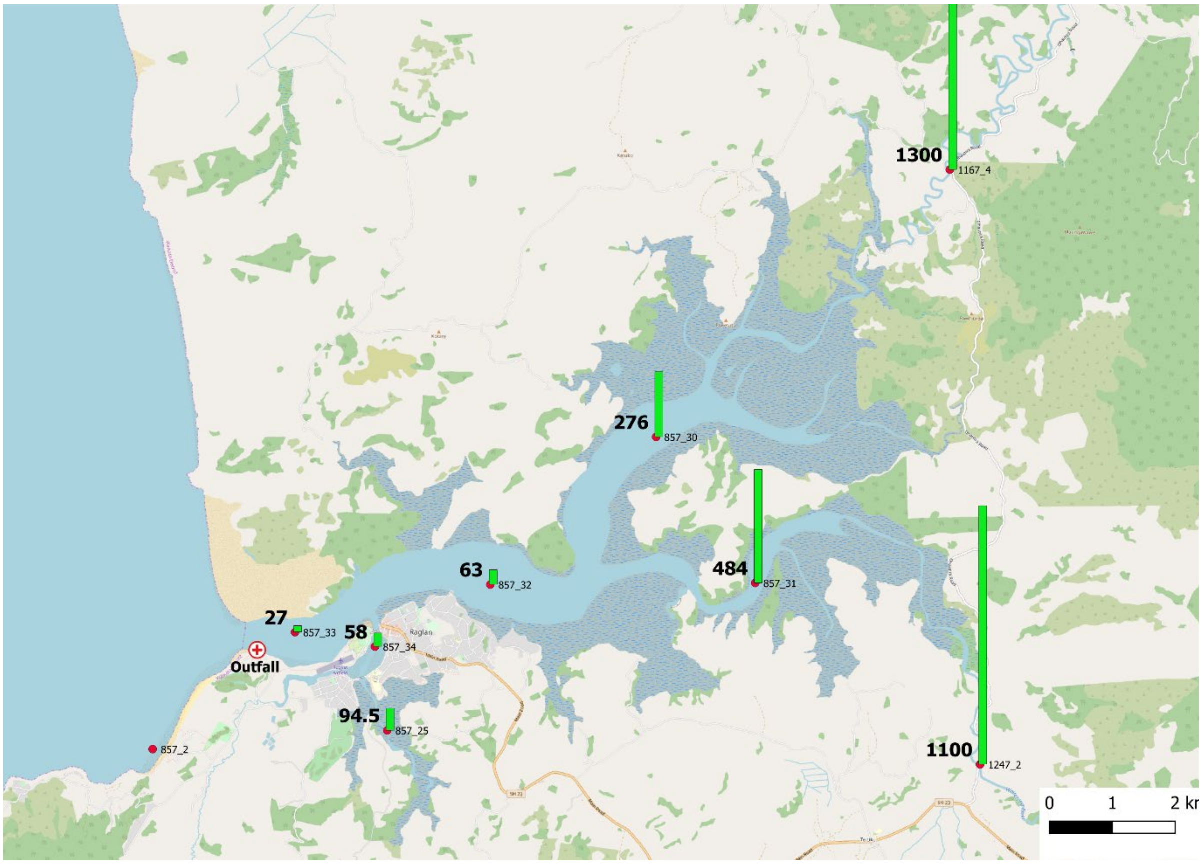


**Figure C-1: Ninety-fifth percentile *E. coli* concentrations calculated from data for January 2017 – June 2019.** Bars are scaled according to the highest value in the data set. The 95th percentile value is shown as the larger, bold number. The smaller number is the WRC site code. Site details are summarised in Table 2-1.

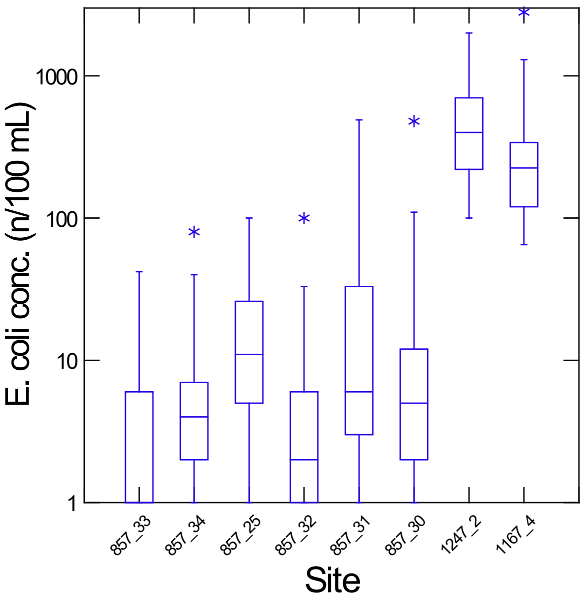


**Figure C-2:** ***E. coli* concentration data for the period January 2017 – June 2019.** An explanation of the symbols used in a box and whisker plot is included in Appendix A. The y-axis has log10 scale. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1. Site 1247_2 and 1167_4 are the major freshwater inflows to the harbour.

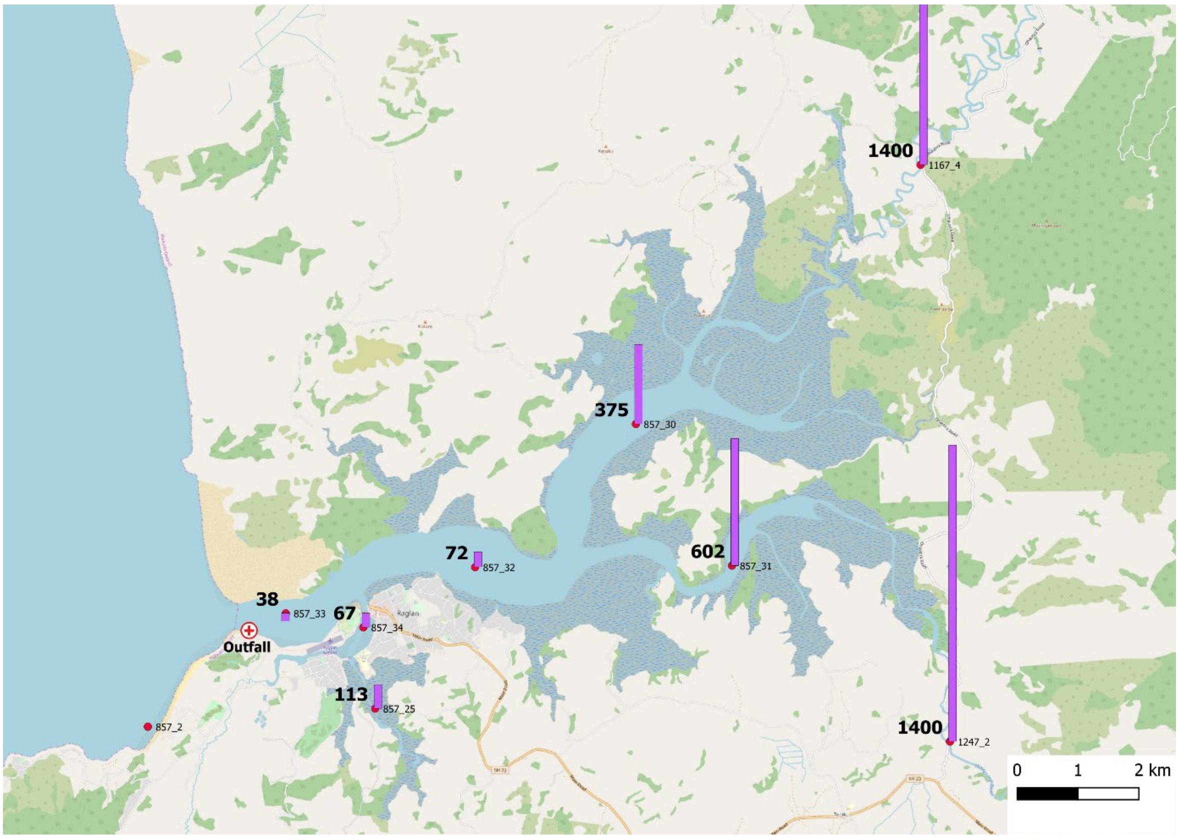


**Figure C-3: Ninety-fifth percentile faecal coliform concentration calculated from data for January 2017-June 2019.** Bars are scaled according to the highest value in the data set. The 95th percentile value is shown as the larger, bold number. The smaller number is the WRC site code. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1.

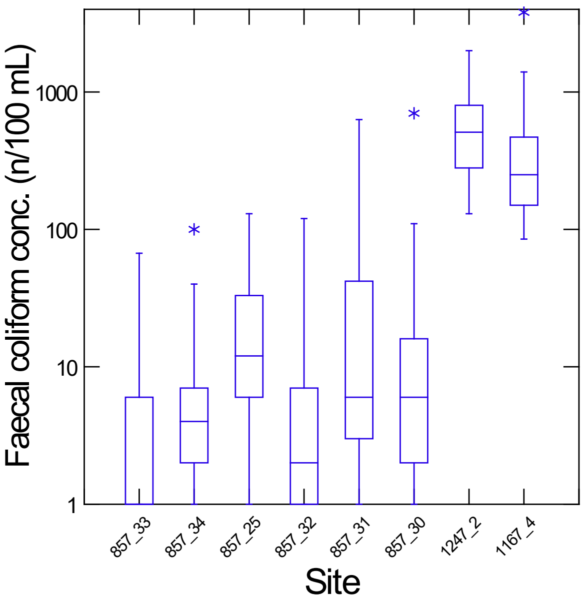


**Figure C-4: Faecal coliform concentration data for the period January 2017 - June 2019.** An explanation of the symbols used in a box and whisker plot is included in Appendix A. The y-axis has log10 scale. Site details are summarised in Table 2-1 and site locations are shown in Figure 2-1.

# Appendix D Time-series plots of faecal contaminants in the Raglan Harbour catchment and harbour

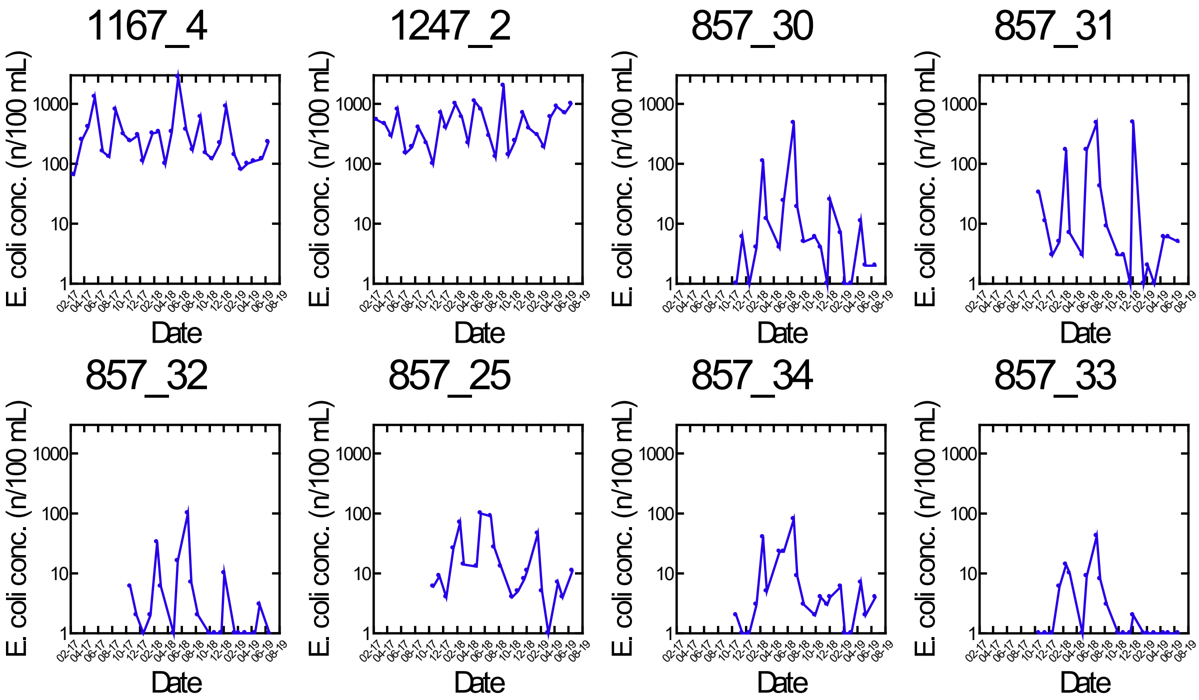


**Figure D-1: Time series concentration of *E. coli* concentrations derived from selected sites in the Raglan Harbour catchment and across Raglan Harbour.** The site codes and descriptions are listed in Table 2-1, and site locations are shown in Figure 1-1 and Figure C-1.

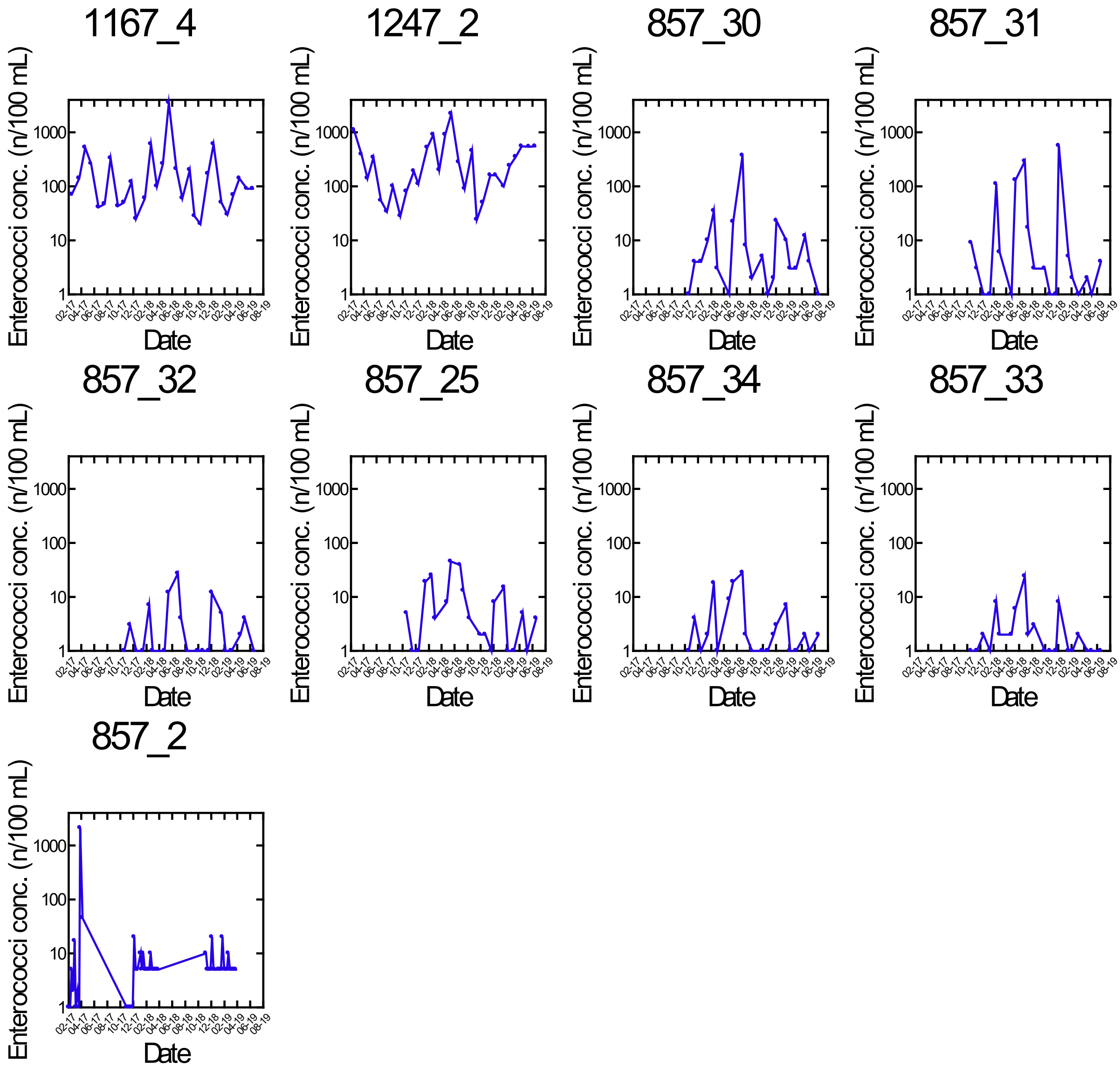


**Figure D-2: Time series concentration of enterococci concentrations derived from selected sites in the Raglan Harbour catchment and across Raglan Harbour.** The site codes and descriptions are listed in Table 2-1, and site locations are shown in Figure 1-1 and Figure 3-4.

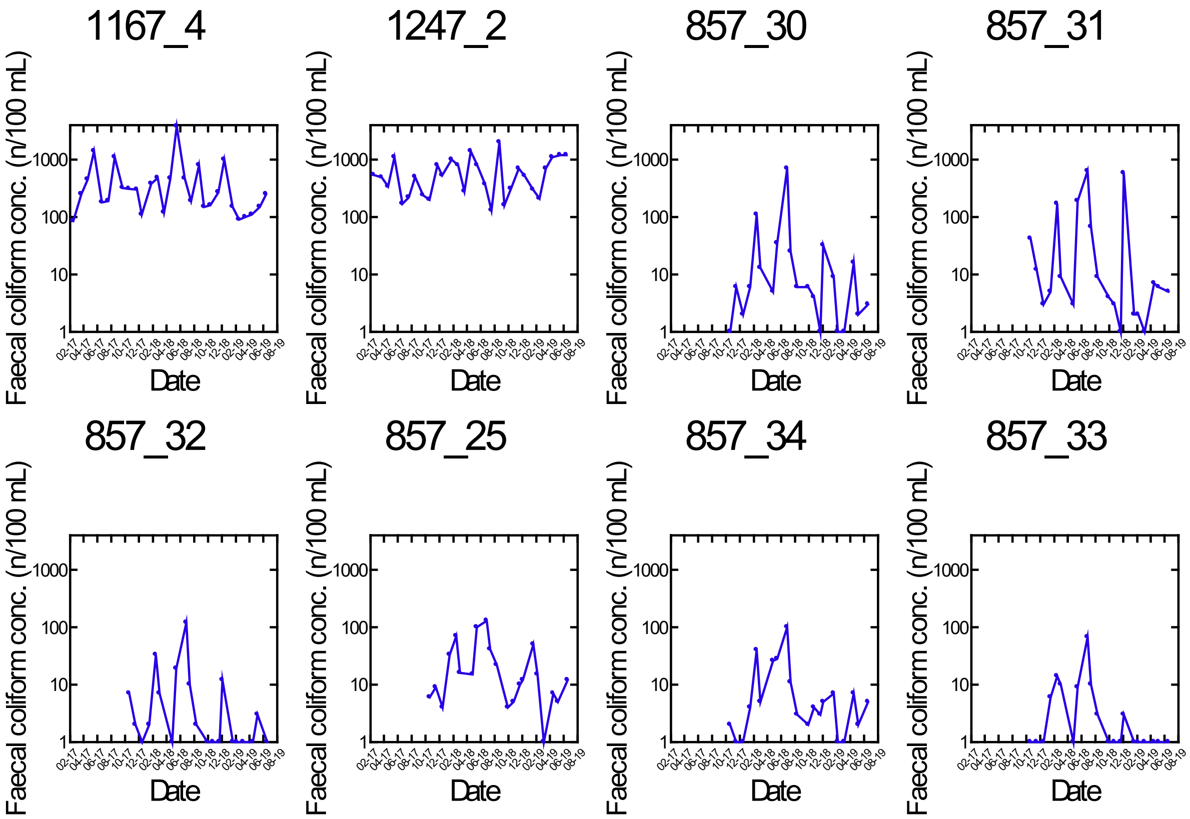


**Figure D-3: Time series concentration of faecal coliform concentrations derived from selected sites in the Raglan Harbour catchment and across Raglan Harbour.** The site codes and descriptions are listed in Table 2-1, and site locations are shown in Figure 1-1 and Figure C-3.

# Appendix E Summary statistics – microbiological water quality variables, 2017-2019 inclusive

| Summary statistics, Site code = 1167_4 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 30 | 30 | 30 |
| Minimum | 65 | 20 | 85 |
| Maximum | 2800 | 3500 | 3800 |
| Arithmetic Mean | 379.17 | 265.43 | 464.5 |
| Standard Deviation | 534.28 | 632.33 | 708.43 |
| Cleveland percentiles | | | |
| 1.00% | 65 | 20 | 85 |
| 5.00% | 80 | 25 | 90 |
| 10.00% | 95 | 29 | 100 |
| 20.00% | 110 | 42 | 115 |
| 25.00% | 120 | 47 | 150 |
| 30.00% | 125 | 50 | 150 |
| 40.00% | 155 | 64.5 | 185 |
| 50.00% | 225 | 90 | 250 |
| 60.00% | 275 | 130 | 305 |
| 70.00% | 330 | 185 | 415 |
| 75.00% | 340 | 210 | 470 |
| 80.00% | 390 | 260 | 475 |
| 90.00% | 850 | 560 | 1050 |
| 95.00% | 1300 | 600 | 1400 |
| 99.00% | 2800 | 3500 | 3800 |

| Summary statistics Site code = 1247_2 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 30 | 30 | 30 |
| Minimum | 100 | 24 | 130 |
| Maximum | 2000 | 2200 | 2000 |
| Arithmetic Mean | 527 | 370.33 | 622.33 |
| Standard Deviation | 404.92 | 444.55 | 446.52 |
| Cleveland percentiles | | | |
| 1.00% | 100 | 24 | 130 |
| 5.00% | 130 | 28 | 160 |
| 10.00% | 145 | 42 | 185 |
| 20.00% | 205 | 85 | 230 |
| 25.00% | 220 | 100 | 280 |
| 30.00% | 250 | 105 | 305 |
| 40.00% | 295 | 160 | 365 |
| 50.00% | 400 | 220 | 510 |
| 60.00% | 570 | 310 | 620 |
| 70.00% | 700 | 420 | 800 |
| 75.00% | 700 | 520 | 800 |
| 80.00% | 800 | 545 | 1050 |
| 90.00% | 1000 | 900 | 1200 |
| 95.00% | 1100 | 1100 | 1400 |
| 99.00% | 2000 | 2200 | 2000 |

| Summary statistics Site code = 857_25 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 100 | 45 | 130 |
| Arithmetic Mean | 22.57 | 9.71 | 27 |
| Standard Deviation | 29.09 | 12.62 | 34.34 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 2.65 | 1 | 2.65 |
| 10.00% | 4 | 1 | 4 |
| 20.00% | 4.7 | 1 | 5 |
| 25.00% | 5 | 1 | 5.75 |
| 30.00% | 5.8 | 1.8 | 6.8 |
| 40.00% | 7.9 | 3.8 | 9.9 |
| 50.00% | 11 | 4 | 12 |
| 60.00% | 13 | 5.3 | 15.1 |
| 70.00% | 16.4 | 9 | 24.2 |
| 75.00% | 26.25 | 13.5 | 35 |
| 80.00% | 32.7 | 16.2 | 43.7 |
| 90.00% | 78 | 30.6 | 82 |
| 95.00% | 94.5 | 41.7 | 113.5 |
| 99.00% | 100 | 45 | 130 |

| Summary statistics Site code = 857_30 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 480 | 370 | 700 |
| Arithmetic Mean | 34.57 | 24.95 | 46.86 |
| Standard Deviation | 104.75 | 79.56 | 151.59 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 1 | 1 | 1 |
| 10.00% | 1 | 1 | 1 |
| 20.00% | 1 | 1.7 | 1.7 |
| 25.00% | 1.75 | 2 | 2 |
| 30.00% | 2 | 2.8 | 2.8 |
| 40.00% | 4 | 3 | 4.9 |
| 50.00% | 5 | 4 | 6 |
| 60.00% | 6.1 | 5.3 | 6.3 |
| 70.00% | 11.2 | 10 | 13.6 |
| 75.00% | 13.75 | 10.5 | 18.25 |
| 80.00% | 20.5 | 15 | 27.1 |
| 90.00% | 59 | 27.8 | 65 |
| 95.00% | 276.5 | 185.75 | 375.5 |
| 99.00% | 480 | 370 | 700 |

| Summary statistics Site code = 857_31 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 490 | 560 | 630 |
| Arithmetic Mean | 69.1 | 54.81 | 83.38 |
| Standard Deviation | 146.85 | 134.93 | 181.53 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 1 | 1 | 1 |
| 10.00% | 1 | 1 | 1.6 |
| 20.00% | 2.7 | 1 | 2.7 |
| 25.00% | 3 | 1 | 3 |
| 30.00% | 3 | 1 | 3 |
| 40.00% | 4.8 | 2 | 4.9 |
| 50.00% | 6 | 3 | 6 |
| 60.00% | 7.2 | 4.1 | 9 |
| 70.00% | 15.4 | 6.6 | 18 |
| 75.00% | 35.25 | 11 | 48.25 |
| 80.00% | 80.4 | 44.9 | 97.9 |
| 90.00% | 294 | 194 | 346 |
| 95.00% | 484.5 | 411.5 | 602.5 |
| 99.00% | 490 | 560 | 630 |

| Summary statistics Site code = 857_32 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 100 | 27 | 120 |
| Arithmetic Mean | 9.38 | 4.19 | 10.81 |
| Standard Deviation | 22.08 | 6.24 | 26.24 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 1 | 1 | 1 |
| 10.00% | 1 | 1 | 1 |
| 20.00% | 1 | 1 | 1 |
| 25.00% | 1 | 1 | 1 |
| 30.00% | 1 | 1 | 1 |
| 40.00% | 1 | 1 | 1 |
| 50.00% | 2 | 1 | 2 |
| 60.00% | 2.1 | 2.1 | 2.1 |
| 70.00% | 6 | 4 | 7 |
| 75.00% | 6.25 | 4.25 | 7.75 |
| 80.00% | 7.9 | 5.6 | 10.6 |
| 90.00% | 22.8 | 12 | 24.6 |
| 95.00% | 63.15 | 18.75 | 72.15 |
| 99.00% | 100 | 27 | 120 |

| Summary statistics<br>Site code = 857_33 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 42 | 24 | 67 |
| Arithmetic Mean | 5.1 | 3.33 | 6.43 |
| Standard Deviation | 9.28 | 5.23 | 14.43 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 1 | 1 | 1 |
| 10.00% | 1 | 1 | 1 |
| 20.00% | 1 | 1 | 1 |
| 25.00% | 1 | 1 | 1 |
| 30.00% | 1 | 1 | 1 |
| 40.00% | 1 | 1 | 1 |
| 50.00% | 1 | 1 | 1 |
| 60.00% | 1.1 | 2 | 1.2 |
| 70.00% | 3.6 | 2 | 3.6 |
| 75.00% | 6.5 | 2.25 | 6.75 |
| 80.00% | 8.3 | 3.9 | 9.3 |
| 90.00% | 11.6 | 8 | 11.6 |
| 95.00% | 26.6 | 15.2 | 37.85 |
| 99.00% | 42 | 24 | 67 |

| Summary statistics<br>Site code = 857_34 | Microbiological water quality variable | | |
|---|---|---|---|
| | *E. coli*_507 | Enterococci_519 | Faecal coliform_510 |
| N of Cases | 21 | 21 | 21 |
| Minimum | 1 | 1 | 1 |
| Maximum | 80 | 28 | 100 |
| Arithmetic Mean | 10.67 | 5.1 | 12.29 |
| Standard Deviation | 18.64 | 7.44 | 22.64 |
| Cleveland percentiles | | | |
| 1.00% | 1 | 1 | 1 |
| 5.00% | 1 | 1 | 1 |
| 10.00% | 1 | 1 | 1 |
| 20.00% | 1.7 | 1 | 1.7 |
| 25.00% | 2 | 1 | 2 |
| 30.00% | 2 | 1 | 2 |
| 40.00% | 3 | 1 | 3 |
| 50.00% | 4 | 2 | 4 |
| 60.00% | 4.1 | 2 | 5 |
| 70.00% | 6.2 | 3.2 | 7 |
| 75.00% | 7.5 | 4.75 | 8 |
| 80.00% | 13.2 | 7.6 | 15.5 |
| 90.00% | 29.8 | 18.4 | 32.8 |
| 95.00% | 58 | 23.05 | 67 |
| 99.00% | 80 | 28 | 100 |

| Summary statistics<br>Site code = 857_2 | Microbiological water quality variable | |
|---|---|---|
| | Enterococci_1287 | Enterococci_519 |
| N of Cases | 36 | 19 |
| Minimum | 5 | 1 |
| Maximum | 20 | 2100 |
| Arithmetic Mean | 6.94 | 114.95 |
| Standard Deviation | 4.36 | 480.82 |
| Cleveland percentiles | | |
| 1.00% | 5 | 1 |
| 5.00% | 5 | 1 |
| 10.00% | 5 | 1 |
| 20.00% | 5 | 1 |
| 25.00% | 5 | 1 |
| 30.00% | 5 | 1 |
| 40.00% | 5 | 1 |
| 50.00% | 5 | 1 |
| 60.00% | 5 | 1 |
| 70.00% | 5 | 1.8 |
| 75.00% | 5 | 2 |
| 80.00% | 10 | 4.1 |
| 90.00% | 10 | 33.8 |
| 95.00% | 20 | 1175.25 |
| 99.00% | 20 | 2100 |

# B

Appendix B – DHI Discharge Assessment

# Raglan Wastewater Treatment Plant Discharge Assessment

BECA

Report 44801398

November 2019

The expert in WATER ENVIRONMENTS

# Raglan Wastewater Treatment Plant Discharge Assessment

| | |
|---|---|
| Prepared for | BECA |
| Represented by | Mr Garrett Hall |

*View of Raglan Harbour to the north-east of the existing outfall location.*

| | |
|---|---|
| Project manager | John Oldman |

| | |
|---|---|
| Project number | 44801398 |
| Approval date | 1/11/2019 |
| Revision | Final |

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# Contents

# 1 Introduction

This report provides details of the calibration of a hydrodynamic model which has been used to assess the discharge of treated wastewater to Raglan Harbour. The discharge is via an outfall near the entrance to the harbour and is timed to occur mostly on the outgoing tide.

The current consent conditions allow the discharge to commence half an hour before high water for a period up to 5.5 hours after high water.

An analysis of the current discharge regime indicates that a mean daily discharge of 1175 $m^3$/day occurs and that the median discharge duration is 2 hours and 15 minutes (averaging 0.058 $m^3$/s) and the median start time of the discharge is 15 minutes after high water.

Under future population growth estimates (to 2055), the mean daily discharge may increase to 2335 $m^3$/day.

Section 2 of this report provides an overview of previous studies carried out in the harbour. Section 3 of the report provides an overview of the data used for setting up and calibrating the hydrodynamic model. Section 4 provides details of the calibration of the model against the field data. Section 5 provides results of the near field and far field modelling that has been carried out to assess the dynamics of the treated wastewater plume under a range of conditions and the level of dilution achieved at key sites both within the harbour and at a number of locations offshore of the harbour. Section 6 provides a summary of the findings of the study and an overview of the potential for offsetting the effects of a future discharge in terms of water quality and public health risk.

# 2 Overview of previous modelling and field studies

An upgrade of the wastewater treatment plant was considered in 1995 (Gibbs and Watson 1996) which included dye tracking (Appendix A), a current meter deployment at the existing discharge location, and a hydrographic survey of the entrance of the harbour.

This work included an assessment of the potential for near-field dilution for a number of alternative outfall locations and diffuser design. The main conclusions from this work was that there is potential for a relatively high degree of near-field mixing due to the combination of strong tidal currents and water depths. Assuming an outfall was fitted with a diffuser, the range of initial dilutions achieved for a projected peak summer flow of 0.15 $m^3$/s was estimated to be between 155 and 710 with highest initial dilution occurring during mid-ebb tide. It was estimate that moving the outfall to deeper water could provide nearly twice the initial dilution as a near-shore discharge. It was estimated that subsequent dilutions that occur 1.5 to 4.5 hours after high tide would produce total dilutions in excess of 40,000 before the wastewater reaches the open sea.

None of the planned upgrades to the outfall were carried out.

More recently, modelling has been carried out to quantify the relative role of catchment derived contaminants and those discharged from the wastewater treatment plant (Greer et al. 2015) and how well flushed the harbour is (Greer et al.

2016). The main conclusions from these studies are that the effect of the treated wastewater discharge is small relative to the catchment derived inputs and that there is a strong gradient in harbour residence time with longest residence times (> 40 days) in the upper parts of the harbour (closest to the major freshwater sources) and much lower residence (~20 days) near the mouth of the harbour. This means that other freshwater sources remain in the harbour for much longer periods of time compared to the treated wastewater discharge which is relatively quickly flushed from the harbour.

A number of studies have been carried out which focus on the role of catchment derived sediments and sediment dynamics in the harbour (Swales et al. 2005, Harrison 2015, Hunt, 2016). These references provide useful background information on the dynamics of the harbour as a whole.

# 3 Model Input Data and Grid Development

## 3.1 Bathymetry data

Bathymetry for the harbour was used from a variety of sources including LiDAR (from the Waikato Regional Council dataset - 2010/2011), bathymetry surveys (Harrison, 2015 and Greer et al. 2015) and the navigation chart data (LINZ chart NZ4421).

All data was converted to a common vertical datum of Mean Sea Level (as defined in Figure 3-1).

The extent and detail of bathymetry grid used for this study are shown in Figures 3-2 and 3-3 respectively.

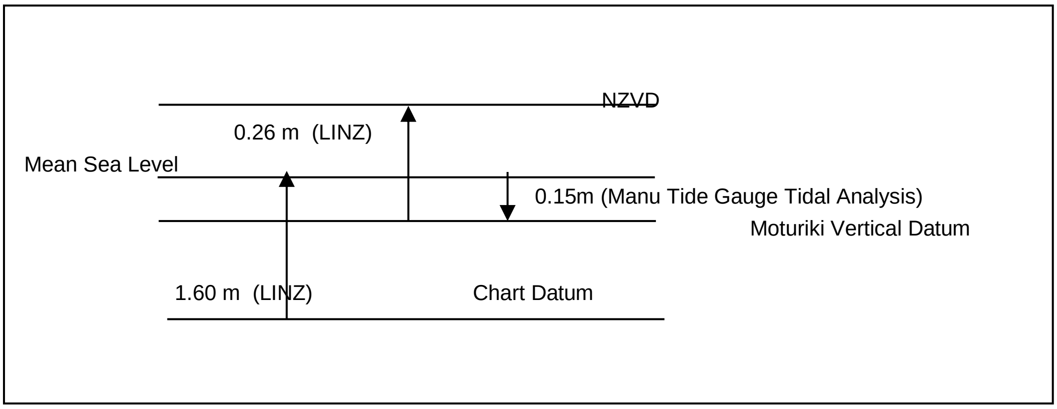


Figure 3-1. Vertical datums used in the various bathymetry datasets.

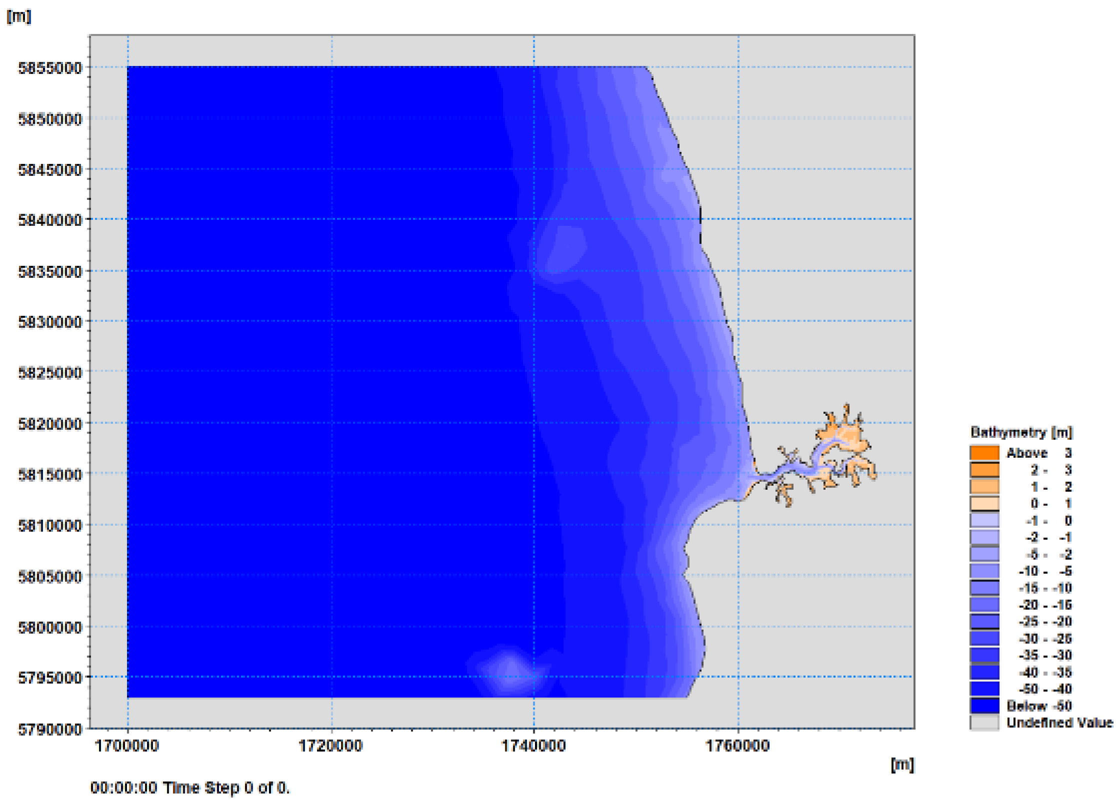


Figure 3-2. Extent of bathymetric grid.

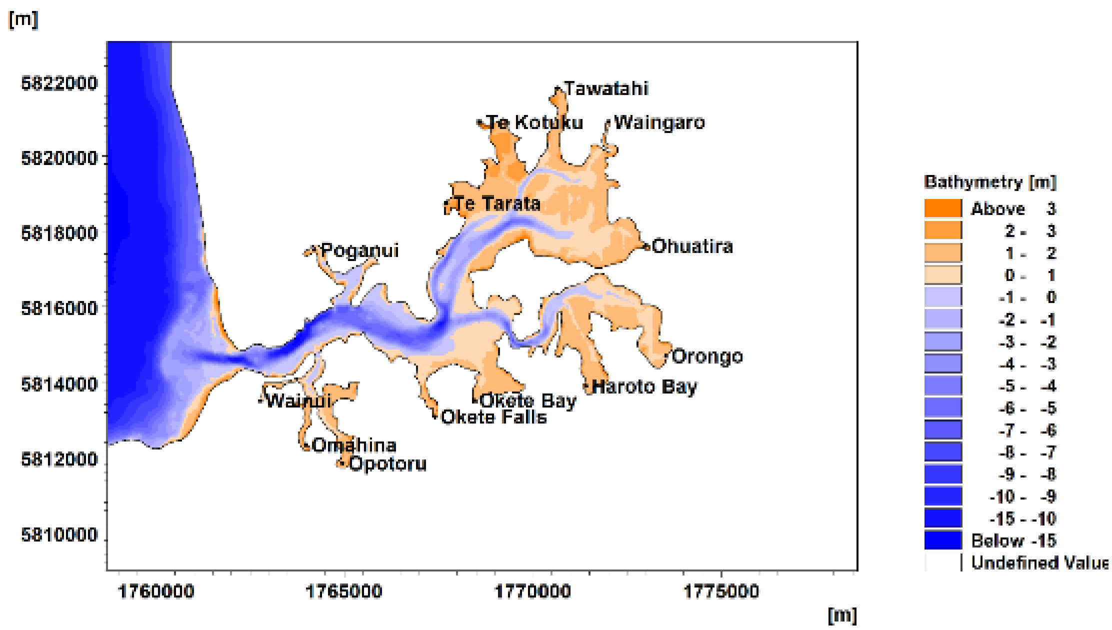


Figure 3-3. Detailed bathymetry within Raglan harbour and the area immediately offshore of the harbour. Labels indicate the freshwater sources included in the model (Section 3.2).

## 3.2 Freshwater inflows

Freshwater inflows from the Waingaro catchment have been collected by the Waikato Regional Council since 2003.

A summary of that data is provided Figure 3-4 and shows the strong seasonality of flows and the inter-annual variability. It can be seen that 2018 is relatively typical in terms of the total annual inflows but has higher than average winter inflows and lower spring inflows. Three relatively large rainfall events in December 2018 resulted in the mean monthly inflow for December 2018 being over two times the average December inflow.

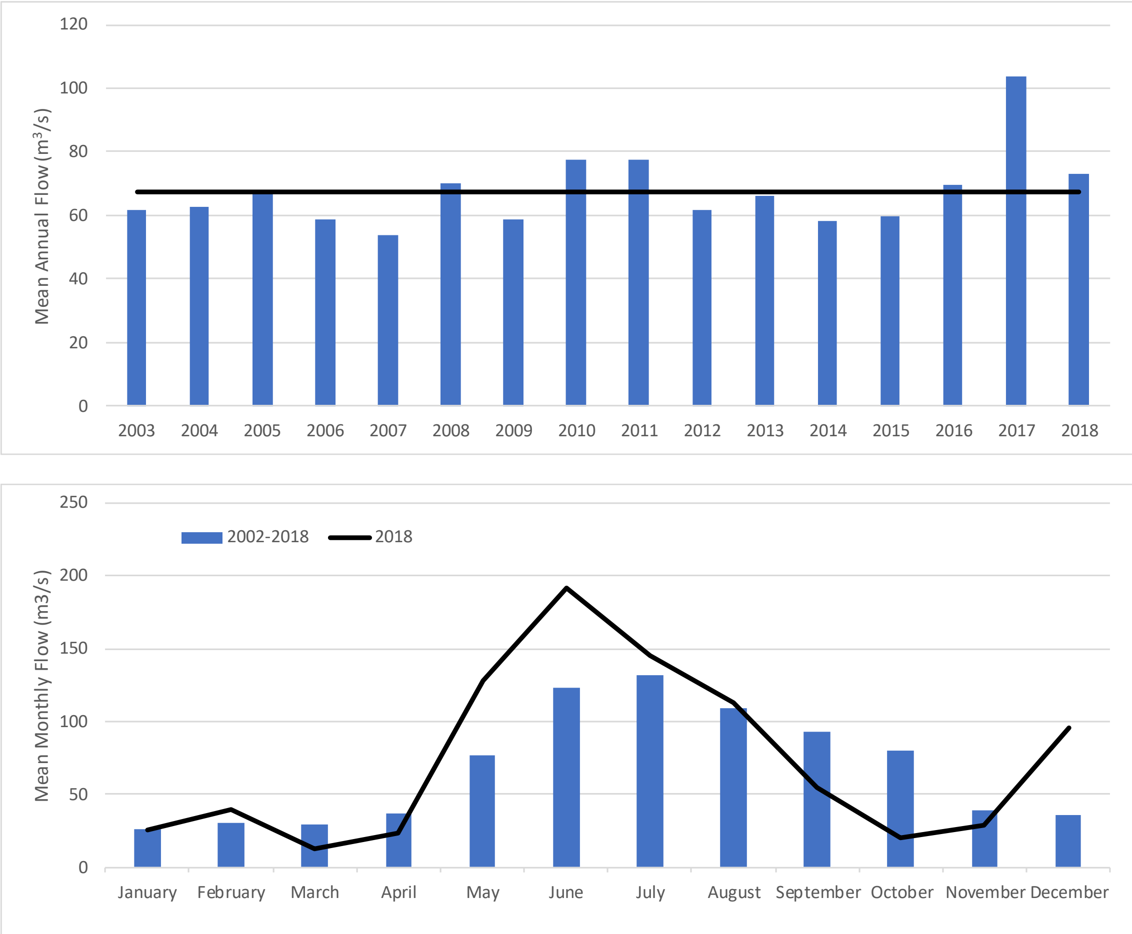


Figure 3-4. Summary of Waingaro river flows 2003-2018. Bar shows the mean annual flow and solid line shows data from 2018 (top panel). Bottom panel shows the mean monthly flow (bar) and solid line shows data from 2018.

## 3.3 Wind and atmospheric pressure data

Regional scale wind and atmospheric data was obtained from the National Centers for Environmental Prediction Climate Forecast System (Saha et al. 2011).

Previous work carried out by DHI (DHI, 2016 and DHI, 2019) has shown that the National Centers for Environmental Prediction Climate Forecast System data provides good estimates of observed winds.

This spatially varying wind and atmospheric pressure data provides robust inputs to regional scale hydrodynamics models and provides the effects of water level variations due to larger scale weather systems and broader scale hydrodynamic forcing. Calibration of the forcings is still required at a local scale as detailed in Section 4.

## 3.4 Existing outfall and discharge regime

An analysis of SCADA data from the Raglan WWTP has been carried out for the period between January 2015 and May 2019. Periods when there were very low discharges and discharges of less than 45 minutes have been filtered from the dataset.

The median discharge duration is 2 hours and 15 minutes, the median start time is 15 minutes after high water and the average flow rate is 0.058 $m^3/s$.

Figures 3-6 and 3-7 show the distribution of the duration of the discharge and the start time of the discharge relative to high water as summarised in Tables 3-1 and 3-2.

A typical sequence of discharges and data from the Manu Bay tide data is shown in Figure 3-8.

The current flow regime has been used to provide the baseline conditions in terms of the potential impact of the treated wastewater plume on the marine receiving environment.

Figure 3-5 shows the location of the existing outfall from a diver inspection report from 2009 (Diver Services Limited 2009). Data in that report indicates there is a 0.22m outer diameter pipeline that emerges some 65 m from sandbank towards the road and that that pipe sits in a depth of approximately 0.3 m below chart datum.


Figure 3-5. Location of the existing pipeline (dashed red line) and discharge point (Diver Services Limited 2009)

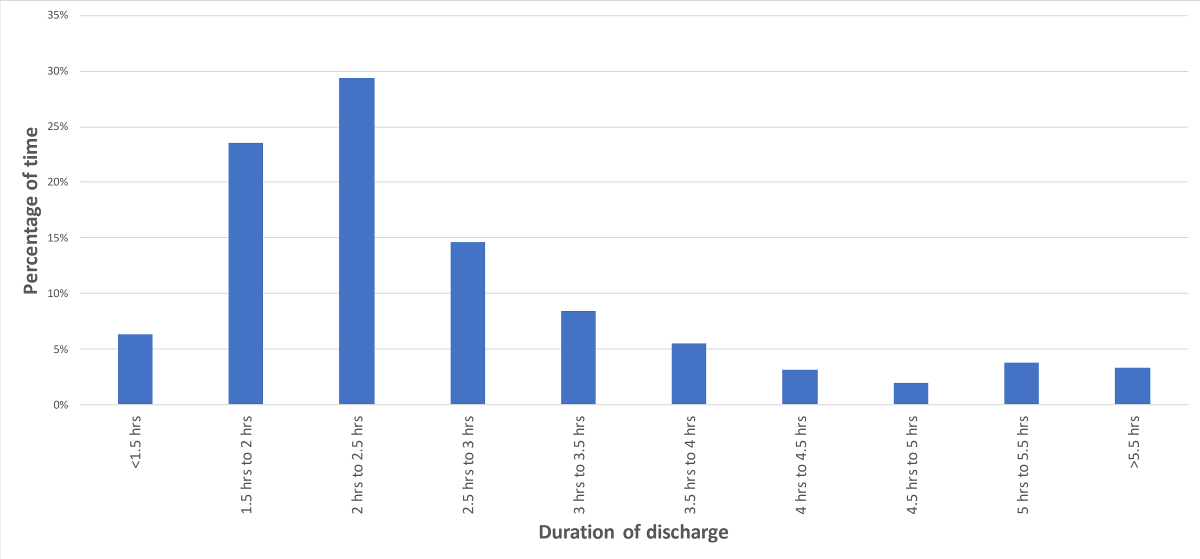


Figure 3-6. Distribution of duration of discharge from the Raglan WWTP. Analysis based on SCADA data from January 2015 through to May 2019.

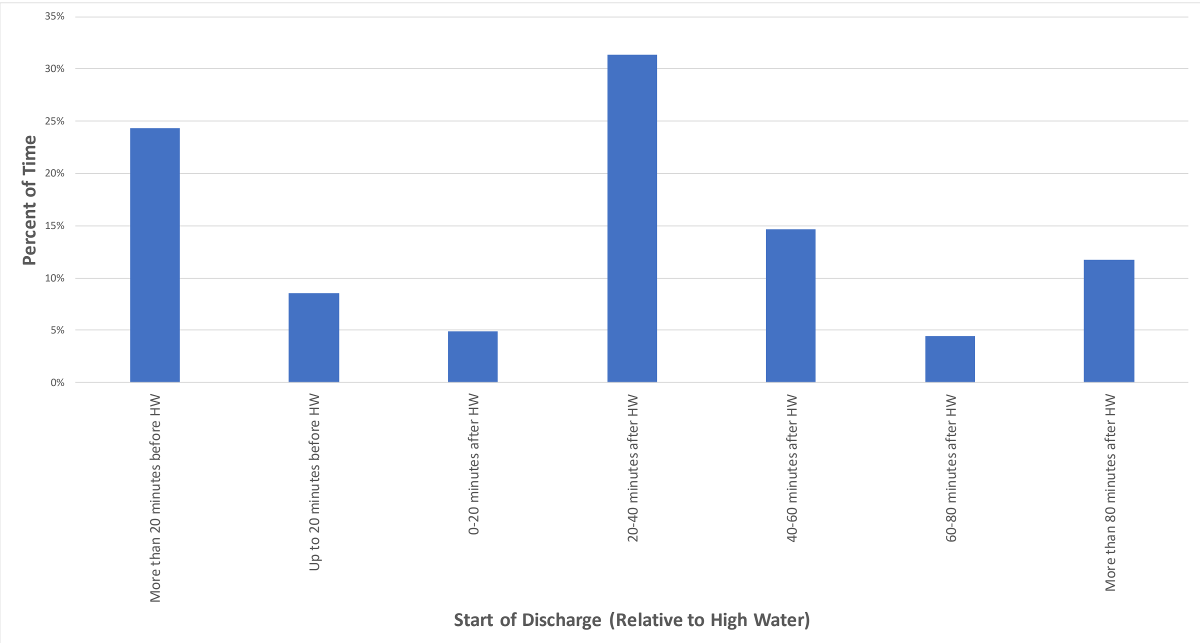


Figure 3-7. Distribution of start time (relative to high tide) of the Raglan WWTP discharge. Analysis based on SCADA data from January 2015 through to May 2019.

Table 3-1. Discharge start time (relative to high water) based on Raglan WWTP discharge data (January 2015 and May 2019).

| Discharge Start Time (relative to High Water) | Percentage of time |
|---|---|
| Before HW | 32.90% |
| HW to HW + 0.5 hrs | 27.06% |
| HW + 0.5 hrs to HW + 1.0 hrs | 24.77% |
| HW + 1.0 hrs to HW + 1.5 hrs | 14.48% |
| More than 1.5 hrs after HW | 0.78% |

Table 3-2. Discharge duration based on Raglan WWTP discharge data (January 2015 and May 2019).

| Discharge Duration (hrs) | Percentage of time |
|---|---|
| <1.5 hrs | 6.3% |
| 1.5 hrs to 2 hrs | 23.6% |
| 2 hrs to 2.5 hrs | 29.4% |
| 2.5 hrs to 3 hrs | 14.6% |
| 3 hrs to 3.5 hrs | 8.4% |
| 3.5 hrs to 4 hrs | 5.5% |
| 4 hrs to 4.5 hrs | 3.2% |
| 4.5 hrs to 5 hrs | 2.0% |
| 5 hrs to 5.5 hrs | 3.8% |
| >5.5 hrs | 3.3% |

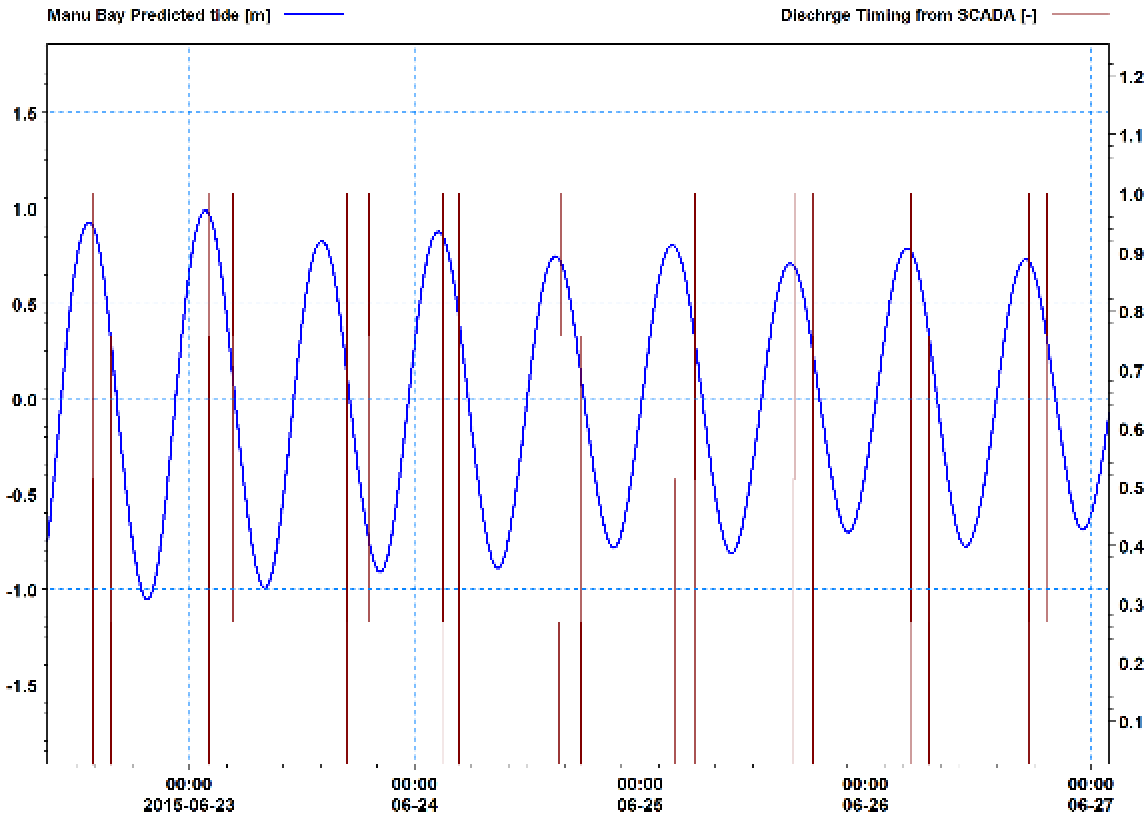


Figure 3-8. Example of timing of the Raglan WWTP discharge using SCADA data and Manu Bay tide gauge data.

## 3.5 Manu Bay Tide data

Waikato Regional Council data from the Manu Bay and Wharf tide gauges have been used to quantify the effects of tides and other forcings (winds and waves) on water level variations.

Figure 3-9 shows the residual (non-tidal) and tidal component of water level variations from the Manu Bay site for the period between the 16th of March 2018 to the 25st of January 2019.

The spring tide range varies from 1.6 to 1.8 and the neap tide range is less than 1.4 m.

In addition, it can be seen that the non-tidal water level variations[1] can be significant with non-tidal residual water levels of more than 0.3 m occurring for around 1.7% of the time and non-tidal residual water levels of less than -0.3 m occurring for around 1.2% of the time.

Table 3-3 shows the amplitude and phasing of the main tidal constituents.

Importantly the relationship between the observed residual at Manu Bay and that at the Wharf during the period where there are overlapping records (6th November 2011 through to the 12th of June 2013) shows a strong correlation (Figure 3-10). This indicates that regional wide forcings (driving the residuals at Manu) propagate through to the Wharf and that any effect of localised variations in winds within the harbour on water level variations is minimal.

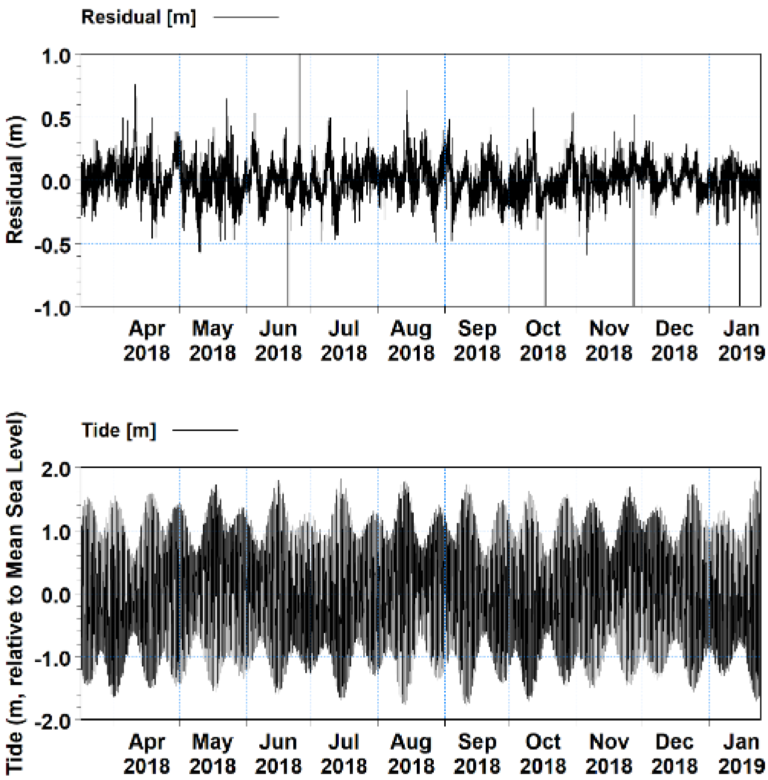


Figure 3-9. Time series of residual (nontidal) and tidal components of water level variations at the Manu Bay tide gauge site.

[1] These are the variations in water levels that occur due to the effects of winds, waves and atmospheric pressure variations.

Table 3-3 Manu Bay tide gauge constituent data

| Constituent | Amplitude (m) | Phase (º) |
|---|---|---|
| M2 | 1.125 | 272.7 |
| S2 | 0.310 | 314.7 |
| N2 | 0.223 | 253.9 |
| K2 | 0.076 | 306.0 |
| K1 | 0.056 | 186.4 |
| NU2 | 0.043 | 259.6 |
| 2N2 | 0.040 | 237.9 |
| MU2 | 0.038 | 239.2 |
| MSM | 0.028 | 99.6 |
| SSA | 0.026 | 170.4 |
| L2 | 0.025 | 284.2 |
| MF | 0.022 | 117.8 |
| MM | 0.021 | 349.7 |
| M4 | 0.019 | 16.4 |
| O1 | 0.017 | 84.3 |
| P1 | 0.014 | 177.3 |
| 2MS6 | 0.011 | 255.2 |
| LDA2 | 0.010 | 278.1 |

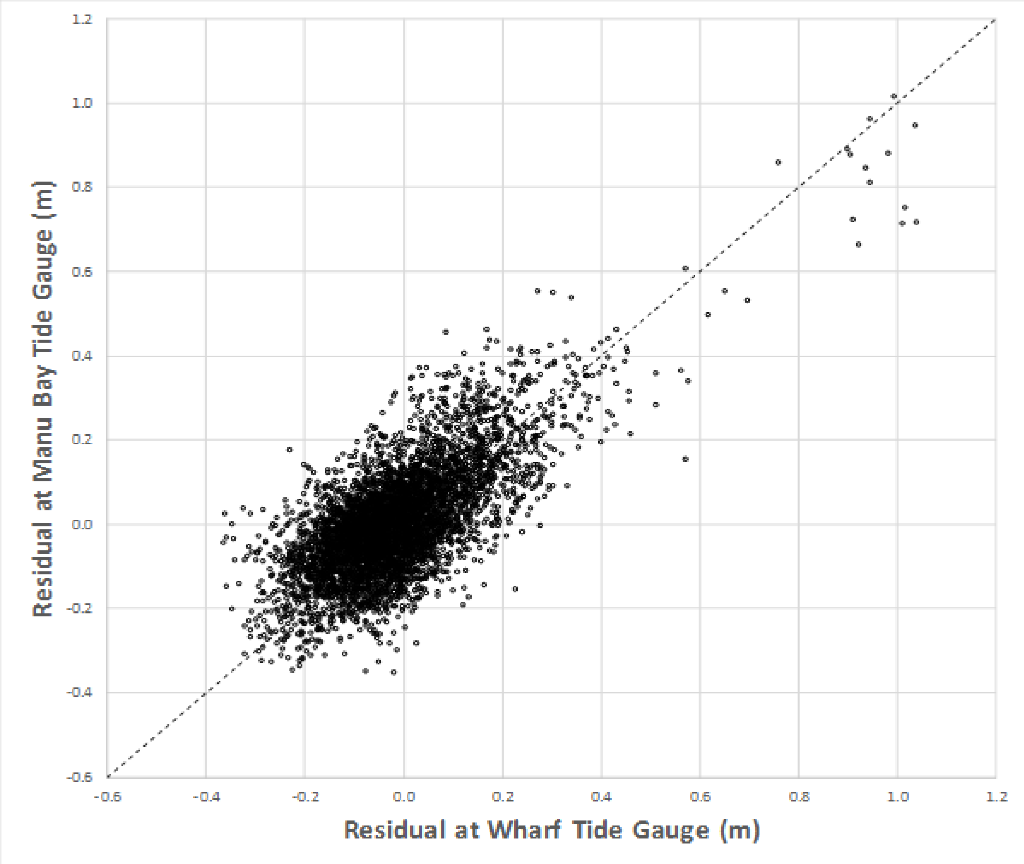


Figure 3-10. Residual water levels at the Wharf and Manu bay for the period of overlapping records (6/11/2012-12/6/2013).

# 4 Model Calibration

This section of the report provides the details of the calibration of the model against observed water level variations, currents and waves.

The three-dimensional model of the harbour and open coast has been calibrated against available field data. This includes tide gauge data at Manu Bay and the Wharf, current meter data at sites near the existing outfall, immediate offshore of the existing outfall in the main channel of the harbour and a site just offshore of the bar. In addition, the model has been calibrated against a series current profile transects running along and across the entrance and predicted salinities have been compared to long term monitoring data in the wider harbour.

## 4.1 Tides

Model simulations were carried out for the calibration of water level variations for both Manu Bay and the Wharf.

Figure 4-1 shows the predicted and observed water levels at Manu Bay for the period May-July 2018. Table 4-1 shows that the overall fit is very good.

Figure 4-2 shows the predicted and observed water levels at the Raglan Wharf for the December 2012. Table 4-2 shows that the overall fit is very good.

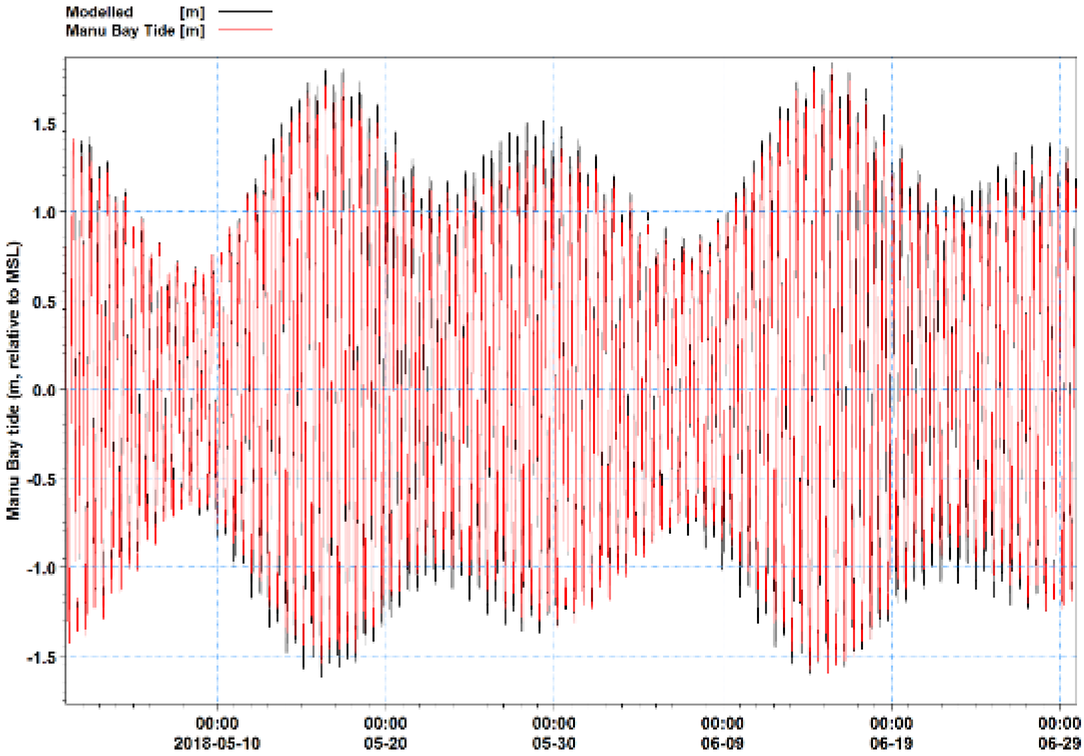


Figure 4-1. Observed and predicted water levels at the Manu Bay May-June 2018.

Table 4-1 Manu Bay tide gauge calibration indices.

| | Value |
|---|---|
| Mean Error | -0.02 m |
| Mean Absolute Error | 0.04 m |
| Root Mean Square Error | 0.05 m |
| Std. dev of Residuals | 0.05 m |
| Coefficient of Determination | 0.995 |
| Coefficient of Efficiency | 0.994 |
| Index of Agreement | 0.999 |

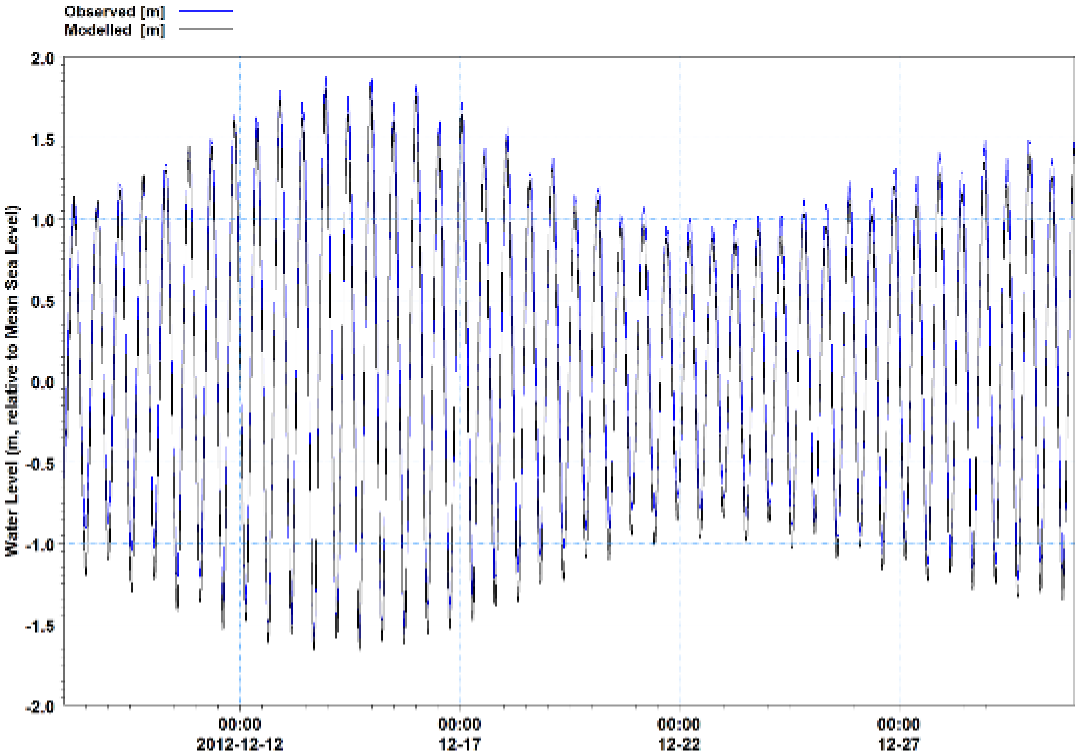


Figure 4-2. Observed and predicted water levels at the Raglan Wharf December 2012.

Table 4-2 Raglan Wharf tide gauge calibration indices.

| | Value |
|---|---|
| Mean Error | 0.06 m |
| Mean Absolute Error | 0.11 m |
| Root Mean Square Error | 0.14 m |
| Std. dev of Residuals | 0.12 m |
| Coefficient of Determination | 0.980 |
| Coefficient of Efficiency | 0.974 |
| Index of Agreement | 0.994 |

## 4.2 NIWA Current meter

The model was run for the period of the current meter deployment at the existing outfall site in 1996 (NIWA, 1996). The exact location of the deployment was given in the NIWA report and, as referenced in the NIWA report, the bathymetry in and around the existing outfall can change with time.

Figure 4-3 shows the time-series plot of the observed data and the data from the model. Given the uncertainties around the actual location of the current meter and the dynamics of the bathymetry near the exiting outfall the overall match between the observed data and the model is a relatively good match as shown by the calibration indices in Table 4-3.

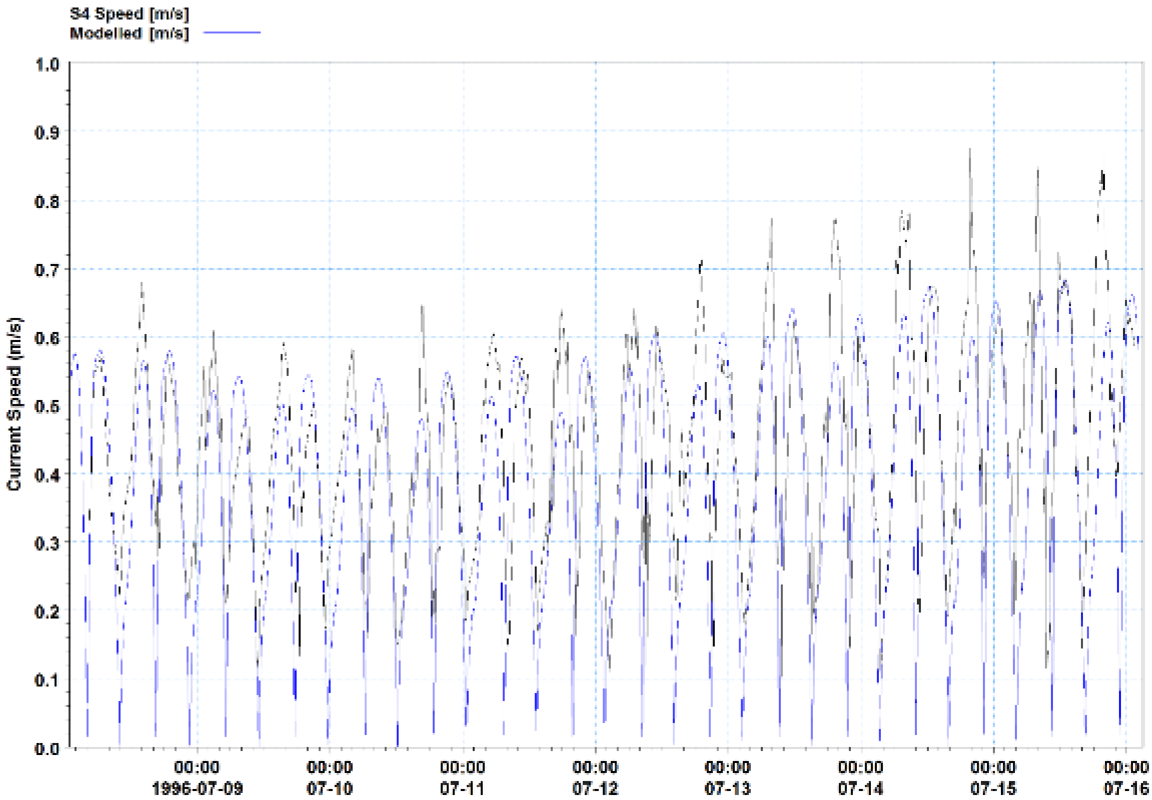


Figure 4-3. Calibration against the NIWA current metre data at the existing outfall location.

Table 4-3 NIWA current meter calibration indices.

| | Value |
|---|---|
| Mean Error | 0.06 m |
| Mean Absolute Error | 0.11 m |
| Root Mean Square Error | 0.14 m |
| Std. dev of Residuals | 0.12 m |
| Coefficient of Determination | 0.525 |
| Coefficient of Efficiency | 0.385 |
| Index of Agreement | 0.818 |

## 4.3 ADCP transects

Measurements from a cross-channel boat-mounted acoustic doppler current profiler (ADCP) survey on 14th of April have been used to validate the MIKE3 model in the area immediately offshore of the entrance. This data has kindly been supplied for use in this study by the University of Waikato and was collected as part of the PhD study looking at the seabed dynamics in ebb-tide deltas (Harrison, 2015).

During the outgoing tide of the 14th of April 2014, a series of transects were run and with the ADCP recording current speed, direction and total water depth. These included a number of transects near the entrance (Figure 4-4), transects offshore of the bar (Figure 4-5) and longitudinal transects running along the thalweg of the entrance channel (Figure 4-6).

A full quantitative calibration of the three-dimensional currents has not been carried out. This is because of the difficulty with matching the data from the ADCP (which is moving in time and recording a value every 0.3 m of the water column) with the equivalent model predictions

However, a qualitative comparison of currents predicted by the MIKE 3 model and those observed through the water column at a number of the transects shows very good agreement in terms of the distribution of speeds through the water column and along the transects. For example, for Transect 21 (running the full length of the entrance channel, Figure 4-6), it can be seen that the overall pattern and magnitude of currents are reasonably well represented in the model (Figure 4-7).

A quantitative comparison of the depth-averaged observed and modelled currents was done for each of the transects.

Firstly, depth-averaged currents were calculated from the ADCP data. Next, the data was averaged in time so that for each minute when observations were taken, a mean location and depth-averaged current was calculated. Model data was then extracted for each of these location at the time of the observations. The scatter plot of the observed and predicted water depth and depth-averaged currents are shown in Figure 4-8 and Figure 4-9 respectively. The differences between the observed and modelled depths provides an indication of the dynamic nature of the bathymetry for this part of the harbour.

Overall, there is a very good match between the observed and modelled data (Table 4-4) with the model slightly over predicting water depths (although the ADCP data does not allow for any heave, pitch or roll adjustments) and tending to slightly over predict some of the highest current.

For the transverse transects, the total observed flux ($m^3$) passing through the transect was compared to the predicted flux form the model (Figure 4-10). The observed and predicted depth averaged currents for the longitudinal transects are shown in Figure 4-11 to Figure 4-15.

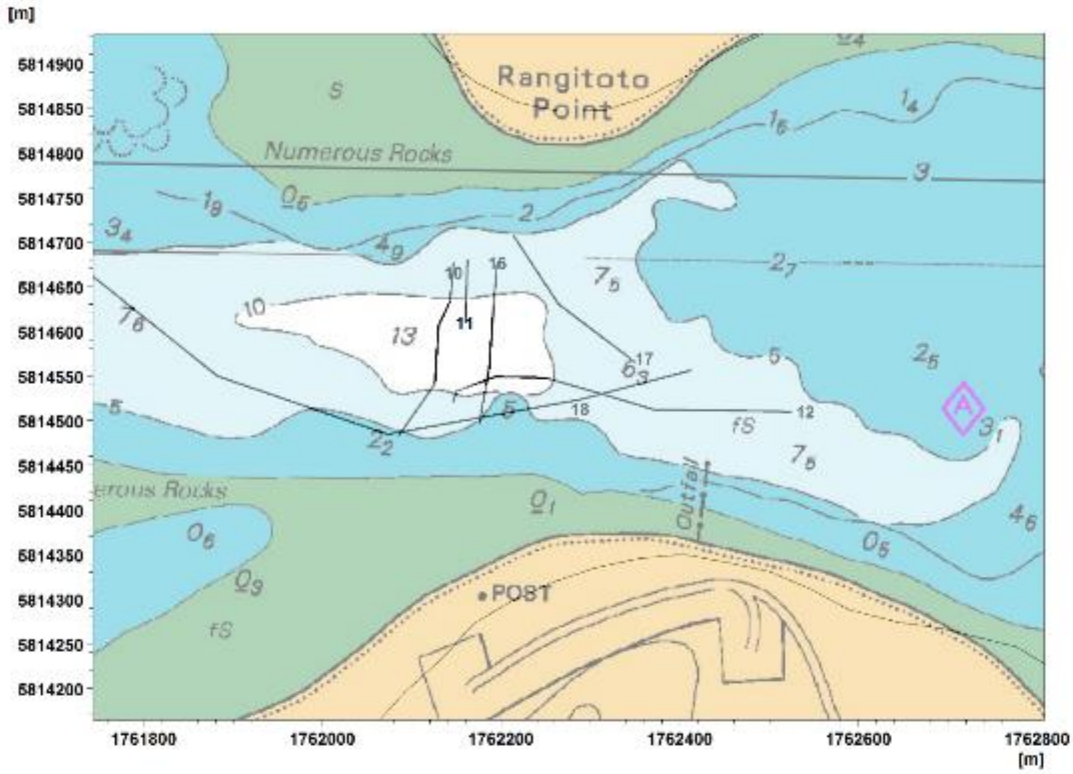

Figure 4-4. Entrance transects where water column currents were observed on outgoing tide of the afternoon of the 14th of April 2014.

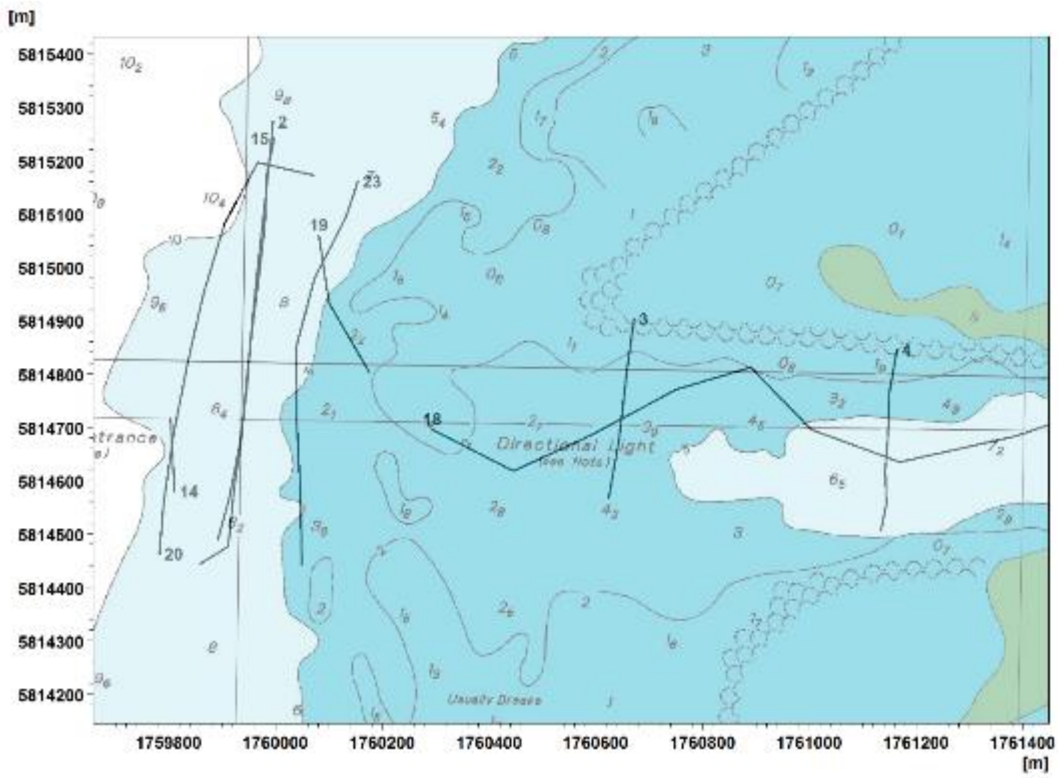

Figure 4-5. Transects offshore of the entrance where water column currents were observed on outgoing tide of the afternoon of the 14th of April 2014.

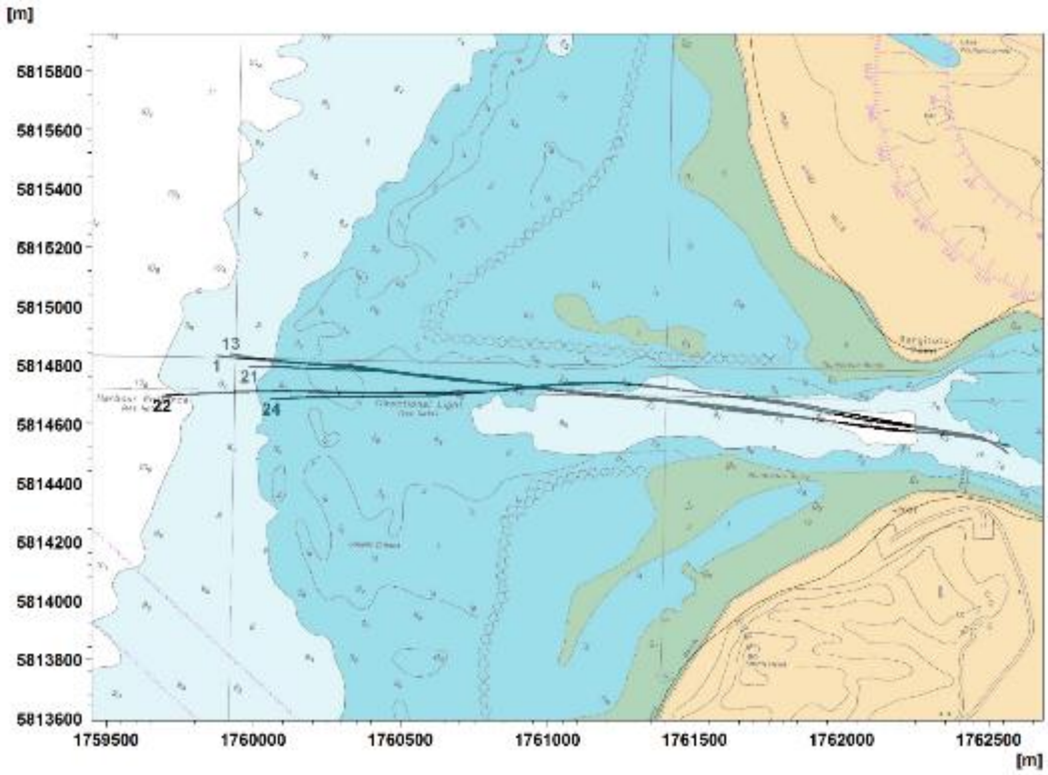


Figure 4-6. Longitudinal transects where water column currents were observed on outgoing tide of the afternoon of the 14th of April 2014.

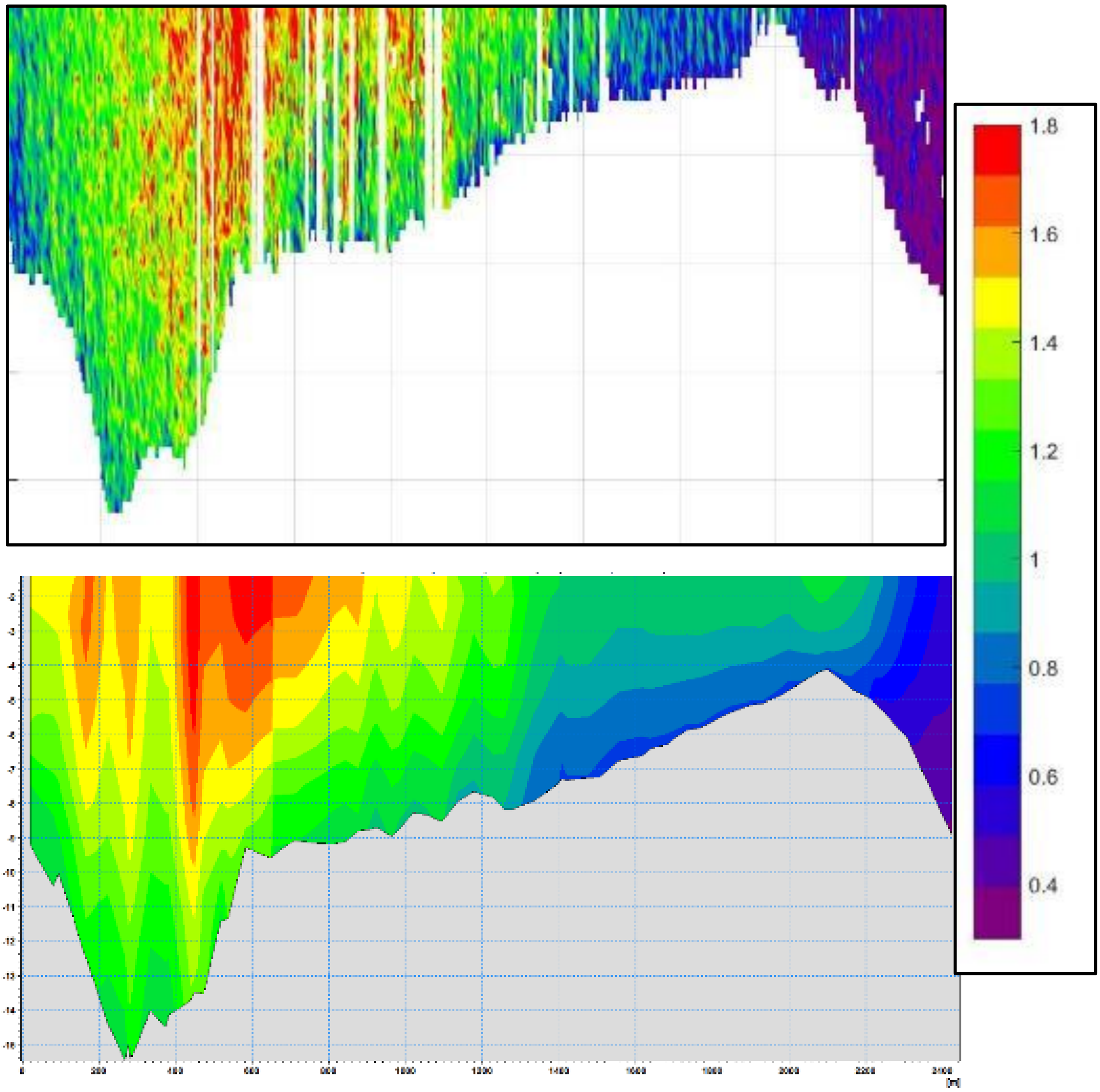


Figure 4-7. Observed (top) and modelled currents for Transect 21 (Figure 4-6).

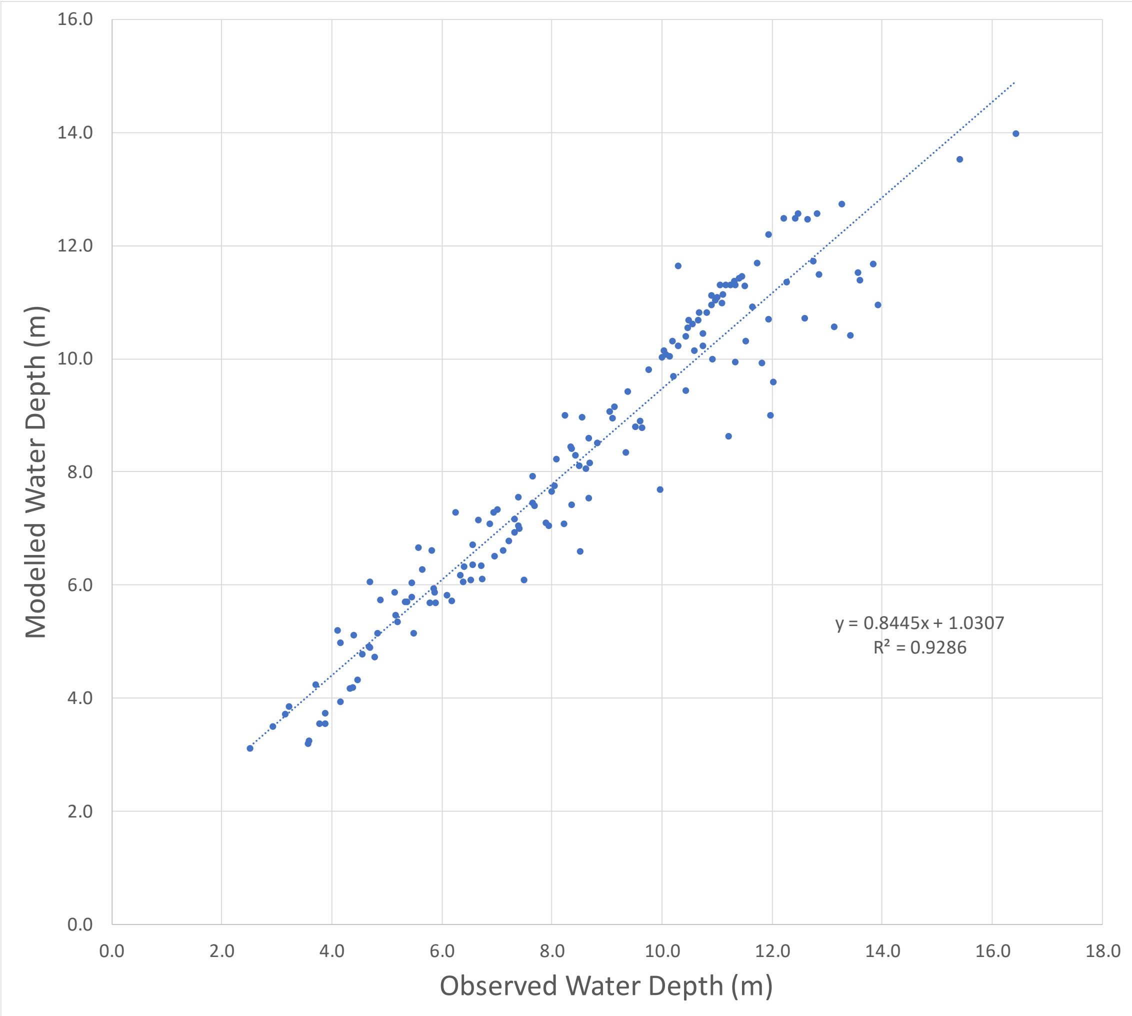


Figure 4-8. Observed and predicted water depths for all of the ADCP data collected on the 14th of April 2014.

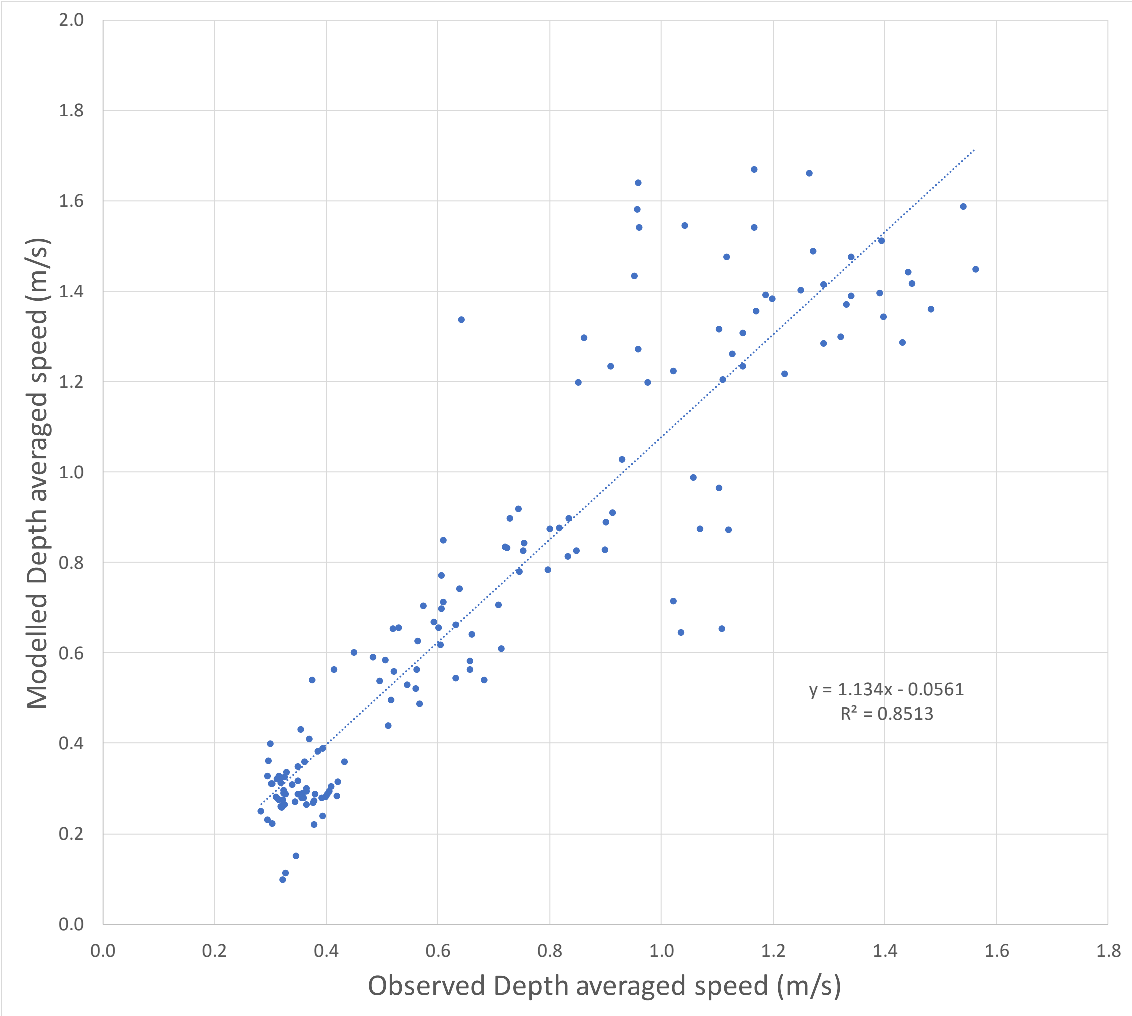


Figure 4-9. Observed and predicted depth-averaged currents for all of the ADCP data collected on the 14th of April 2014.

Table 4-4 Statistical indices for all of the ADCP transect data

| Indices | Depth Averaged Current | Water Depth |
|---|---|---|
| Mean Error | -0.04 m/s | 0.30 m |
| Mean Absolute Error | 0.12 m/s | 0.59 m |
| Root Mean Square Error | 0.18 m/s | 0.90 m |
| Std. dev of Residuals | 0.18 m/s | 0.85 m |
| Coefficient of Determination | 0.852 | 0.929 |
| Coefficient of Efficiency | 0.833 | 0.884 |
| Index of Agreement | 0.948 | 0.974 |

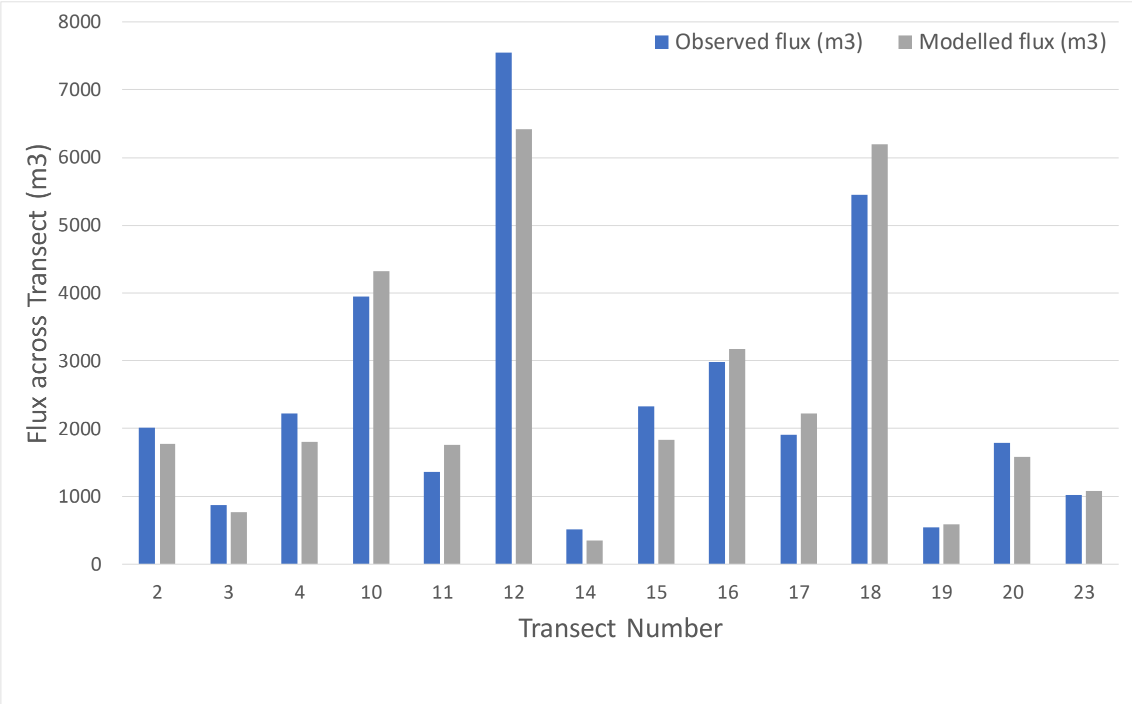


Figure 4-10. Observed and predicted total flux passing through the transverse transects.

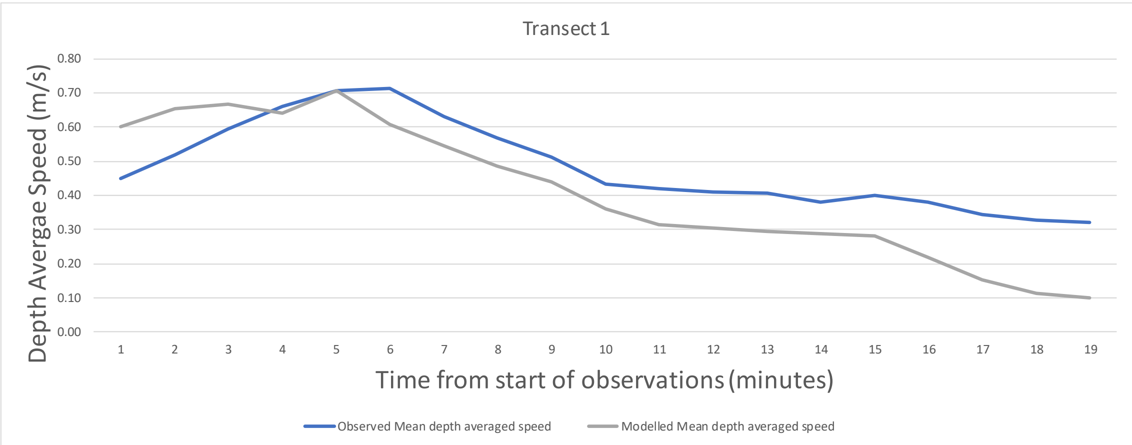


Figure 4-11. Observed and predicted depth-averaged currents for longitudinal Transect 1 (Figure 4-6).

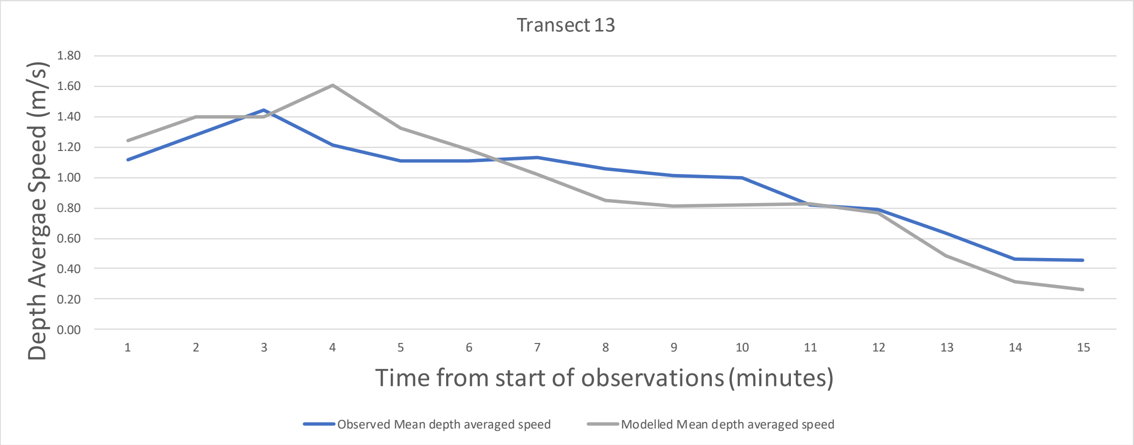


Figure 4-12 Observed and predicted depth-averaged currents for longitudinal Transect 13 (Figure 4-6)

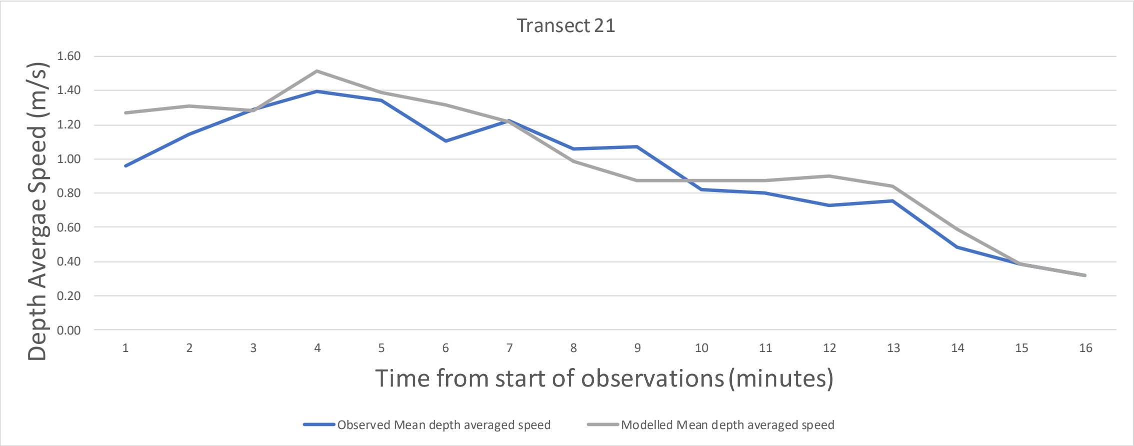


Figure 4-13 Observed and predicted depth-averaged currents for longitudinal Transect 21 (Figure 4-6).

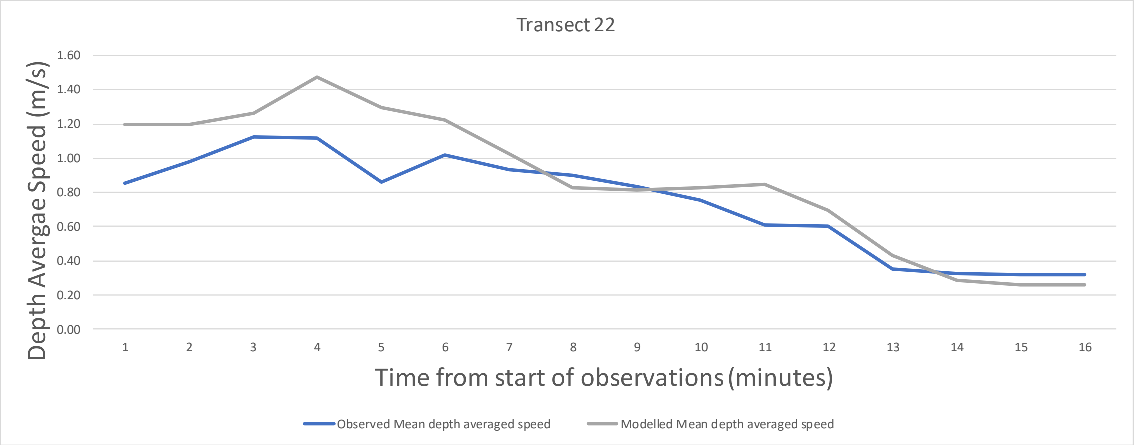


Figure 4-14 Observed and predicted depth-averaged currents for longitudinal Transect 22 (Figure 4-6).

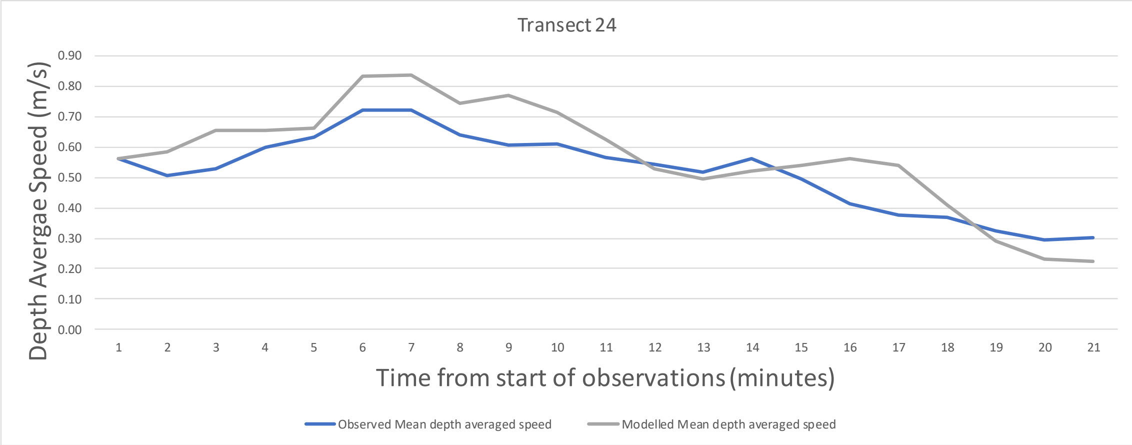


Figure 4-15 Observed and predicted depth-averaged currents for longitudinal Transect 24 (Figure 4-5).

## 4.4 Harbour Wide Salinity

The predicted minimum average salinities from the 2018 model run were extracted at the Waikato Regional Council monitoring sites (Figure 4-16) and compared to the minimum of the observed data (). Figure 4-17 shows the regression plot of the observed mean and the modelled mean. While this is not a strict calibration of the model (as the model and observations do not overlap) this figure shows that overall mixing of freshwater during periods of higher freshwater inflows in the harbour is reasonably well modelled especially when considering that the inputs of freshwater to the system have only a limited influence of the flushing of the harbour near the entrance (Greer et al. 2016) and therefore have little influence on the dynamics of the treated wastewater plume.


Figure 4-16. Waikato Regional Council water quality monitoring sites.

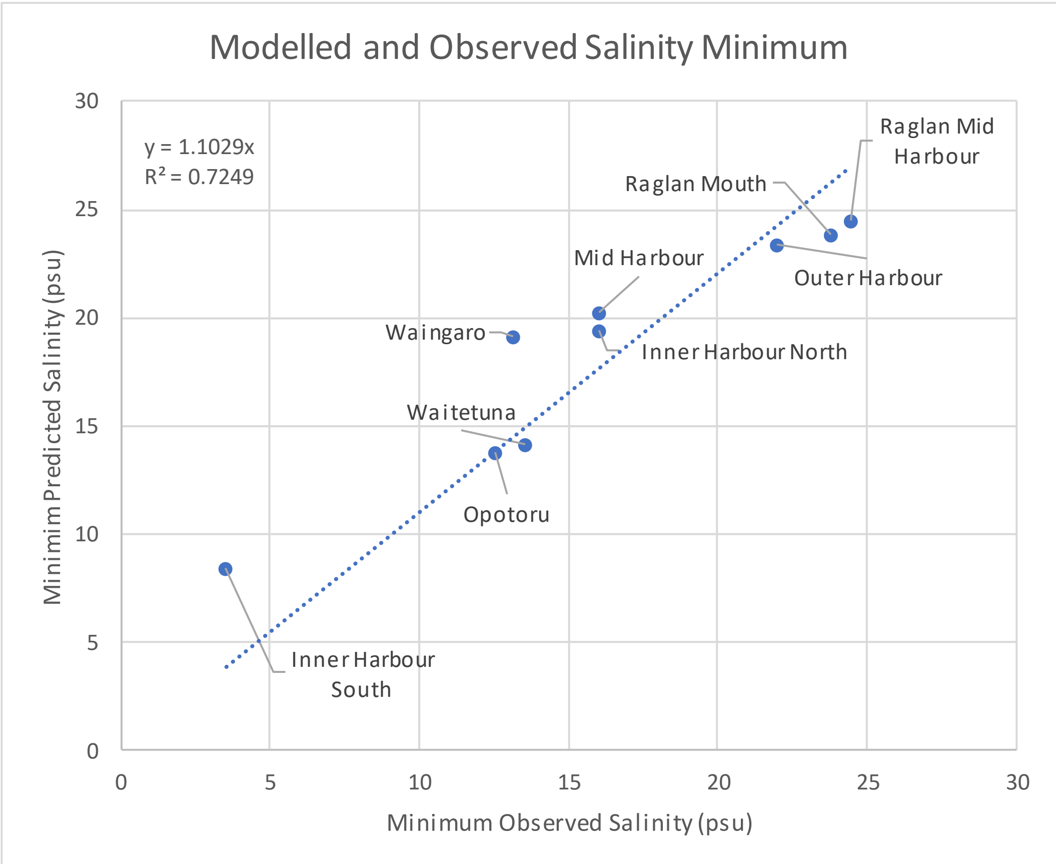


Figure 4-17. Predicted minimum salinity from the 2081 model simulation versus the observed salinity minimum the Waikato Regional Council monitoring sites (Figure 4-16).

## 4.5 Waves

Spectral wave conditions were modelled in the study area using the MIKE 21 SW model (DHI, 2017). The multi-scale processes controlling the generation, propagation and dissipation of waves from offshore to the Raglan Harbour entrance were modelled applying a nesting approach. Four domains (Global, New Zealand, Waikato and Raglan) using flexible meshes were setup providing directional spectral conditions along the open-boundaries of the nested domains as shown in Figure 4-18.

Global hourly surface winds extracted from the CFSR product (NCAR, 2016) were used to force the spectral wave model. Variations of the water elevation were represented in the MIKE 21 SW Raglan domain using timeseries of predicted tidal elevation obtained from the harmonic analysis of measurements in Manu Bay between 16/03/2018 and 25/01/2019.

Model significant wave heights and peak wave periods were compared against wave buoy data at Position WB (Table 4-5 and Figure 4-18) over a 2-week period in 2009. Results are shown in Figure 4-19.

Because of the limited number of swell events over this period, no statistics were calculated to quantify the degree of agreement between measurements and model outputs.

Results showed the model was realistically capturing the wave climate offshore of Raglan Harbour. While the model wave heights presented a good agreement with measurements, it has been showed that the model was over-estimating the peak wave periods. Considering the uncertainty related to the representation of the sand bars and wave-induced radiation stresses in hydrodynamic models in general, this bias in the peak wave period data is not considered to impact the results of the outfall dilution study. The spectral wave model was, therefore, used to generate one year of wave climate (2018) over the study area following the same strategy. Wave-induced radiation stresses were saved every 1 hour to force the hydrodynamic model.

Table 4-5 Coordinates of Position WB where measurements of wave heights and periods were provided for March 2009.

| Position | Coordinates (NZTM) | |
|---|---|---|
| | X (m) | Y (m) |
| WB | 1760137 | 5814817 |

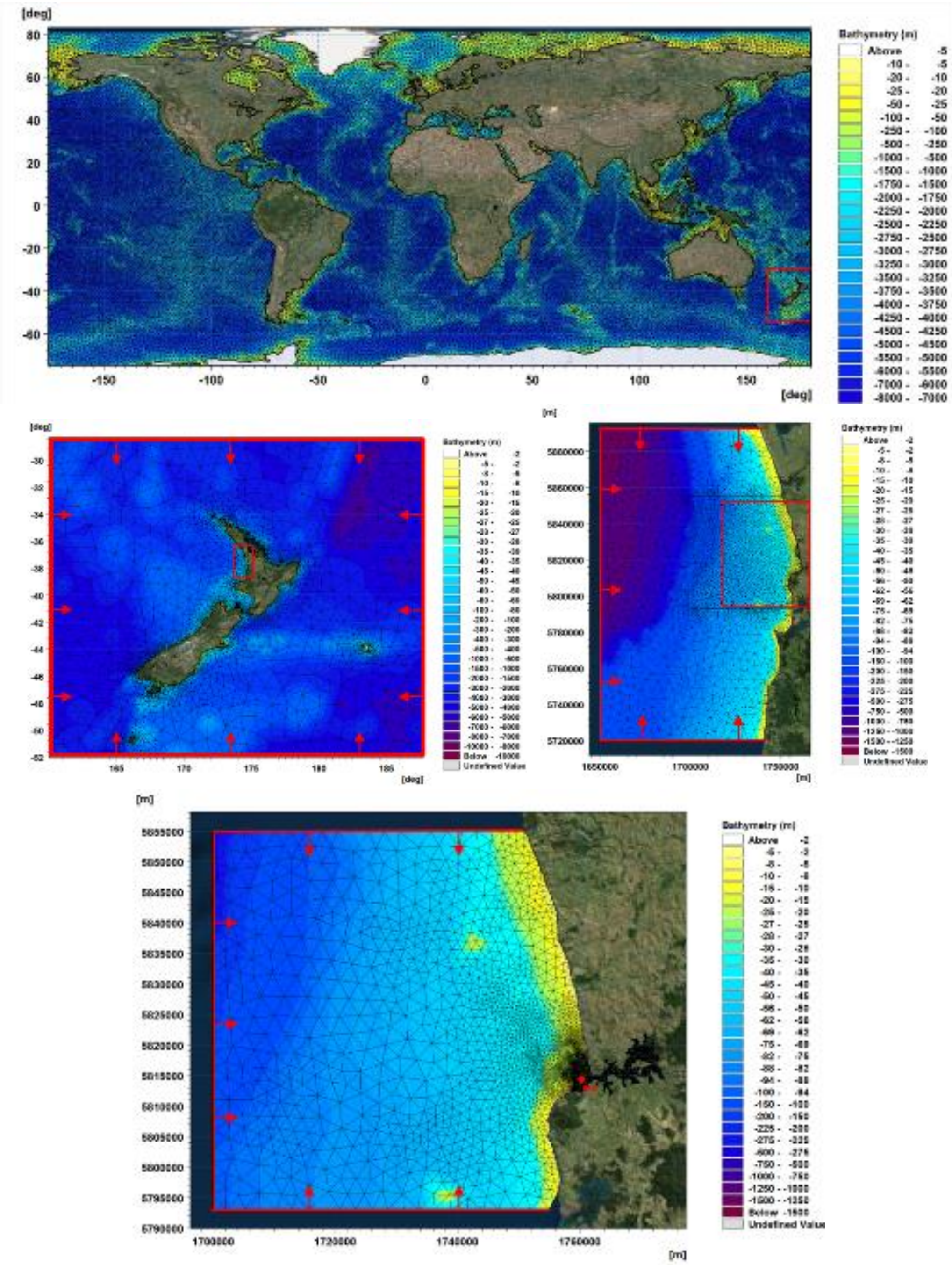


Figure 4-18 Nested domains set up in MIKE 21 SW to generate the spectral wave conditions from global to local scales. Directional spectra were applied from coarse to fine domains along the open boundaries (red lines and arrows). Model wave heights and periods were validated at the WB location over two weeks in 2009.


Figure 4-19 Comparisons between measured and model significant wave heights (top panel) and peak wave periods (bottom panels) at Position WB from 18/03/2009 to 01/04/2009.

# 5 Model Results

## 5.1 Hydrodynamics

Figure 5-1 shows the time-series of the predicted surface layer currents for a site in the middle of the main channel near the entrance to Raglan Harbour. The data shows that there is a relatively strong imbalance at this site between the incoming (flood) and outgoing (ebb) currents with stronger currents occurring on the outgoing tide (which is more pronounced on during spring tides).

Appendix D provides snapshots of the  predicted surface layer currents from the calibrated model over a neap tidal cycle (20-21st March 2009) and spring tidal cycle (27-28th March 2009). The figures show the relatively strong gradients in currents across the channel from the existing outfall location and the complexities of the offshore currents which are influenced by both wind and wave conditions.

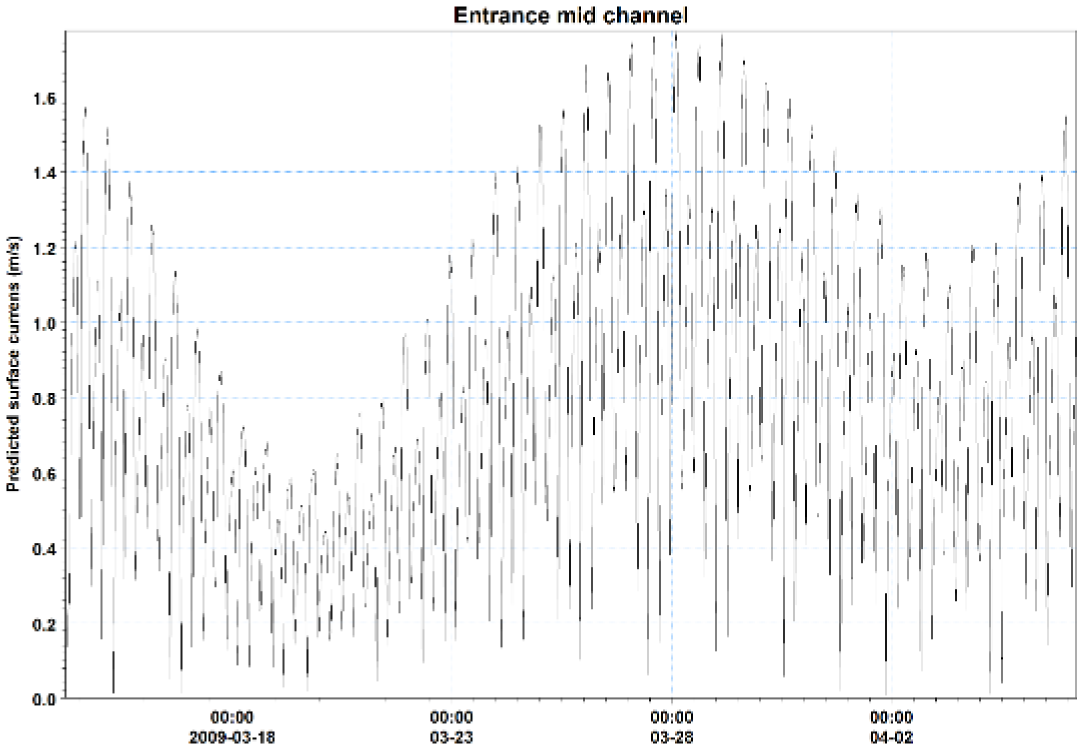


Figure 5-1. Time-series of predicted surface currents within the main channel near the entrance to Raglan Harbour.

## 5.2 Near Field Modelling – Existing Outfall

The following section of the report provides an overview of the near-field modelling that has been done. The near-field zone is defined as the area where the mixing behaviour of the plume is driven by the geometry of the outfall, along with the momentum and buoyancy of the wastewater plume.

For the near-field modelling, CORMIX has been used to predict the behaviour of the wastewater plume within the first few hundred metres of the discharge point. For further details of near-field modelling methodology see the CORMIX User Manual (Doneker and Jirka, 2007).

Inputs for the CORMIX model include the treated wastewater discharge rate, its density, the geometry of the outfall and the properties of the receiving water body (i.e. current speed and assumed density). The model then calculates the mixing behaviour of the wastewater as it is discharges from the outfall and as it interacts with the receiving water body, which is assumed to be stationary in time and spatially uniform.

Based on outputs from the calibrated hydrodynamic model, a series of CORMIX model runs have been completed for a number of schematic conditions which are representative of the ambient conditions that occur during the discharge window (Table 5-1).

Table 5-1. Schematic concentrations modelled using CORMIX and a description of when those conditions occur.

| Case | Water depth (m) | Ambient current (m/s) | Description |
|---|---|---|---|
| A | 3.3 | 0.20 | Conditions towards the end of the discharge window (Neap tide) |
| B | 2.7 | 0.40 | Conditions towards the end of the discharge window (Mean tide) |
| C | 2.3 | 0.60 | Conditions towards the end of the discharge window (Spring tide) |
| D | 4.3 | 0.10 | Just after high tide (Neap tide) |
| E | 5.5 | 0.10 | Just after high tide (Spring tide) |
| F | 4.3 | 0.50 | Early ebb tide (Spring tide) |
| G | 3.8 | 0.50 | Early ebb tide (Mean tide) |
| H | 3.4 | 0.50 | Early ebb tide (Neap tide) |

Table 5-2 provide the summary of the CORMIX results. Appendix B provides detailed outputs of the different schematic conditions modelled.

Because of the relatively low discharge rate (0.06 $m^3/s$) being considered and the diameter of the pipe (0.22m), the jet momentum is relatively low so the extent of the CORMIX near-field zone is less than 10m from the discharge point.

Typically, a mixing zone extends beyond the near-field region and considers some degree of "reasonable mixing[2]".

The results show that the lowest level of dilution occur near high water when ambient currents are the lowest. Under these conditions, the plume occupies the top 10% of the water column and a dilution of around 50-fold is achieved 100 m from the outfall. Based on observed current at the outfall site, such conditions only occur for around than 5% of the time (i.e. about 45 minutes per tide).

At other phases of the tide, much higher levels of dilutions are achieved and the plume becomes fully mixed through the water column within 180 metres of the outfall.

Based on these CORMIX simulations, a mixing zone of 150 m (i.e. before complete vertical mixing) would be appropriate for the existing outfall.

Table 5-2. CORMIX model results for the schematic conditions modelled.

| Case | Description | Plume thickness (% of water column) | Edge of Near Field Mixing (m) | Dilution at edge of Near Field Mixing | Distance to fully mixed (m) | Dilution when fully mixed | Dilution at 100 m |
|---|---|---|---|---|---|---|---|
| A | Conditions towards the end of the discharge window (Neap tide) | 24% | 4 | 15 | 177 | 507 | 184 |
| B | Conditions towards the end of the discharge window (Mean tide) | 49% | 6 | 23 | 77 | 148 | 157 |
| C | Conditions towards the end of the discharge window (Spring tide) | 47% | 7 | 23 | 46 | 57 | 68 |
| D | Just after high tide (Neap tide) | 10% | 6 | 12 | - | - | 50 |
| E | Just after high tide (Spring tide) | 10% | 7 | 18 | - | - | 54 |
| F | Early ebb tide (Spring tide) | 46% | 8 | 39 | 136 | 459 | 336 |
| G | Early ebb tide (Mean tide) | 47% | 9 | 42 | 115 | 239 | 200 |
| H | Early ebb tide (Neap tide) | 46% | 10 | 41 | 80 | 130 | 137 |

[2] US EPA OpenFOAM documentation.

## 5.3 Near Field Modelling - Alternative Discharge Locations

To quantify the potential for increasing the initial dilution achieved by an extended outfall, a series of simulations of near-field mixing for a number of alternative outfall locations (Figure 5-2) were carried out using the empirical formula from the Water Research Centre Design Guide for Marine Treatment Schemes (WRC, 1990) as set out in Appendix C. These equations assumed a non-stratified water column, consider buoyancy effects and port configuration to provide estimates of near-field dilution. Previous work (DHI, 2016) has shown there is good agreement between the WRC approach and results obtained by the CORMIX model. The empirical formula was applied for the predicted water depths and currents from the calibrated hydrodynamic model that occur during the discharge window over each of the outfalls for all of 2018.

The minimum dilution achieved at each of the outfalls is shown in Table 5-3 for a 0.3m diameter pipe and for an outfall fitted with a 6 port, 100 mm diffuser (as recommended in NIWA, 1996).

The minimum dilution achieved using the WRC formula is consistent with the edge of near-field CORMIX results shown in Table 5-2. Results show that -

- Extending the outfall by 60m provides around three times the level of dilution and provides around the same increase in dilution as fitting a diffuser to the existing outfall.
- Moving the outfall further offshore and to the east (site A) doubles the minimum dilution achieved.
- Extending the outfall further offshore of Site A (Site B) provides the highest level of minimum dilution (twice that of Site A) because this site is in the deepest part of the channel and in an area of high currents (Section Table 5-1).
- Sites C to I provide a range of minimum dilutions which reflect the different depths at these sites and the gradients in predicted currents across the channel (Section 5.1).

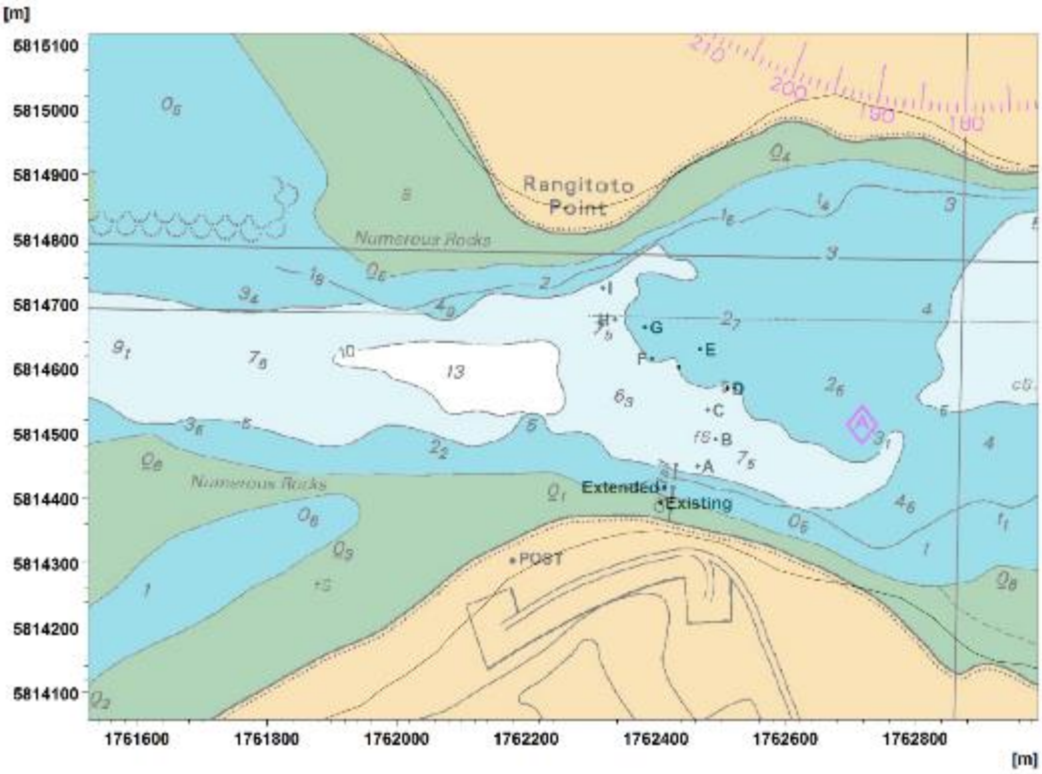


Figure 5-2. Existing outfall location and the alternative outfall sites considered.

Table 5-3. Minimum predicted dilution for all of 2018 using the WRC formula (Appendix C) for the existing and alternative outfall locations considered.

| Outfall | Minimum Dilution over outfall (0.3 m outfall) | Minimum Dilution over outfall (6 port, 100 mm diffuser) |
|---|---|---|
| Existing | 13 | 44 |
| Extended | 43 | 144 |
| A | 78 | 259 |
| B | 142 | 475 |
| C | 133 | 443 |
| D | 105 | 351 |
| E | 83 | 277 |
| F | 136 | 454 |
| G | 123 | 412 |
| H | 65 | 217 |
| I | 70 | 233 |

## 5.4 Wave conditions

Figure 5-3 shows the predicted wave heights at the existing outfall location for 2018. Significant wave heights of greater than 0.3m are exceeded 8% of the time and the maximum significant wave height for 2018 exceeds 1.0m. At a site 2500 m directly west of the existing outfall significant wave heights are regularly 2.0m with a maximum significant wave height of 5.2 m occurring.

Figure 5-4 shows the predicted significant wave height and direction for the event on the 6th of January 2018.

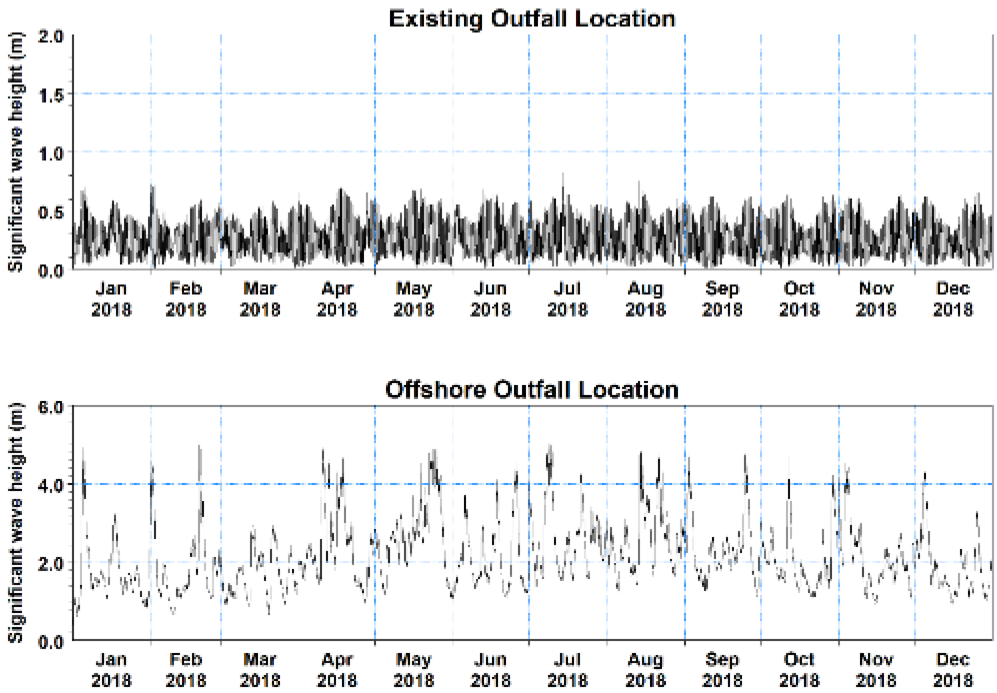


Figure 5-3. Predicted significant wave height at the existing outfall location and at an offshore outfall located 2500m offshore of the existing outfall.

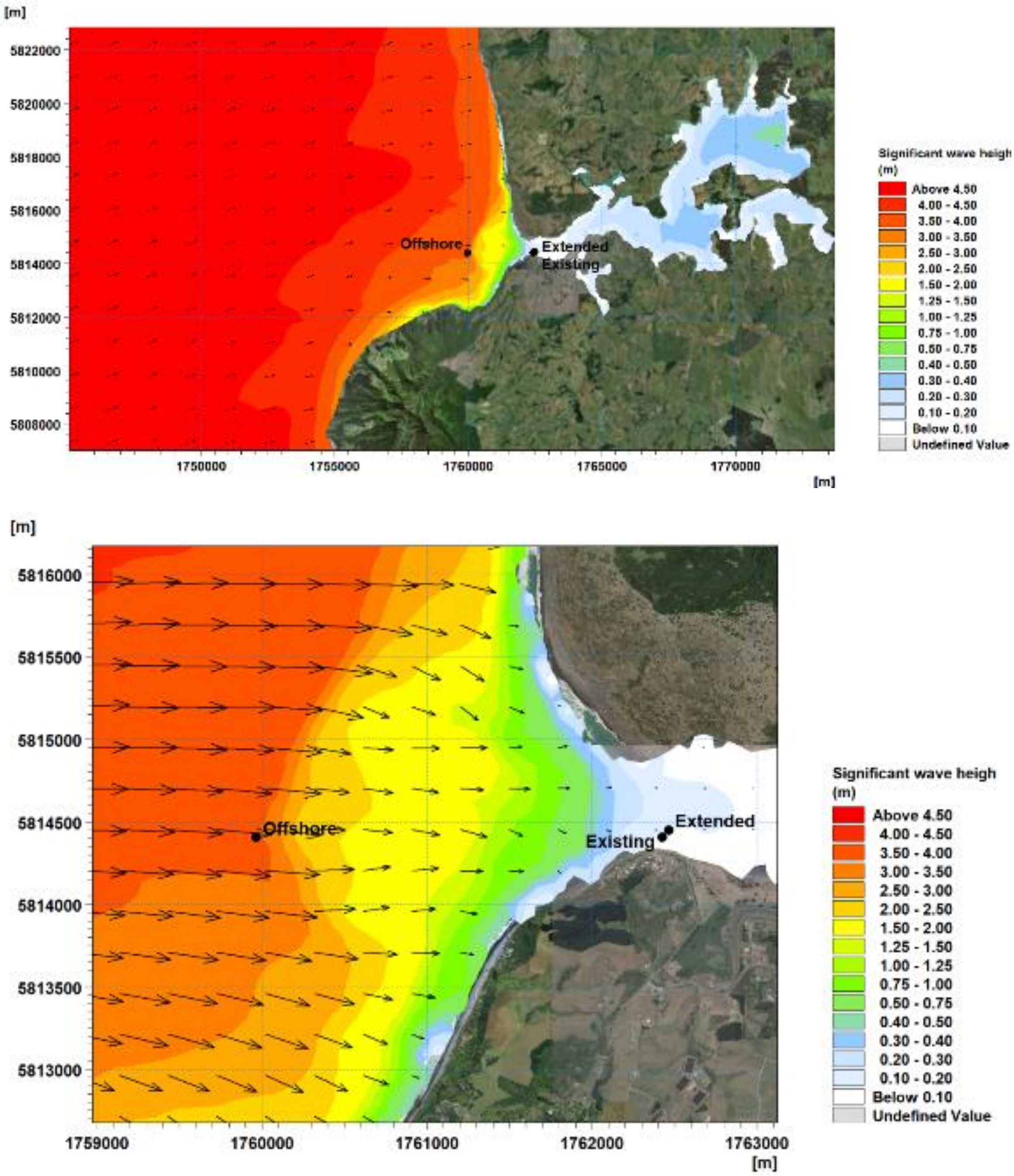


Figure 5-4. Predicted wave height and direction for a typical south-westerly swell event on the 6th of January 2018.

## 5.5 Far Field Modelling

In this section of the report results from the calibrated model are used to assess the relative impacts of the existing outfall and alternative configurations or locations. These results are used for comparative purposes only and are not used in the assessment of public health risk.

For the existing outfall, model results for the existing outfall under current and future discharge regimes are presented for a conservative tracer (i.e. no decay) and representative inactivation for both Enterococci and Viruses.

### 5.5.1 Alternative Discharge Locations

Here results from simulations with a constant inactivation of 0.083 $h^{-1}$ are used to compare the potential improvements that could be achieved with fitting a diffuser to the existing outfall, extending the outfall slightly offshore of the existing location and moving the outfall to an offshore location. In all cases the assumed end of pipe concentration has been set to 1000 cfu/100 mL (noting this concentration is much higher than those concentrations actually discharged).

Figure 5-5 shows the predicted 95$^{th}$ percentile concentration from the 2018 model simulation. Concentrations will be less than those presented for 95 percent of the time.

The area of highest 95$^{th}$ percentile concentrations extends around 200 m of the outfall with much lower concentrations extending some 1500 m inshore of the discharge point and up to 2000 m offshore.

Away from the discharge point, similar estimates occur when a diffuser is fitted to the existing outfall (Figure 5-6). Fitting a diffuser ensures full mixing of the treated wastewater plume which improves the level of dilution achieved within the first 500 m of the discharge point. Beyond this point, a discharge from the existing outfall without a diffuser is fully mixed due to the strong ambient currents (Section 5.1).

Extending the existing outfall some 60 m offshore improves the level of dilution achieved through a combination of achieving full mixing, increased ambient currents and greater water depths. Consequently, 95$^{th}$ percentile concentrations are reduced compared to the scenario when the discharge is via the existing outfall (Figure 5-7).

An offshore outfall provides the opportunity for much greater dilution due to increased water depth and the combined effect of currents and waves ensuring full mixing in the water column (Figure 5-8).

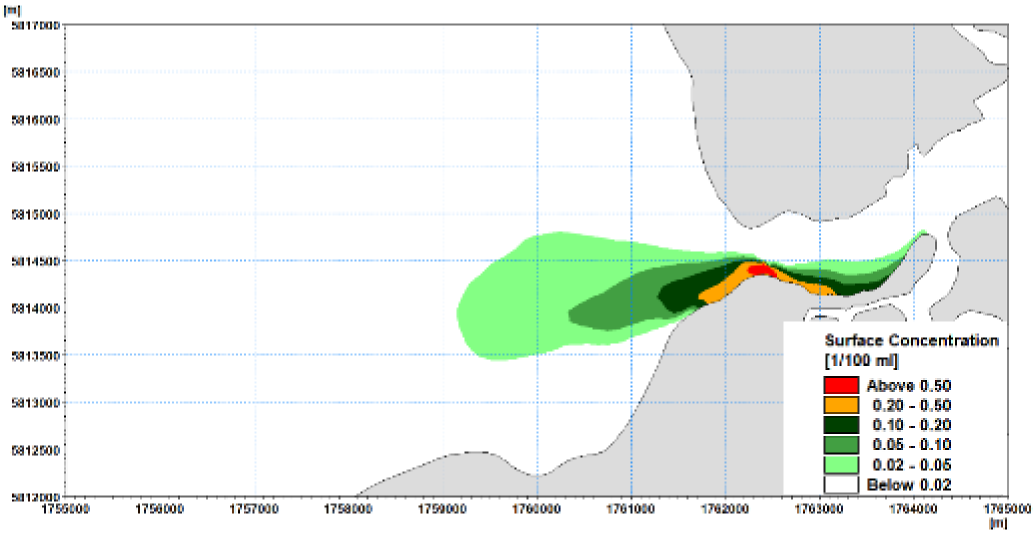


Figure 5-5. Estimated 95th percentile concentration for the existing outfall for a 2018 discharge regime. Concentrations are for an assumed end of pipe concentration of 1000 cfu/100 mL with a constant inactivation rate of 0.083 h-1.

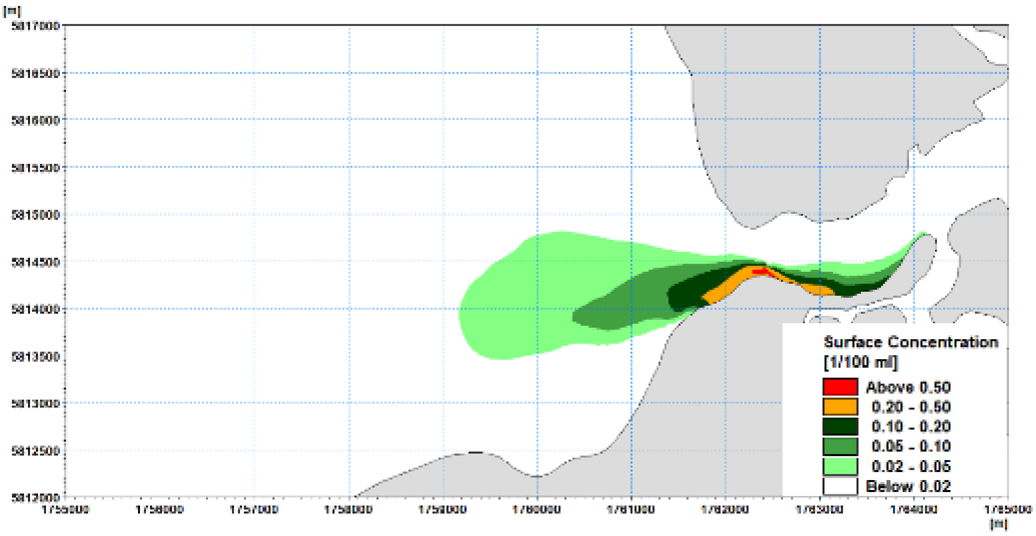


Figure 5-6. Estimated 95th percentile concentration for the existing outfall with a diffuser for a 2018 discharge regime. Concentrations are for an assumed end of pipe concentration of 1000 cfu/100 mL with a constant inactivation rate of 0.083 $h^{-1}$.

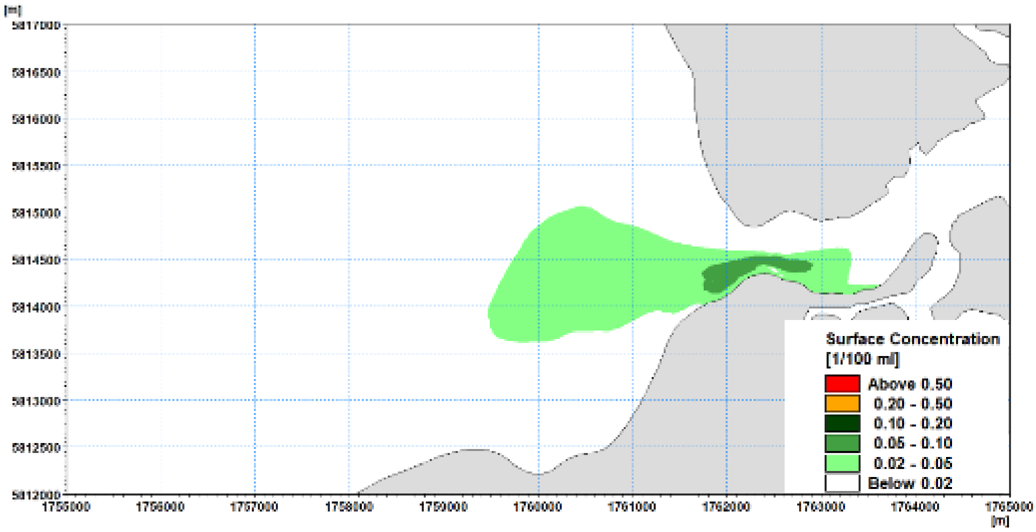


Figure 5-7. Estimated 95th percentile concentration for an extended outfall for a 2018 discharge regime. Concentrations are for an assumed end of pipe concentration of 1000 cfu/100 mL with a constant inactivation rate of 0.083 $h^{-1}$.

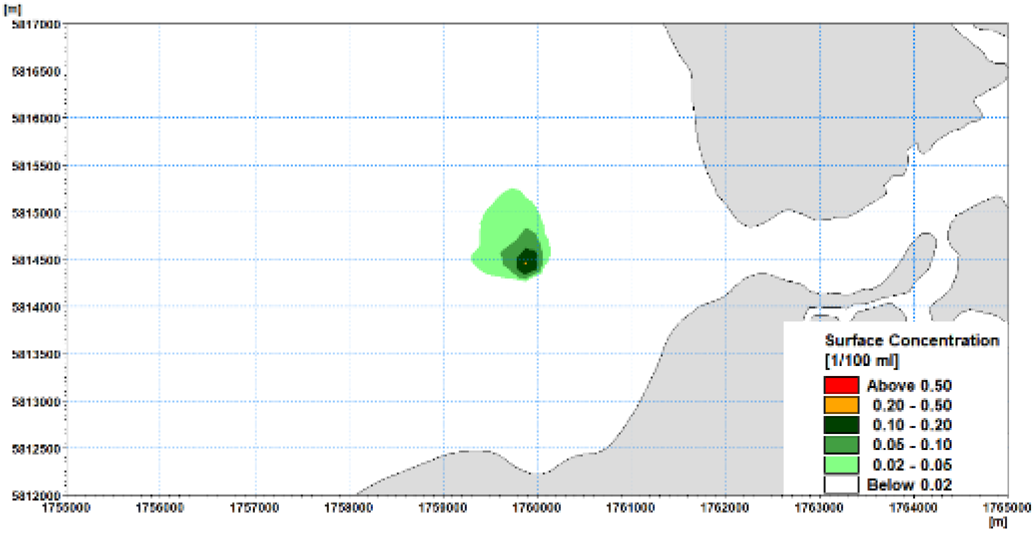


Figure 5-8. Estimated 95th percentile concentration for an offshore outfall for a 2018 discharge regime. Concentrations are for an assumed end of pipe concentration of 1000 cfu/100 mL with a constant inactivation rate of 0.083 $h^{-1}$.

### 5.5.2 Dilution Maps and Summary of Predictions at QMRA sites - Current Discharge Rate and Existing Discharge Regime

Figures 5-9 to 5-11 show the predicted 5$^{th}$ percentile dilution for the conservative tracer (no decay) and for Enterococci and Viruses based on the 2018 annual simulation using the current discharge regime (Section 3.4).

The plots show high level of dilution achieved outside the harbour itself and predominance of the northerly directed offshore currents. The general pattern of dispersal is consistent with the observed results from the dye test (Appendix A) .

The minimum dilution achieved 150 m from the existing outfall (i.e. the nominal mixing zone) is 70 which is consistent with the near-field model results (Section 5.1).

The results show that beyond 150 m of the outfall the 5$^{th}$ percentile dilutions are greater than 1000-fold, beyond 700 m of the outfall the 5$^{th}$ percentile dilutions are greater than 2000-fold and that beyond 1200 m the 5$^{th}$ percentile dilutions are greater than 4000-fold. These levels of dilution are achieved due to a combination of the level of near-field dilution (Section 5.1), the staging of the discharge which results a high degree of subsequent (far-field) dilution and little cumulative effect between individual discharges

Beyond 1000 m from the outfall there is a significant degree of dilution with the 5$^{th}$ percentile dilution often exceeding 30000-fold at the QMRA sites (Figure 5-12).

These levels of dilutions can be used to provide some context around specified water quality standards when considering end-of-pipe contaminant levels and the concentrations that may be achieved inside the mixing zone.

Tables 5-4 to 5-6 show the predicted percentile estimates at the 25 QMRA sites for the conservative tracer (no decay) and for Enterococci and Viruses.

At the outfall sites and Site R1 (nearest the outfall) the 1$^{th}$ percentile value (i.e. dilution is more than this for 99 percent of the time) is just under 200. At sites S1, S2, S4 and R2 (the next closest to the outfall) ) the 1$^{th}$ percentile values exceed 2000. At other offshore sites (S3, R10, R7, R3, R4, R5 and R6) the 1$^{th}$ percentile values exceed 12000, Inside the harbour (excluding Site S4) ) the 1$^{th}$ percentile values exceed 8000 and there is a strong gradient moving from sites nearer the entrance towards the middle section of the harbour.

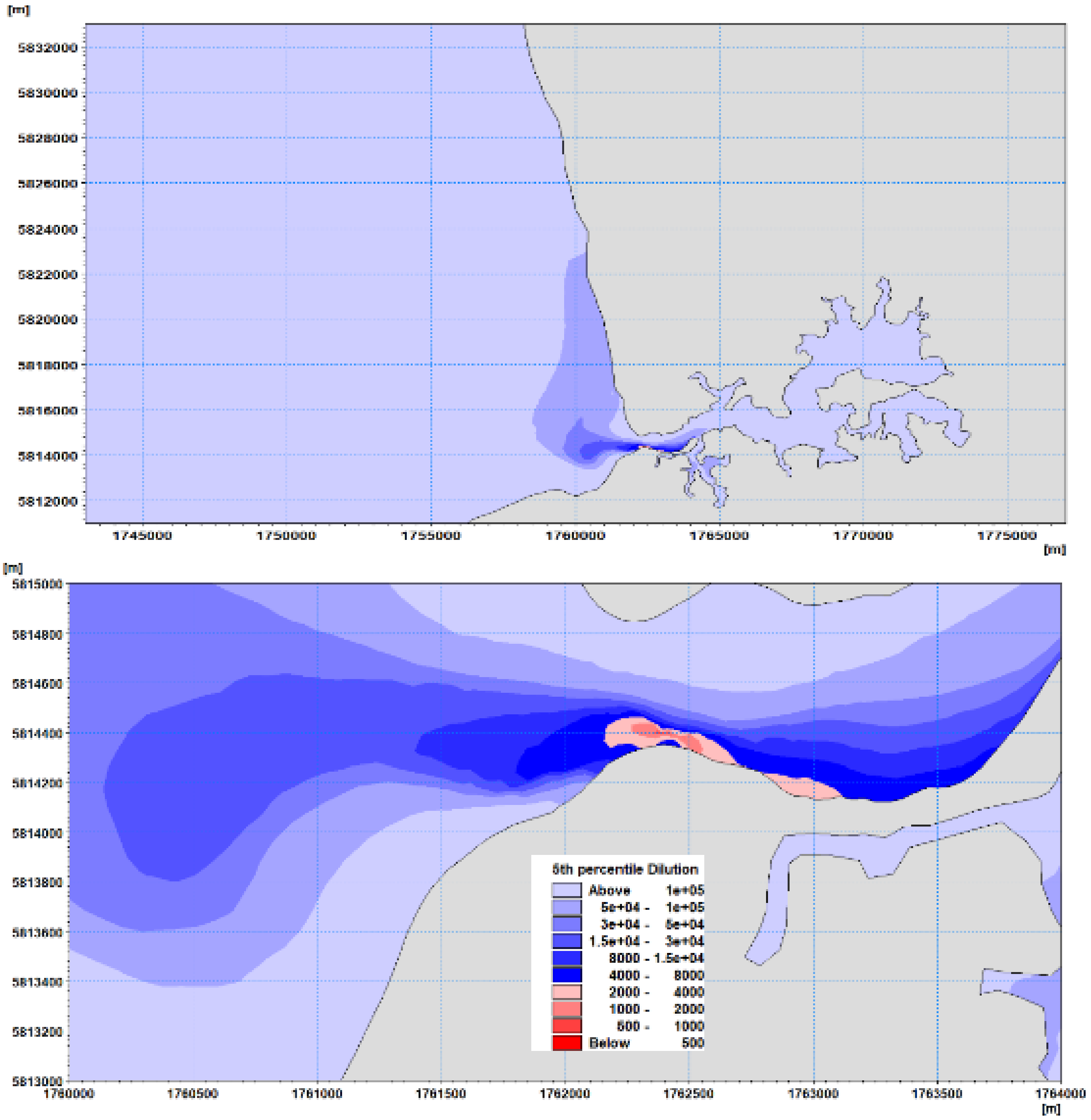


Figure 5-9. 5th percentile dilution from the 2018 model simulation of a conservative tracer under the existing discharge regime. Top panel shows the whole harbour and the area offshore of it and the bottom panel shows the area in the immediate vicinity of the existing discharge point.

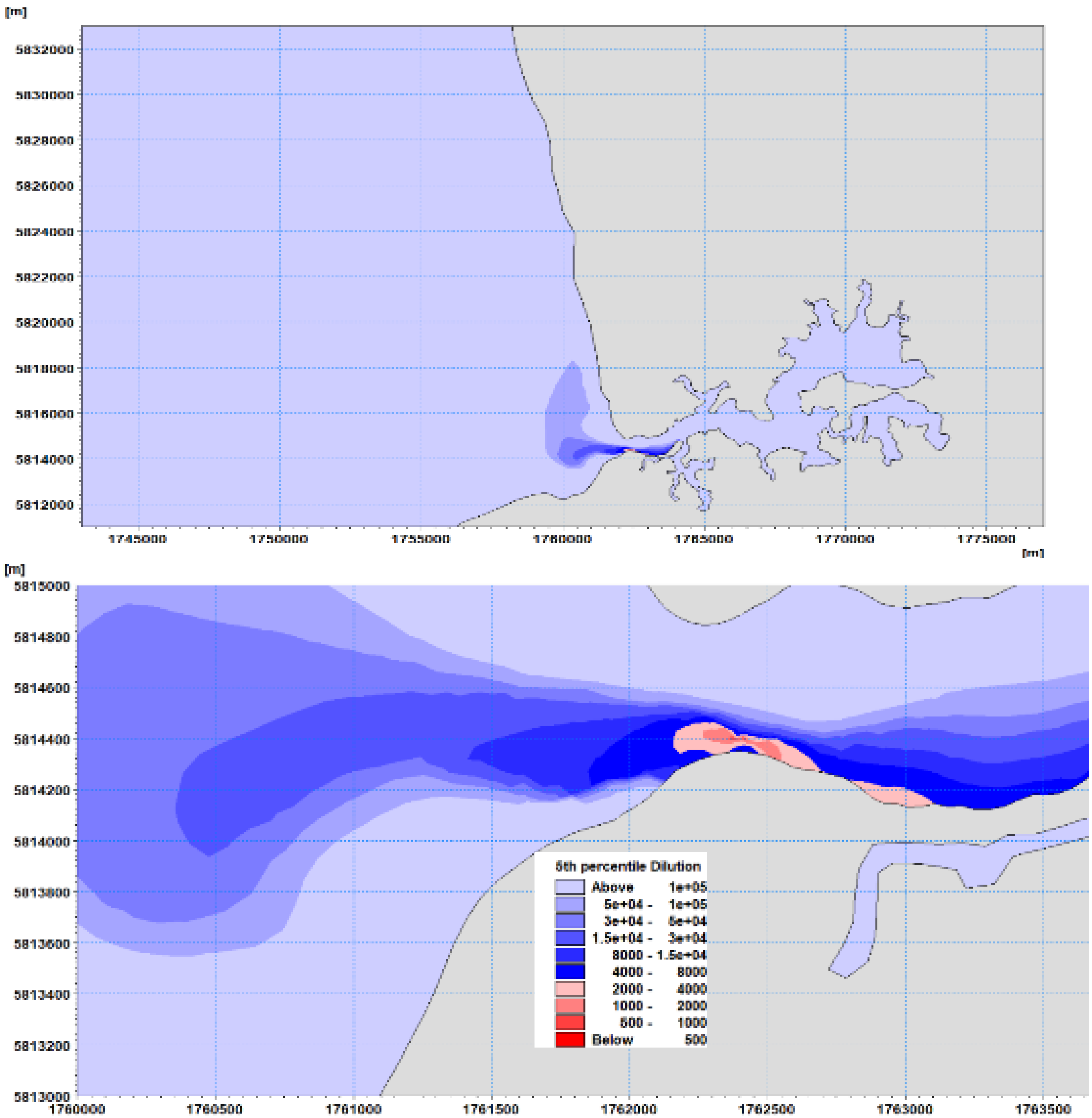


Figure 5-10. 5th percentile dilution from the 2018 model simulation of Enterococci under the existing discharge regime. Top panel shows the whole harbour and the area offshore of it and the bottom panel shows the area in the immediate vicinity of the existing discharge point.

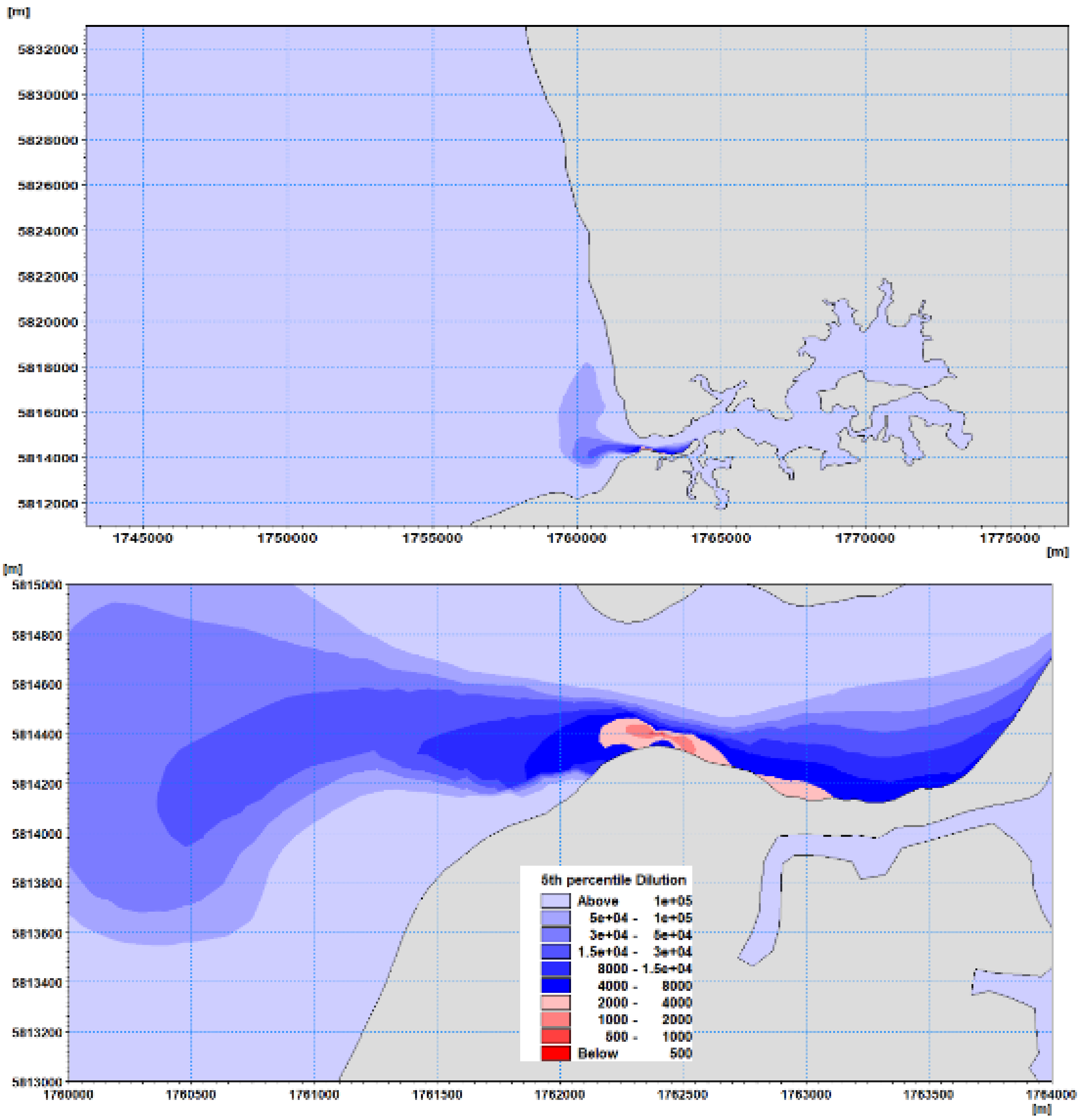


Figure 5-11. 5th percentile dilution from the 2018 model simulation of Viruses under the existing discharge regime. Top panel shows the whole harbour and the area offshore of it and the bottom panel shows the area in the immediate vicinity of the existing discharge point.

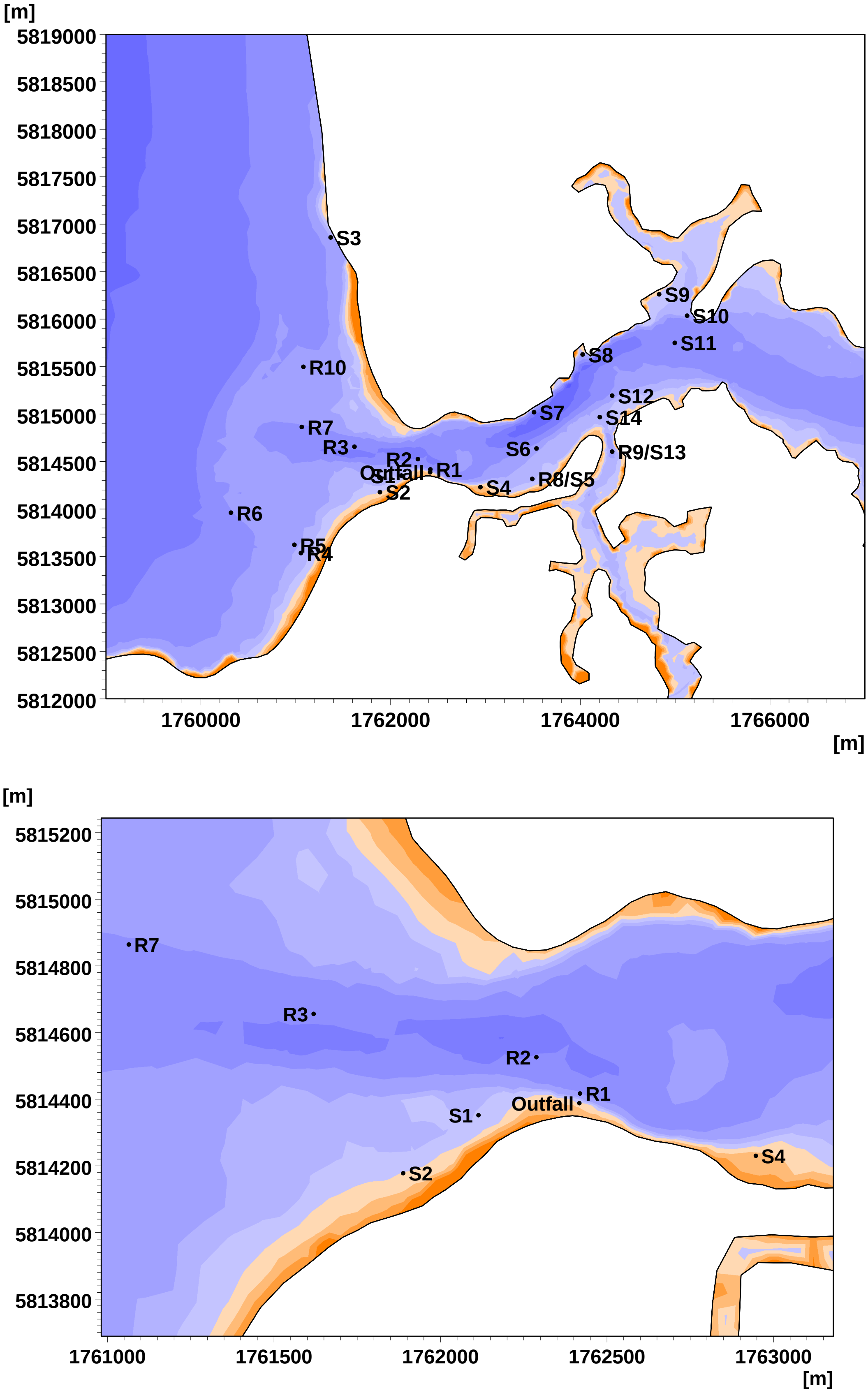


Figure 5-12. Sites where model data has been extracted for the Quantitative Microbial Risk Assessment process. Top panel shows all sites and the bottom panel shows the sites in the immediate vicinity of the outfall.

Table 5-4. Percentile estimates of dilution at the 25 QMRA sites for the conservative tracer and the current discharge regime.

| Site Reference | Site Description | 1th percentile | 5th percentile | 10th percentile |
|---|---|---|---|---|
| Outfall | Outfall | 183 | 294 | 386 |
| R1 | Kite Surf Inner | 196 | 604 | 863 |
| R2 | Inshore Kite surf | 2086 | 11138 | 17240 |
| R3 | Entrance kite surf | 12795 | 59531 | 98633 |
| R4 | Northern swimming | 45595 | 141221 | 325585 |
| R5 | Northern surfing | 42738 | 111734 | 249389 |
| R6 | Bar surf | 16077 | 26616 | 43842 |
| R7 | Offshore kite surf/Maui | 23947 | 63620 | 100880 |
| R8 (S5) | Western Swimming & Shellfish | 8019 | 13984 | 47648 |
| R9 (S13) | Domain Recreation/Shellfish | 63569 | 99986 | 156043 |
| R10 | Maui North | 49150 | 90404 | 128702 |
| S1 | Eastern end of Shellfish | 3537 | 5200 | 6690 |
| S2 | Mid-point of Shellfish | 5636 | 8337 | 66540 |
| S3 | Mussel Rocks | 61031 | 96666 | 125874 |
| S4 | Western Cockle/Pipi | 2660 | 3704 | 4757 |
| S5 (R8) | Western Swimming & Shellfish | 8019 | 13984 | 47648 |
| S6 | Western Shellfish in Harbour A | 25108 | 63058 | 108482 |
| S7 | Western Shellfish in Harbour B | 69668 | 111806 | 154464 |
| S8 | Mid Harbour Shellfish | 71007 | 112452 | 153089 |
| S9 | Inner Harbour Shellfish A | 62649 | 104441 | 150766 |
| S10 | Inner Harbour Shellfish B | 69368 | 110364 | 146271 |
| S11 | Inner Harbour Shellfish C | 70679 | 111576 | 148178 |
| S12 | Inner Harbour Shellfish D | 57086 | 100332 | 138408 |
| S13 (R9) | Domain Recreation/Shellfish | 63569 | 99986 | 156043 |
| S14 | Inner Harbour Shellfish | 39089 | 85628 | 127179 |

Table 5-5. Percentile estimates of dilution at the 25 QMRA sites for Enterococci and the current discharge regime.

| Site Reference | Site Description | 1th percentile | 5th percentile | 10th percentile |
|---|---|---|---|---|
| Outfall | Outfall | 184 | 295 | 387 |
| R1 | Kite Surf Inner | 196 | 605 | 866 |
| R2 | Inshore Kite surf | 2162 | 11608 | 18141 |
| R3 | Entrance kite surf | 13298 | 87441 | 171191 |
| R4 | Northern swimming | 73864 | 385487 | 1193947 |
| R5 | Northern surfing | 67257 | 261866 | 736094 |
| R6 | Bar surf | 18545 | 31128 | 57424 |
| R7 | Offshore kite surf/Maui | 25373 | 96542 | 182935 |
| R8 (S5) | Western Swimming & Shellfish | 8583 | 15630 | 64404 |
| R9 (S13) | Domain Recreation/Shellfish | 147126 | 416721 | 779250 |
| R10 | Maui North | 64811 | 160456 | 275087 |
| S1 | Eastern end of Shellfish | 3633 | 5319 | 6848 |
| S2 | Mid-point of Shellfish | 5833 | 8534 | 99906 |
| S3 | Mussel Rocks | 85760 | 161551 | 233914 |
| S4 | Western Cockle/Pipi | 2748 | 3852 | 4942 |
| S5 (R8) | Western Swimming & Shellfish | 8583 | 15630 | 64404 |
| S6 | Western Shellfish in Harbour A | 27178 | 96509 | 316212 |
| S7 | Western Shellfish in Harbour B | 162806 | 389783 | 686534 |
| S8 | Mid Harbour Shellfish | 166189 | 415735 | 749243 |
| S9 | Inner Harbour Shellfish A | 140595 | 297456 | 507581 |
| S10 | Inner Harbour Shellfish B | 168076 | 410163 | 770986 |
| S11 | Inner Harbour Shellfish C | 163812 | 434352 | 807735 |
| S12 | Inner Harbour Shellfish D | 90973 | 293705 | 592205 |
| S13 (R9) | Domain Recreation/Shellfish | 147126 | 416721 | 779250 |
| S14 | Inner Harbour Shellfish | 47168 | 211757 | 445611 |

Table 5-6. Percentile estimates of dilution at the 25 QMRA sites for Viruses and the current discharge regime.

| Site Reference | Site Description | 1th percentile | 5th percentile | 10th percentile |
|---|---|---|---|---|
| Outfall | Outfall | 184 | 295 | 387 |
| R1 | Kite Surf Inner | 196 | 605 | 866 |
| R2 | Inshore Kite surf | 2149 | 11617 | 18168 |
| R3 | Entrance kite surf | 13405 | 86585 | 170669 |
| R4 | Northern swimming | 74657 | 375158 | 1048023 |
| R5 | Northern surfing | 66767 | 259317 | 674337 |
| R6 | Bar surf | 18884 | 31437 | 58249 |
| R7 | Offshore kite surf/Maui | 25174 | 94937 | 183049 |
| R8 (S5) | Western Swimming & Shellfish | 8626 | 15399 | 64683 |
| R9 (S13) | Domain Recreation/Shellfish | 147252 | 391367 | 678484 |
| R10 | Maui North | 68594 | 153860 | 264009 |
| S1 | Eastern end of Shellfish | 3632 | 5320 | 6853 |
| S2 | Mid-point of Shellfish | 5843 | 8524 | 101145 |
| S3 | Mussel Rocks | 87255 | 155886 | 226113 |
| S4 | Western Cockle/Pipi | 2724 | 3834 | 4942 |
| S5 (R8) | Western Swimming & Shellfish | 8626 | 15399 | 64683 |
| S6 | Western Shellfish in Harbour A | 27217 | 95135 | 290846 |
| S7 | Western Shellfish in Harbour B | 155989 | 352424 | 607676 |
| S8 | Mid Harbour Shellfish | 162566 | 370037 | 660345 |
| S9 | Inner Harbour Shellfish A | 133230 | 275912 | 457369 |
| S10 | Inner Harbour Shellfish B | 158916 | 368640 | 668347 |
| S11 | Inner Harbour Shellfish C | 162039 | 386176 | 714967 |
| S12 | Inner Harbour Shellfish D | 92639 | 272610 | 528927 |
| S13 (R9) | Domain Recreation/Shellfish | 147252 | 391367 | 678484 |
| S14 | Inner Harbour Shellfish | 47169 | 209302 | 417285 |

### 5.5.3 Dilution Maps and Summary of Predictions at QMRA sites - Future Discharge Rate and Existing Discharge Regime

For the future discharge rate simulation, the minimum dilution achieved 150 m from the existing outfall (i.e. the nominal mixing zone) is 40-fold and the 1$^{th}$, 5$^{th}$ and 10$^{th}$ percentile values are 95, 225 and 360 respectively. These values reflect the increase in discharge rate under the future discharge regime (1175 $m^3$/day currently increasing to 2335 $m^3$/day).

Table 5-7 shows the predicted percentile estimates at the 25 QMRA sites for the conservative tracer (no decay).

As discussed in Section 5.3, Table 5-3, extending the outfall by 60m provides around three times the level of initial dilution and provides around the same increase in dilution as fitting a diffuser to the existing outfall. Thus, there is scope with these options to increase the level of dilution achieved along the shoreline immediately inshore of the outfall and within the immediate vicinity of the existing discharge point.

This would increase the level of dilution achieved at sites S1, R1, R2 and Outfall compared to the values presented in Table 5-7 (highlighted) but have little influence on the predicted dilutions elsewhere.

At all other sites except sites R3, S2 and S5, the 1th percentile dilution is greater than 10,000 which results in the lowest predicted levels of public health risk (NIWA, 2019 – Figures 4-5 and 4-10).

The dilution achieved at sites inside the harbour (e.g. S4, S5) could be increased with optimisation of the timing so that the discharge occurs more around mid-tide rather than before or at high water. However, doing this may decrease the level of dilution achieved at sites S2 and R3.but, as discussed in the QMRA report (NIWA, 2019), the risk of infection and illness is generally low at all sites considered and, given dilutions achieved at these two sites are much higher than those achieved closer to the outfall, any decrease in dilution achieved at these sites due to any optimisation of the timing is likely to results in very low level of risk.

Table 5-7. Percentile estimates of dilution at the 25 QMRA sites for the conservative tracer and the current discharge regime. Highlighted cells are the QMRA sites where relocation or fitting of a diffuser could provide improvements to dilution.

| Site Reference | Site Description | 1th percentile | 5th percentile | 10th percentile |
|---|---|---|---|---|
| Outfall | Outfall | 96 | 154 | 202 |
| R1 | Kite Surf Inner | 99 | 315 | 451 |
| R2 | Inshore Kite surf | 1041 | 5809 | 8999 |
| R3 | Entrance kite surf | 6485 | 30974 | 51435 |
| R4 | Northern swimming | 23798 | 73677 | 170001 |
| R5 | Northern surfing | 22297 | 58240 | 130378 |
| R6 | Bar surf | 8386 | 13865 | 22882 |
| R7 | Offshore kite surf/Maui | 12412 | 33186 | 52636 |
| R8 (S5) | Western Swimming & Shellfish | 4185 | 7291 | 24837 |
| R9 (S13) | Domain Recreation/Shellfish | 33128 | 52151 | 81520 |
| R10 | Maui North | 25453 | 47135 | 66955 |
| S1 | Eastern end of Shellfish | 1846 | 2712 | 3491 |
| S2 | Mid-point of Shellfish | 2942 | 4351 | 34717 |
| S3 | Mussel Rocks | 31640 | 50407 | 65542 |
| S4 | Western Cockle/Pipi | 1388 | 1931 | 2482 |
| S5 (R8) | Western Swimming & Shellfish | 4185 | 7291 | 24837 |
| S6 | Western Shellfish in Harbour A | 13097 | 32915 | 56569 |
| S7 | Western Shellfish in Harbour B | 36351 | 58320 | 80578 |
| S8 | Mid Harbour Shellfish | 37046 | 58684 | 79834 |
| S9 | Inner Harbour Shellfish A | 32721 | 54494 | 78602 |
| S10 | Inner Harbour Shellfish B | 36209 | 57580 | 76315 |
| S11 | Inner Harbour Shellfish C | 36818 | 58139 | 77347 |
| S12 | Inner Harbour Shellfish D | 29766 | 52322 | 72232 |
| S13 (R9) | Domain Recreation/Shellfish | 33128 | 52151 | 81520 |
| S14 | Inner Harbour Shellfish | 20388 | 44653 | 66348 |

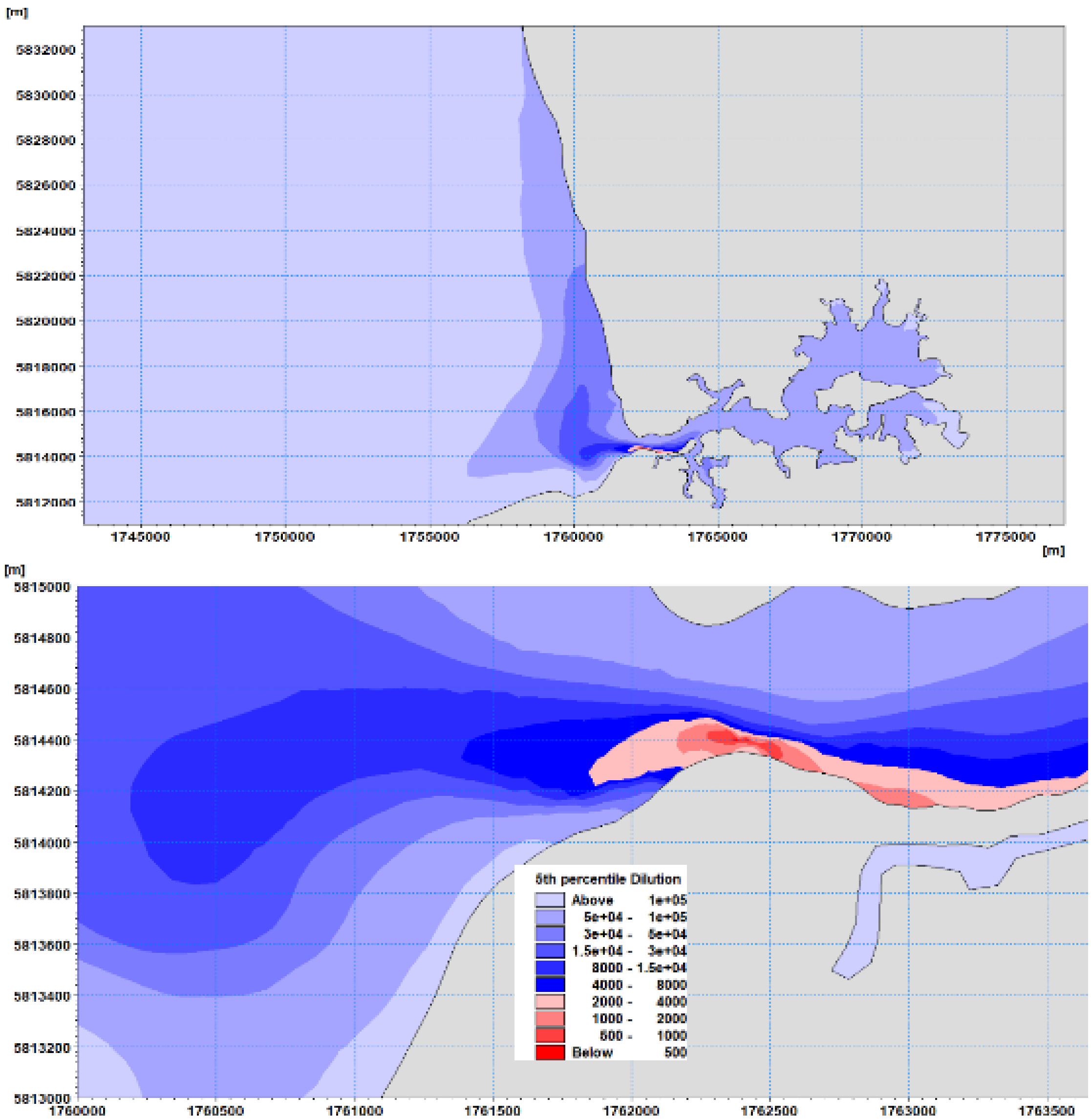


Figure 5-13. 5th percentile dilution from the 2018 model simulation of a conservative tracer under a future discharge rate and the current discharge timing. Top panel shows the whole harbour and the area offshore of it and the bottom panel shows the area in the immediate vicinity of the existing discharge point.

# 6 Summary

This report provides details of the calibration of a hydrodynamic model which has been used to assess the discharge of treated wastewater to Raglan Harbour. The discharge is via an outfall near the entrance to the harbour and is timed to occur mostly on the outgoing tide.

Overall the models used to assess the current discharge to the harbour provide good predictions of water level variations, currents and waves in the harbour and in the immediate vicinity of the current discharge.

Currents near the entrance to Raglan Harbour are relatively strong and often exceed 1.0 m/s. Combined with the depths immediately offshore of the existing outfall, these currents result in a significant level of dilution when the treated wastewater moves offshore of the existing discharge location.

Significant wave height offshore of the entrance can exceed 5m but at the existing outfall location significant wave heights rarely exceed 0.3m.

Near field modelling indicates that the combination of water depths and ambient currents at the current outfall location provide a relatively good level of initial dilution. Within 100m of the outfall, dilutions in excess of 50 occur and, because of the staging of the discharge, near-field dilutions are often much higher than this.

An potential option of extending the existing outfall by 60m provides around three times the level of initial dilution and provides around the same increase in dilution as fitting a diffuser to the existing outfall. This would increase the level of dilution achieved along the shoreline immediately inshore of the outfall and within a few hundred metres of existing discharge point.

Relocating the outfall further offshore of the existing site would provide an opportunity for further increases in initial dilution.

Far field model results show that beyond the first few hundred metres of the outfall there is a significant level of dilution achieved.

At sites closest the existing outfall dilutions of greater than 200 occur for 99 percent of the time. At sites in the immediate vicinity of the outfall dilutions of greater than 2000 occur for 99 percent. Offshore of the harbour dilutions of greater than 12000 occur for 99 percent of the time while within the harbour itself dilutions of greater than 8000 occur for 99 percent of the time with much higher dilutions occurring towards the middle of the harbour compared to sites nearer the entrance.

Offshore of the harbour, these results reflect the fact that the treated wastewater plume becoming fully mixed in water column within the first few hundred metres of the discharge point and then being subsequently transported and dispersed by the relatively strong currents that occur outside the entrance to the harbour.

When the discharge occurs near high water (at the start of the consented discharge window), lower levels of dilution do occasionally occur inside the harbour and highest levels are observed within the first few hundred metres inshore of the discharge point. When the treated wastewater plume is transported back into the harbour on the incoming tide following a discharge, very high levels of dilution are achieved inside the harbour.

There are opportunities for offsetting the future increases in discharge rate due to population growth through a combination of fitting a diffuser to the existing outfall, relocating the outfall further offshore of the existing site (into deeper water) or optimising the timing of the discharge so that it occurs more around mid-tide (when currents are at their strongest).

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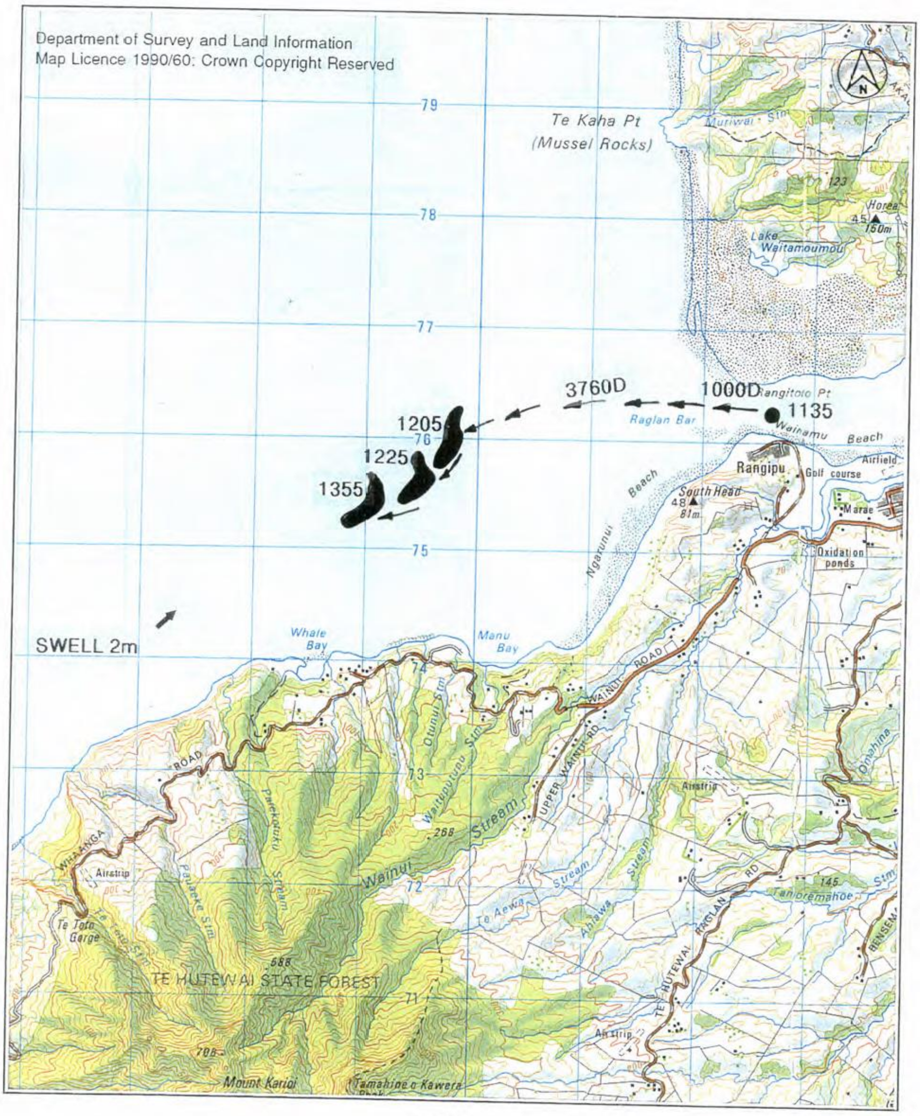
# Appendix A - NIWA Dye test results



**Figure 5**

Dye track from the first release on 31 July 1996.
Arrows indicate the line of the of the dye movement out of the entrance channel to the position where it pooled off Ngarunui Beach and Manu Bay. Numbers are times (2400 h clock). Numbers followed by D are dilution estimates at the points marked. Grid scale is 1 km.



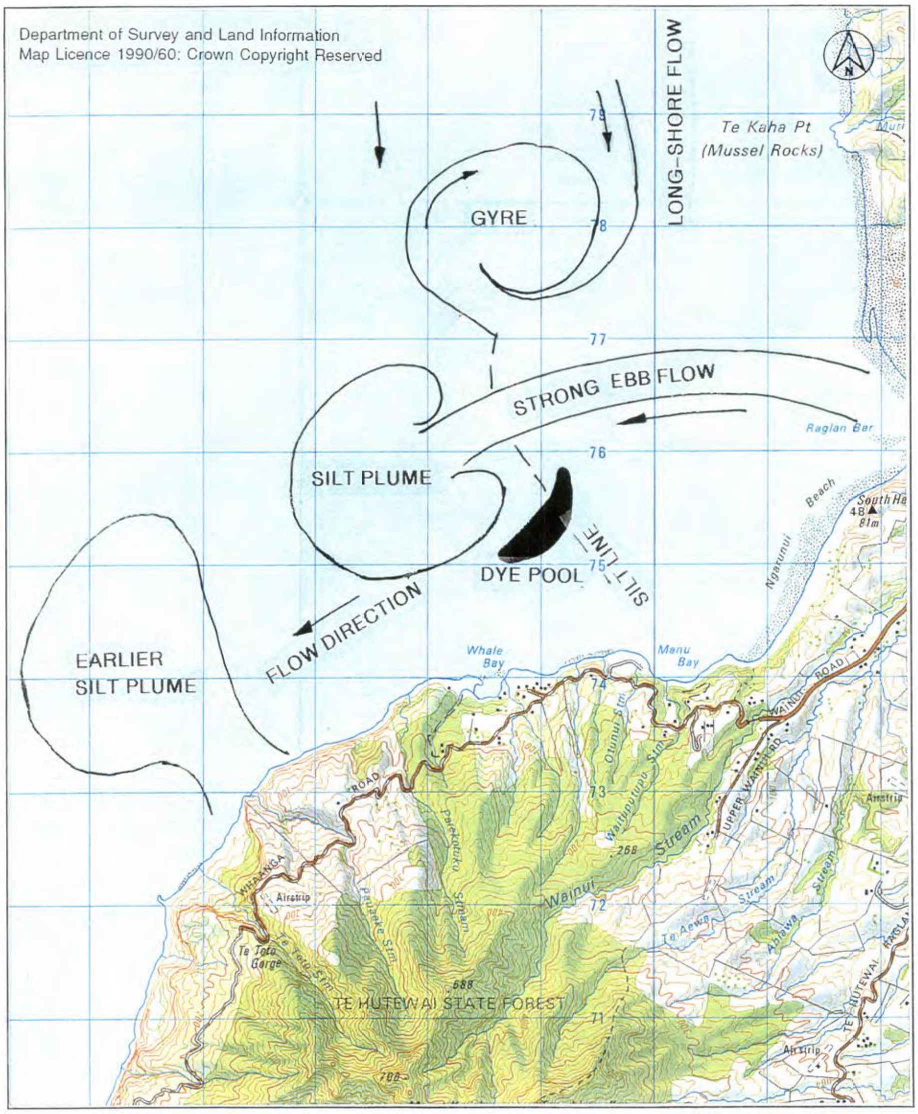

NIWA

Figure 7

Silt patterns indicating flow across the long-shore current which was moving from north to south across the harbour entrance. A large gyre (eddy) formedon the northern side of the harbour discharge caused by the long-shore current.

Further south of the immediate silt plume was evidence of another silt plume presumably from the previous ebb tidal discharge.

The pooling dye from the first release 3 hours earlier is marked. This was slowly being diluted as it moved south.

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# Appendix B - CORMIX results

Table B-1. CORMIX schematic scenarios

| Case | Water depth (m) | Ambient current (m/s) | Description |
|---|---|---|---|
| A | 3.3 | 0.20 | Conditions towards the end of the discharge window (Neap tide) |
| B | 2.7 | 0.40 | Conditions towards the end of the discharge window (Mean tide) |
| C | 2.3 | 0.60 | Conditions towards the end of the discharge window (Spring tide) |
| D | 4.3 | 0.10 | Just after high tide (Neap tide) |
| E | 5.5 | 0.10 | Just after high tide (Spring tide) |
| F | 4.3 | 0.50 | Early ebb tide (Spring tide) |
| G | 3.8 | 0.50 | Early ebb tide (Mean tide) |
| H | 3.4 | 0.50 | Early ebb tide (Neap tide) |

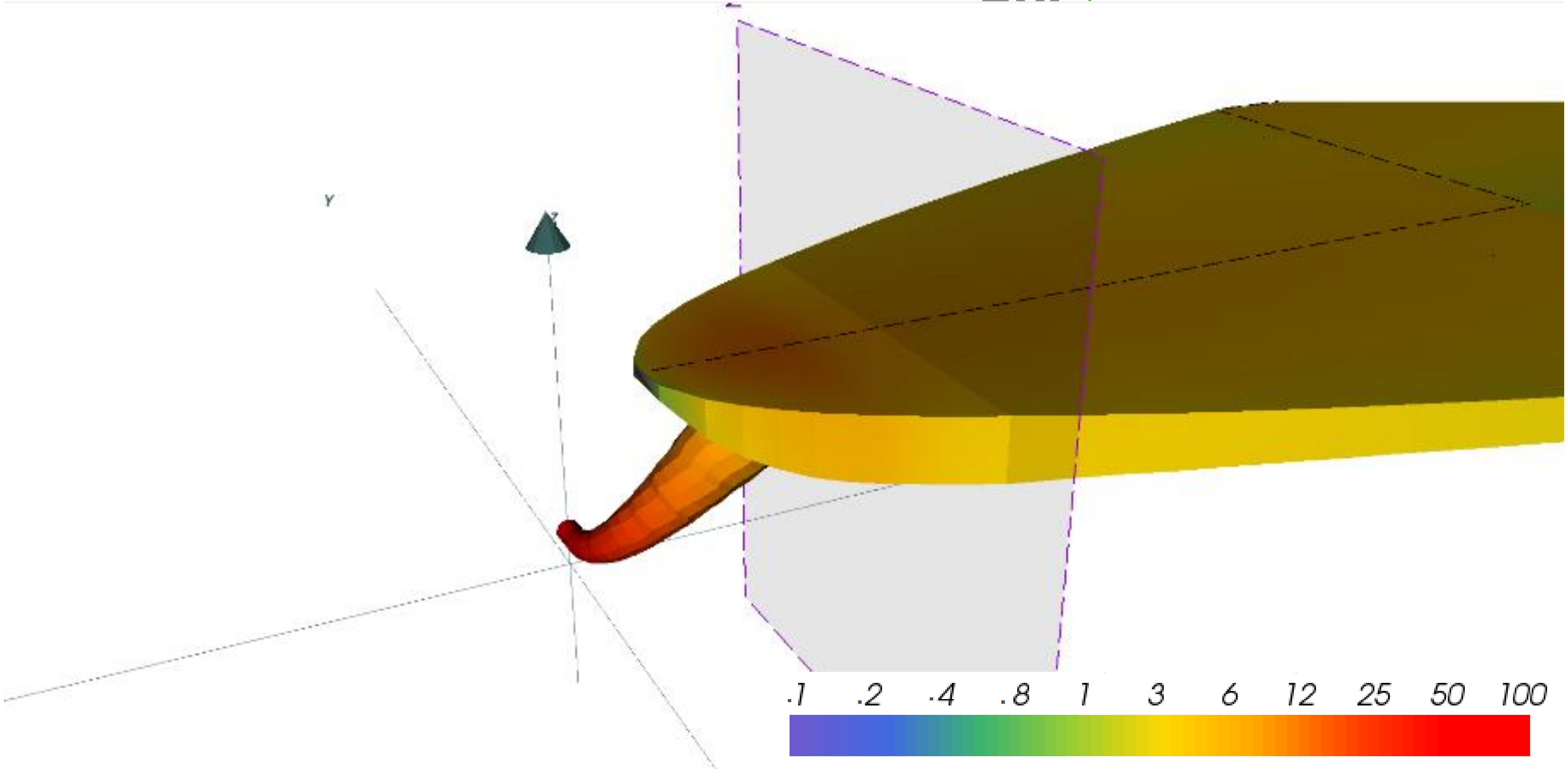


Figure B-1. Dynamics of the treated wastewater plume for Case A ambient conditions – representing conditions towards the end of the discharge window during a neap tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

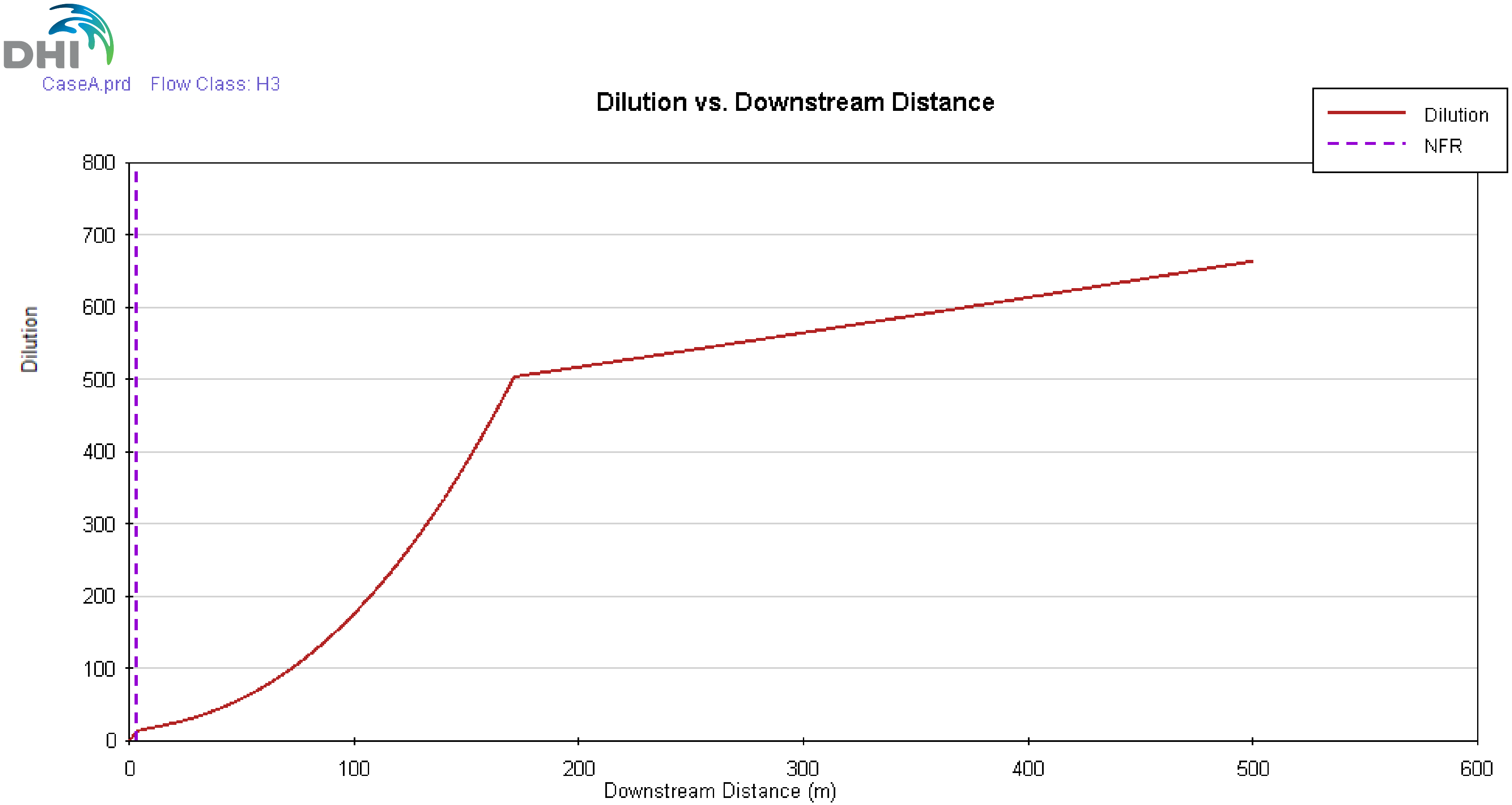


Figure B-2. Dilution versus distance from discharge point (m) in the far field and for Case A ambient conditions – representing conditions towards the end of the discharge window during a neap tide.

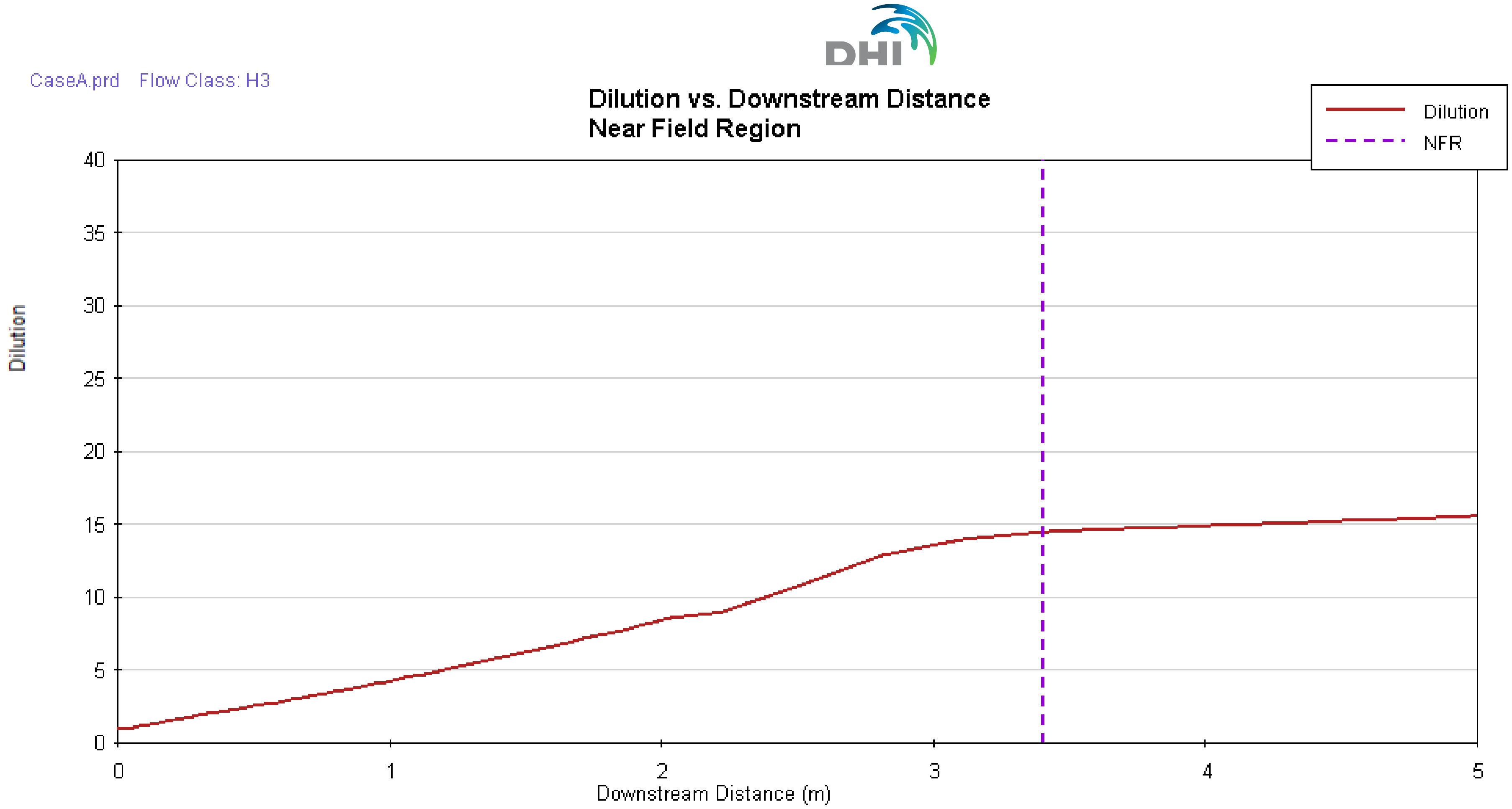


Figure B-3. Dilution versus distance from discharge point (m) in the near field for Case A ambient conditions – representing conditions towards the end of the discharge window during a neap tide.

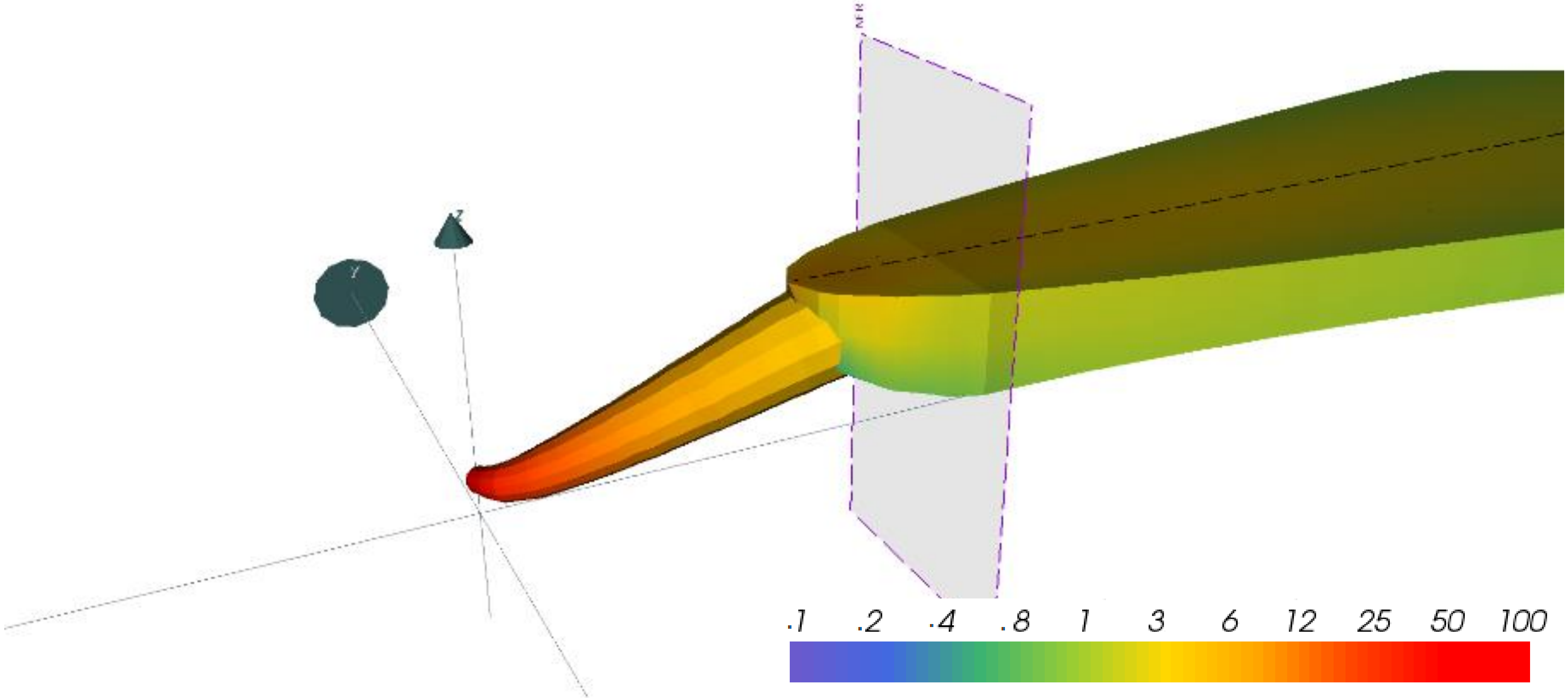


Figure B-4. Dynamics of the treated wastewater plume for Case B ambient conditions – representing conditions towards the end of the discharge window during a mean tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

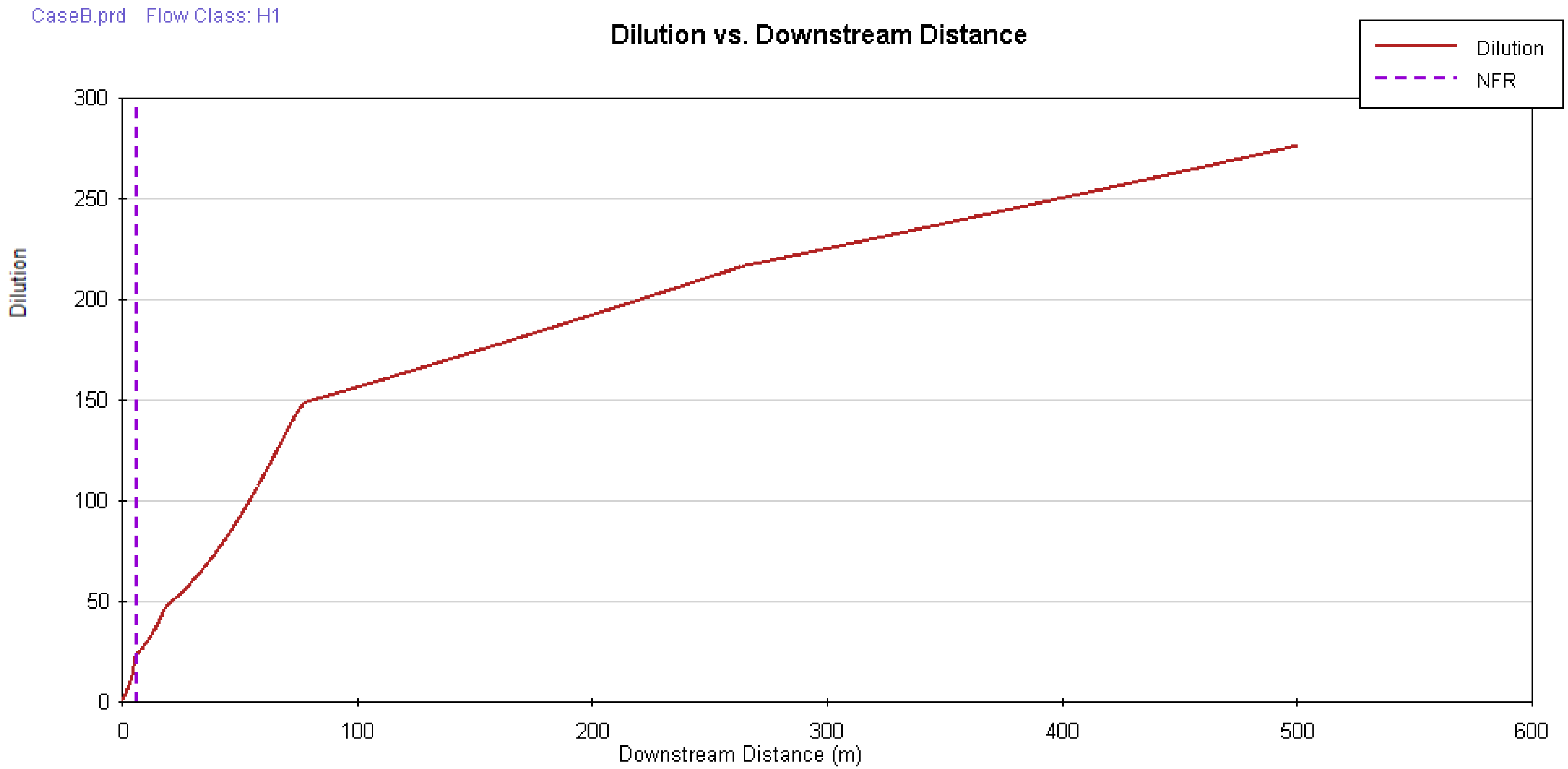


Figure B-5. Dilution versus distance from discharge point (m) in the far field and for Case B ambient conditions – representing conditions towards the end of the discharge window during a mean tide.

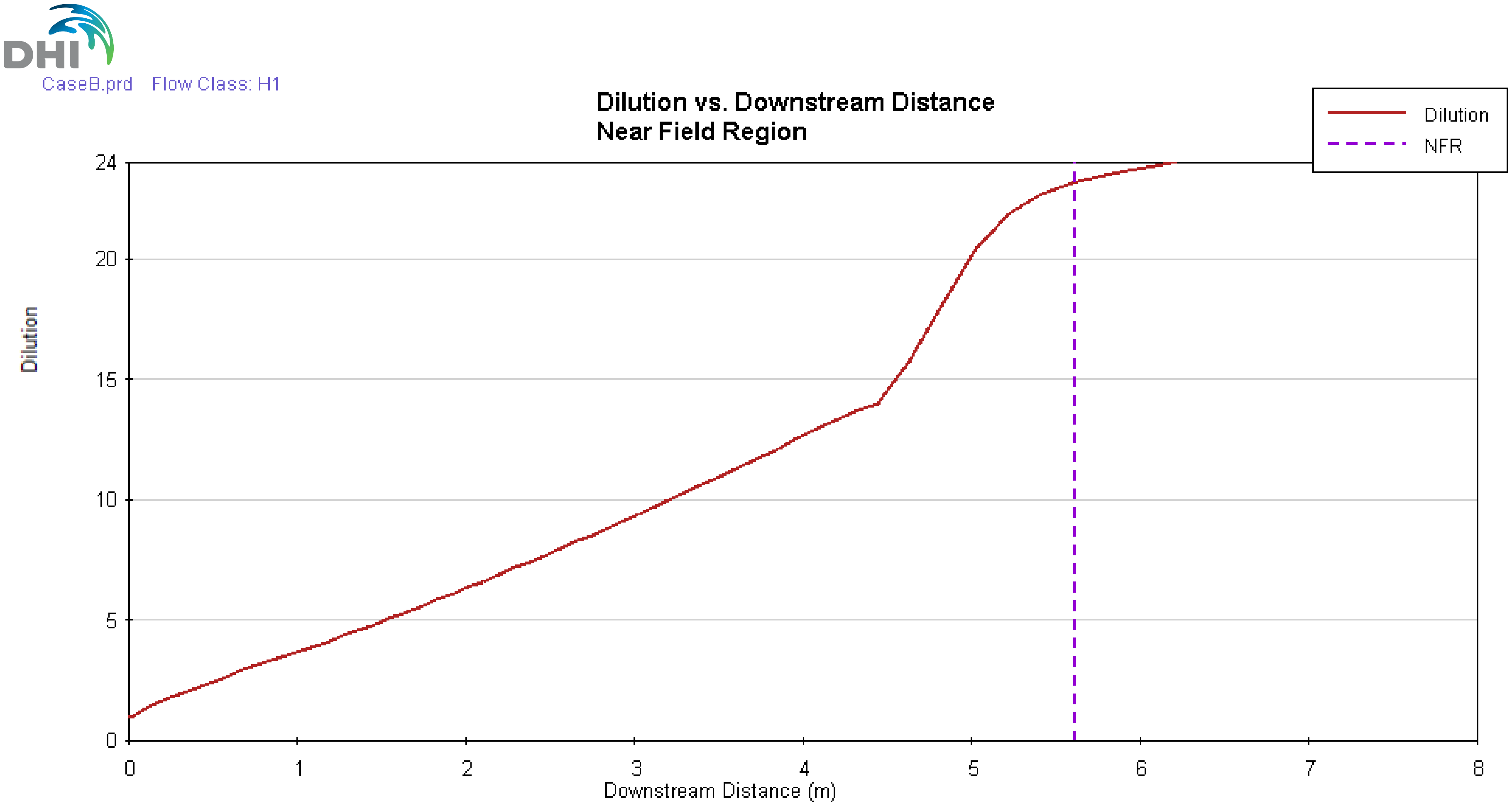


Figure B-6. Dilution versus distance from discharge point (m) in the near field and for Case B ambient conditions – representing conditions towards the end of the discharge window during a neap tide.

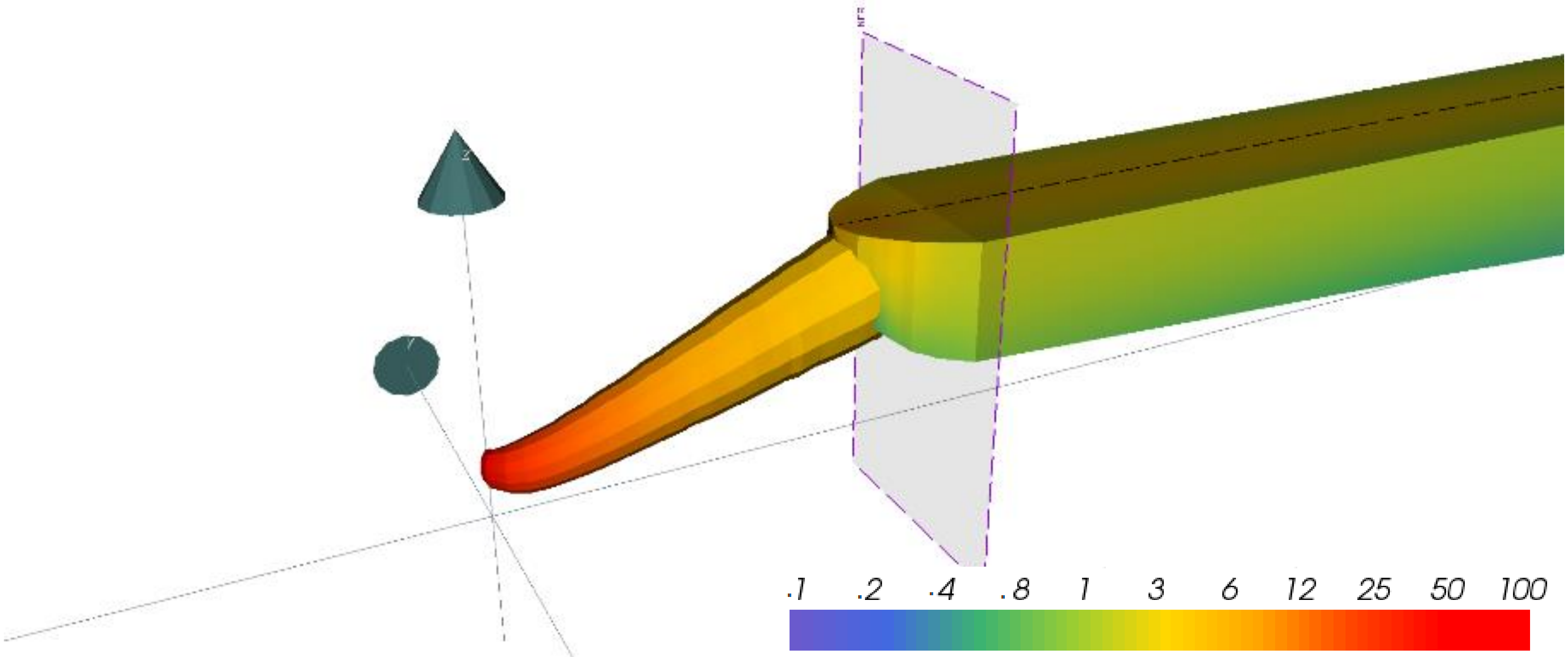


Figure B-7. Dynamics of the treated wastewater plume for Case C ambient conditions – representing conditions towards the end of the discharge window during a spring tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

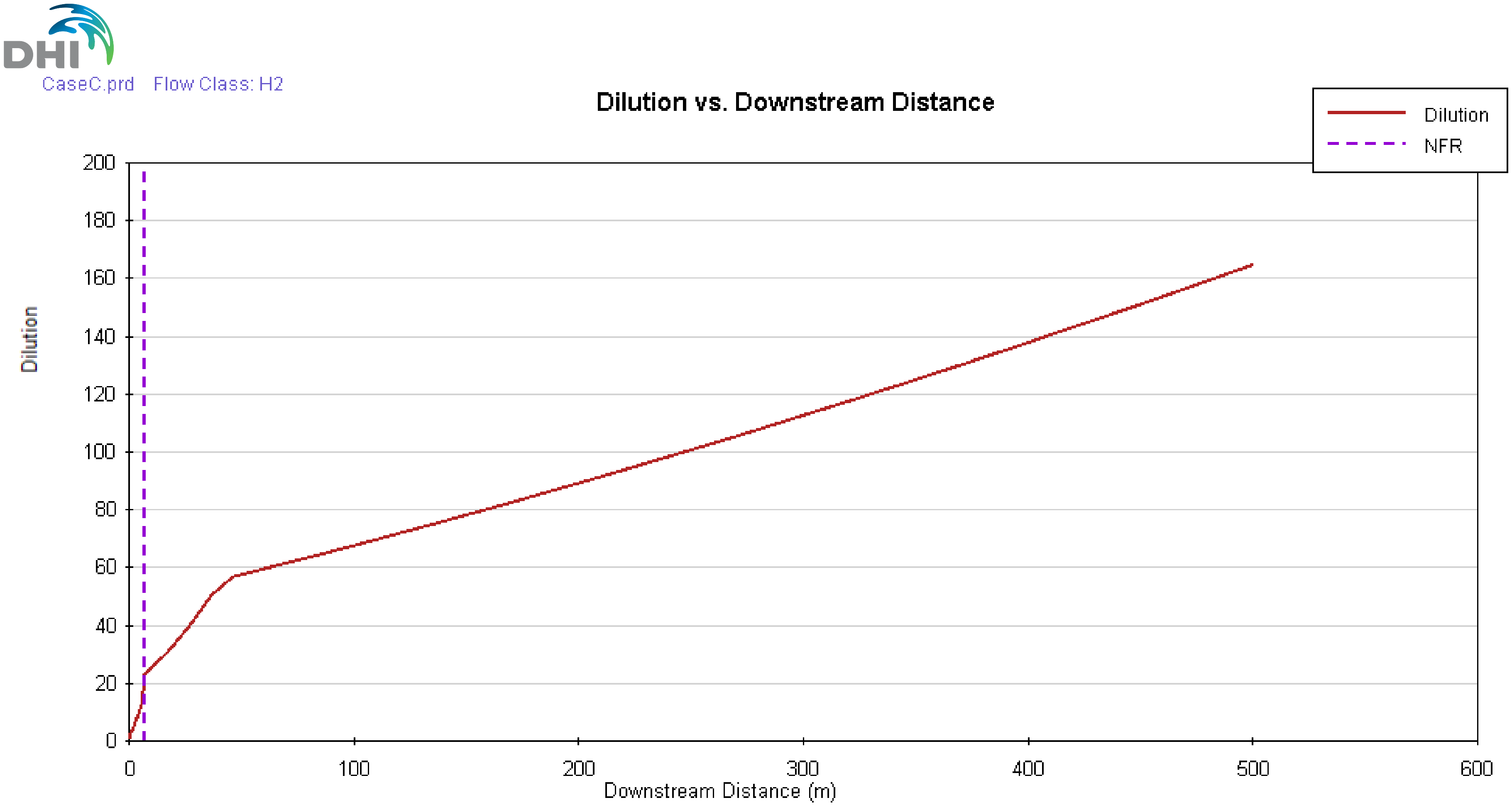


Figure B-8. Dilution versus distance from discharge point (m) in the far field and for Case C ambient conditions – representing conditions towards the end of the discharge window during a spring tide.

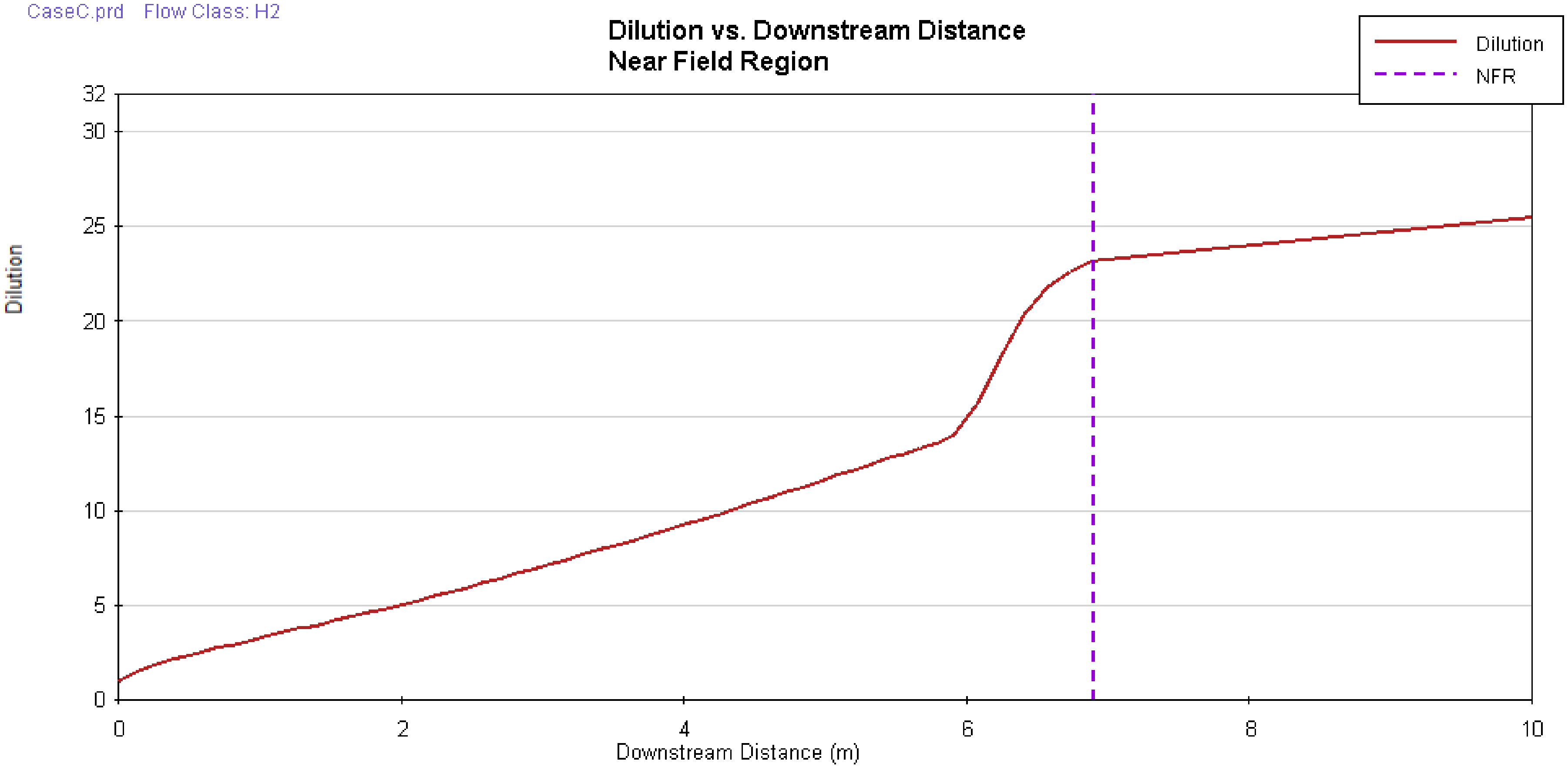


Figure B-9. Dilution versus distance from discharge point (m) in the near field and for Case C ambient conditions – representing conditions towards the end of the discharge window during a spring tide.

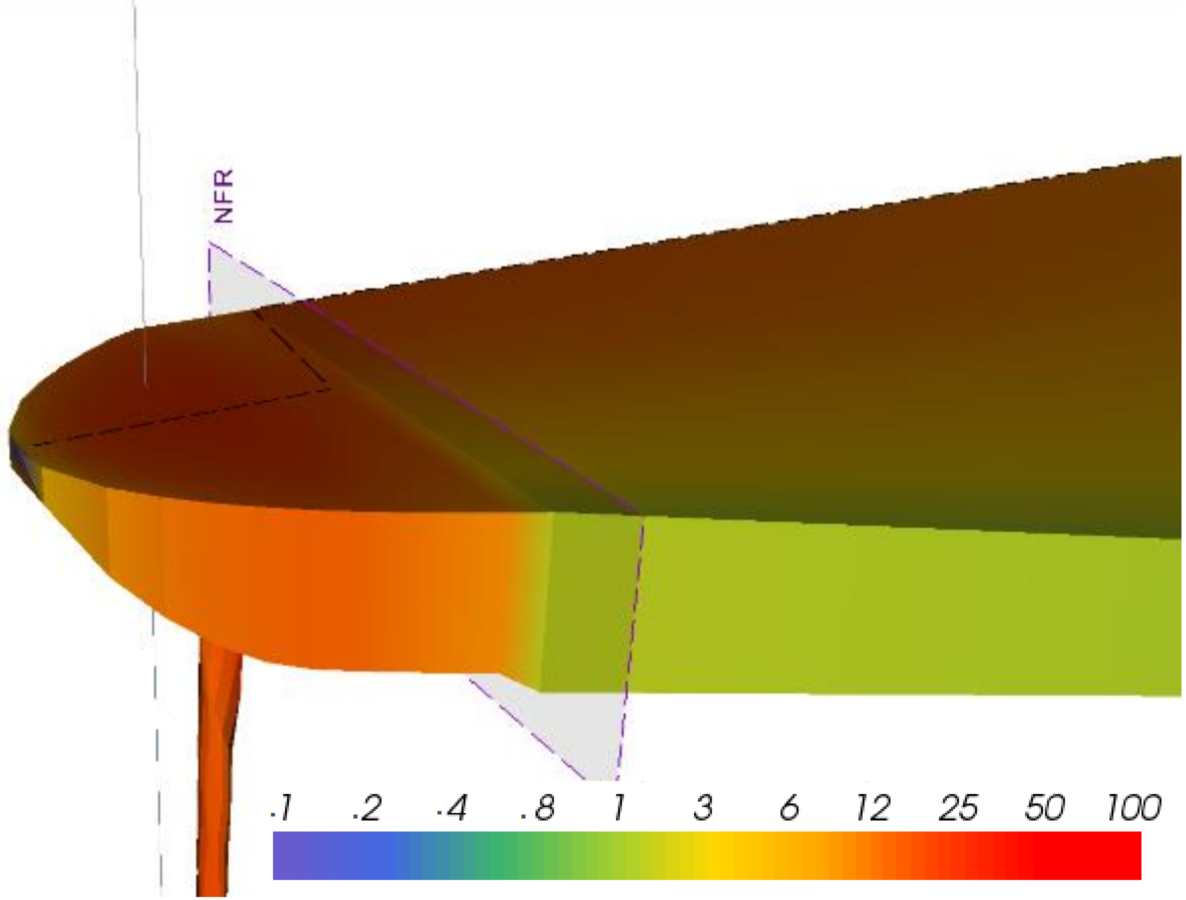


Figure B-10. Dynamics of the treated wastewater plume for Case D ambient conditions – representing conditions just after a neap high tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

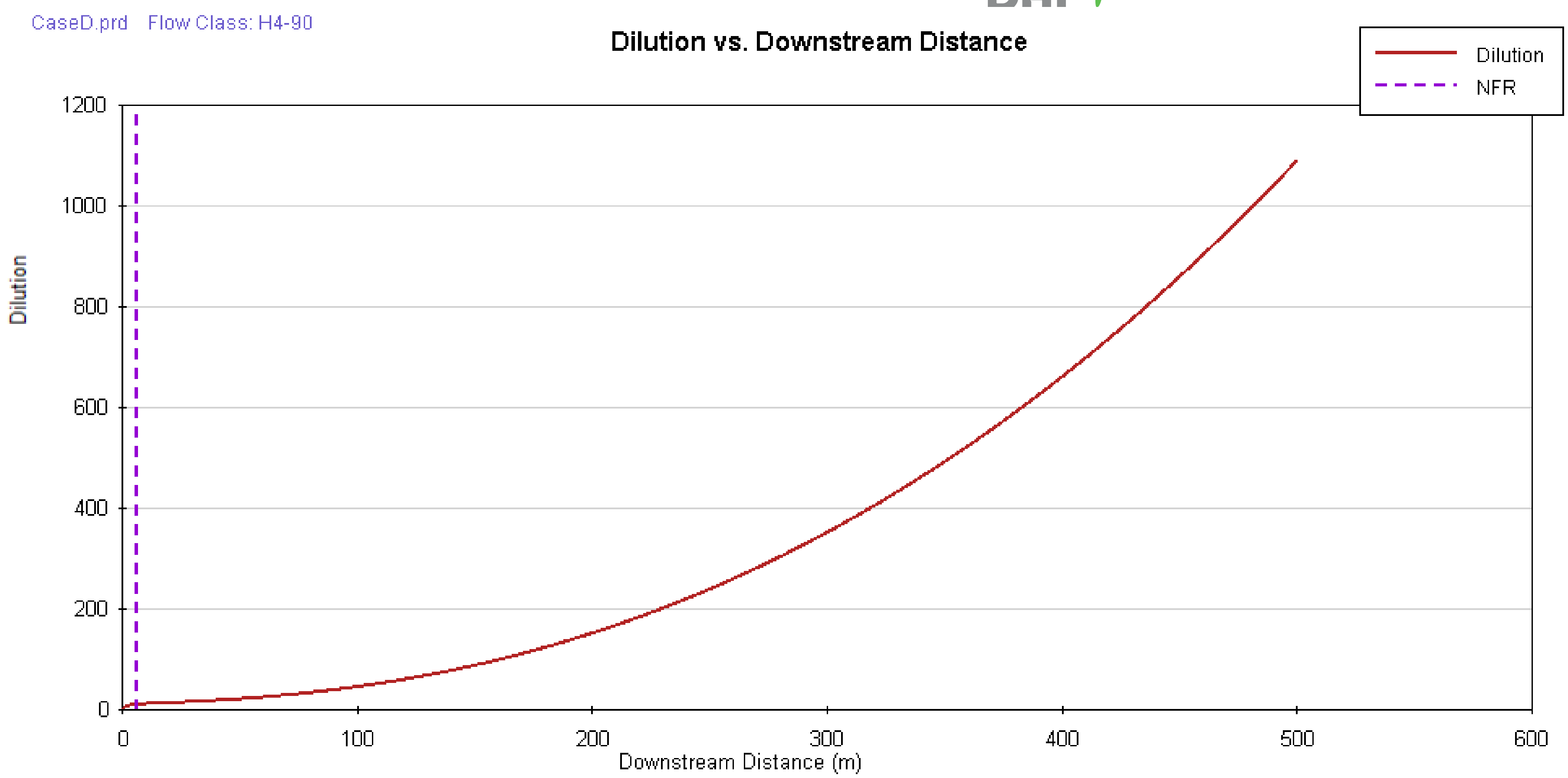


Figure B-11. Dilution versus distance from discharge point (m) in the far field and for Case D ambient conditions – representing conditions just after a neap high tide.

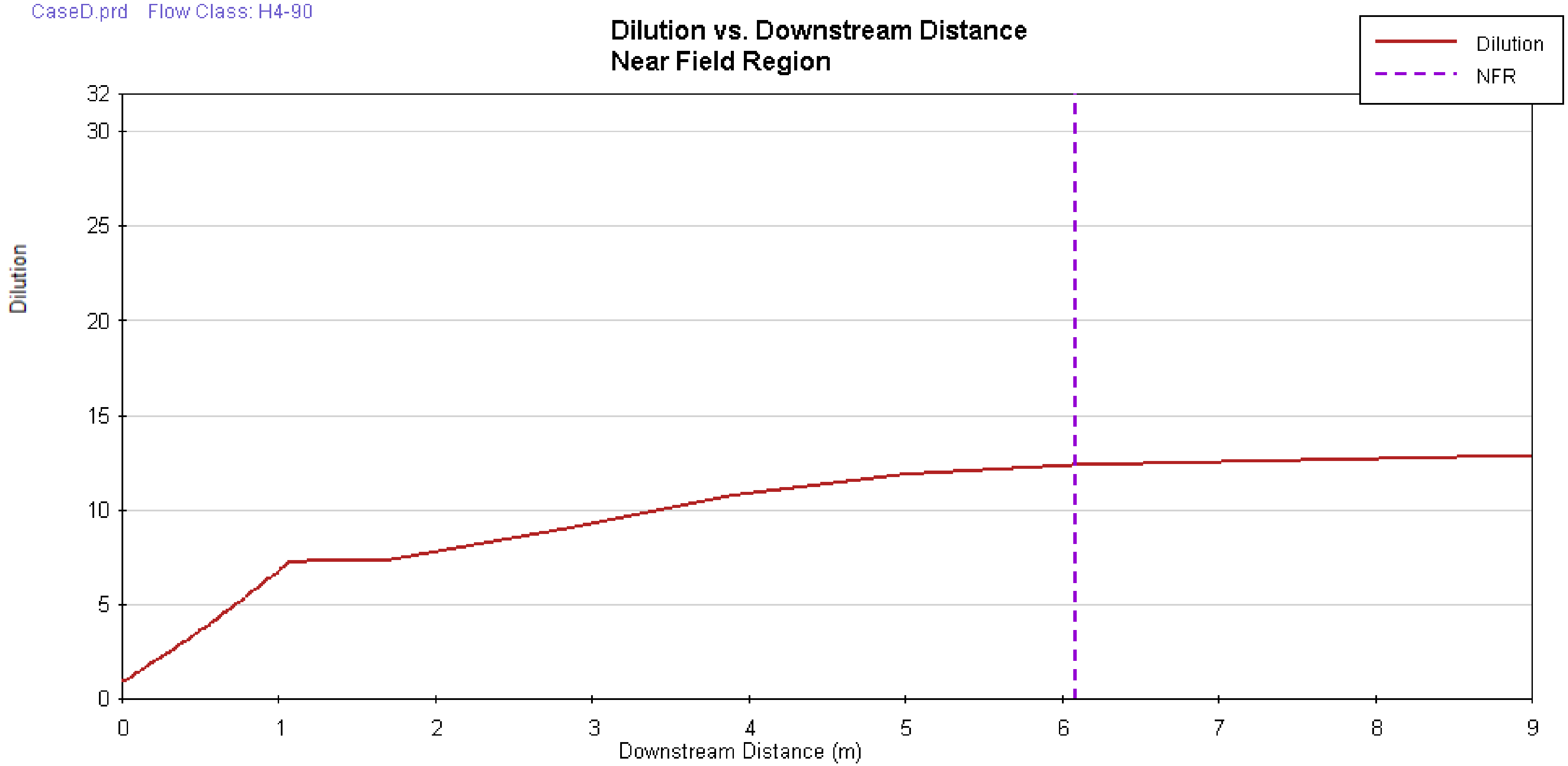


Figure B-12. Dilution versus distance from discharge point (m) in the near field and for Case D ambient conditions – representing conditions just after a neap high tide.

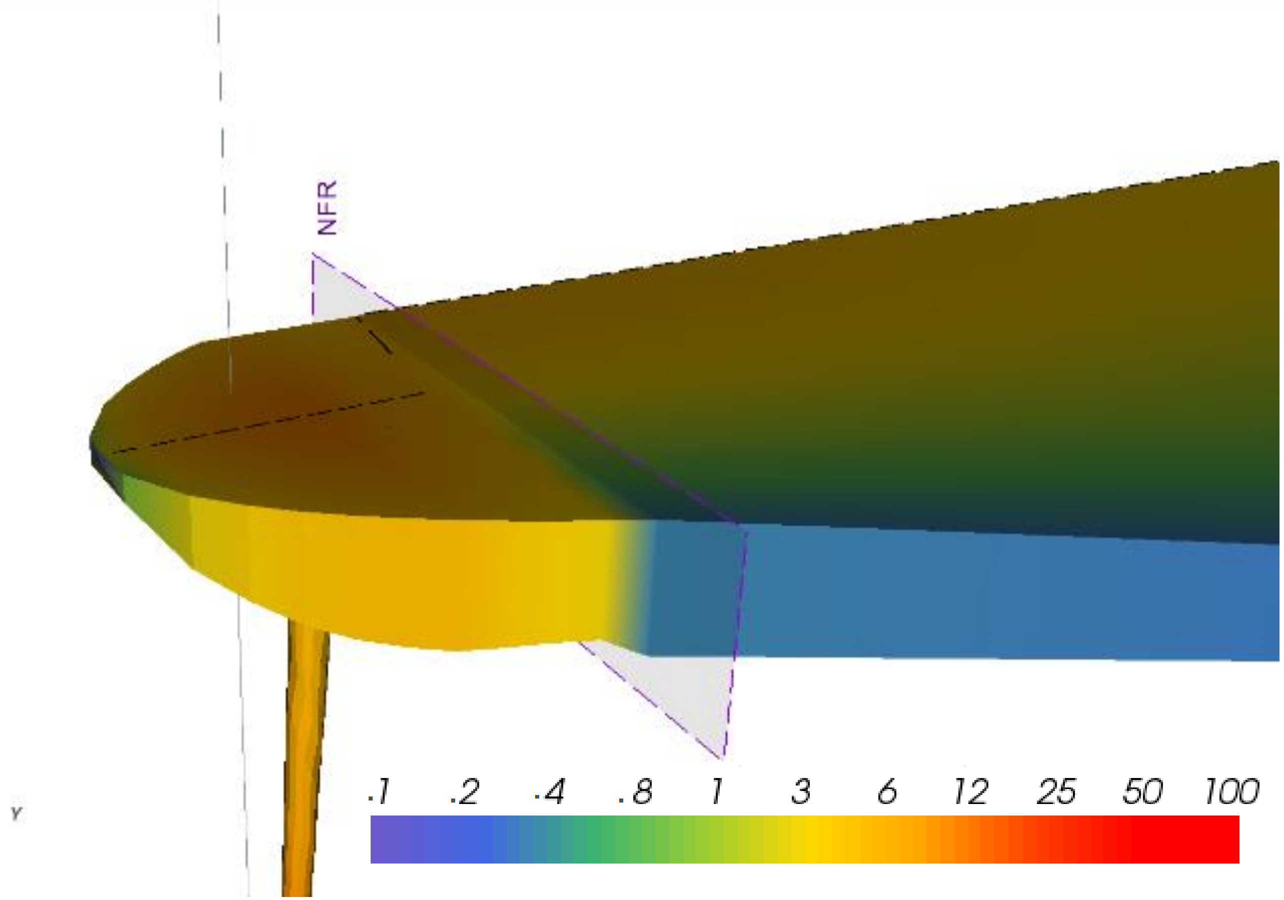


Figure B-13. Dynamics of the treated wastewater plume for Case E ambient conditions – representing conditions just after a neap spring tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

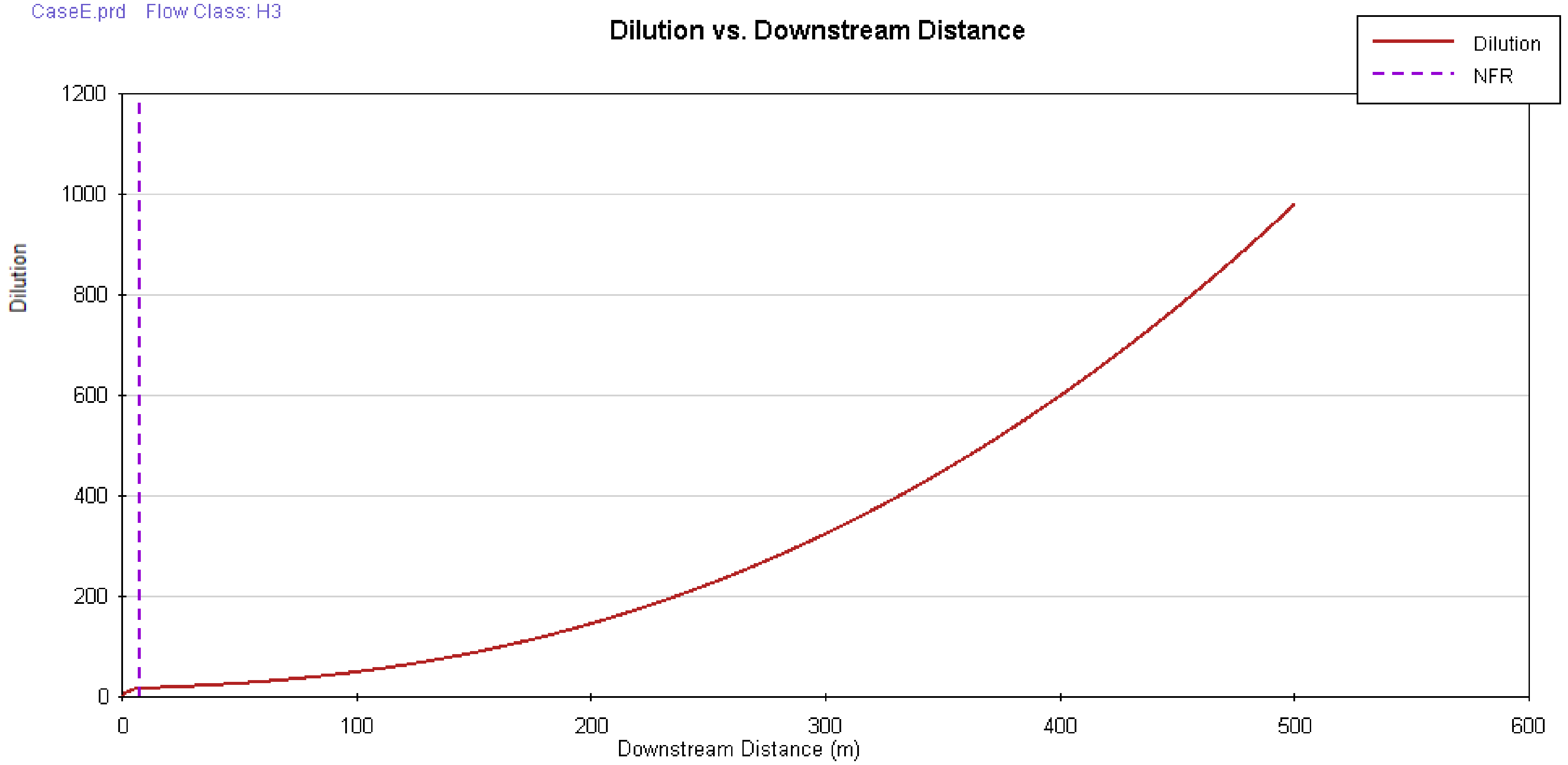


Figure B-14. Dilution versus distance from discharge point (m) in the far field and for Case E ambient conditions – representing conditions just after a spring high tide.

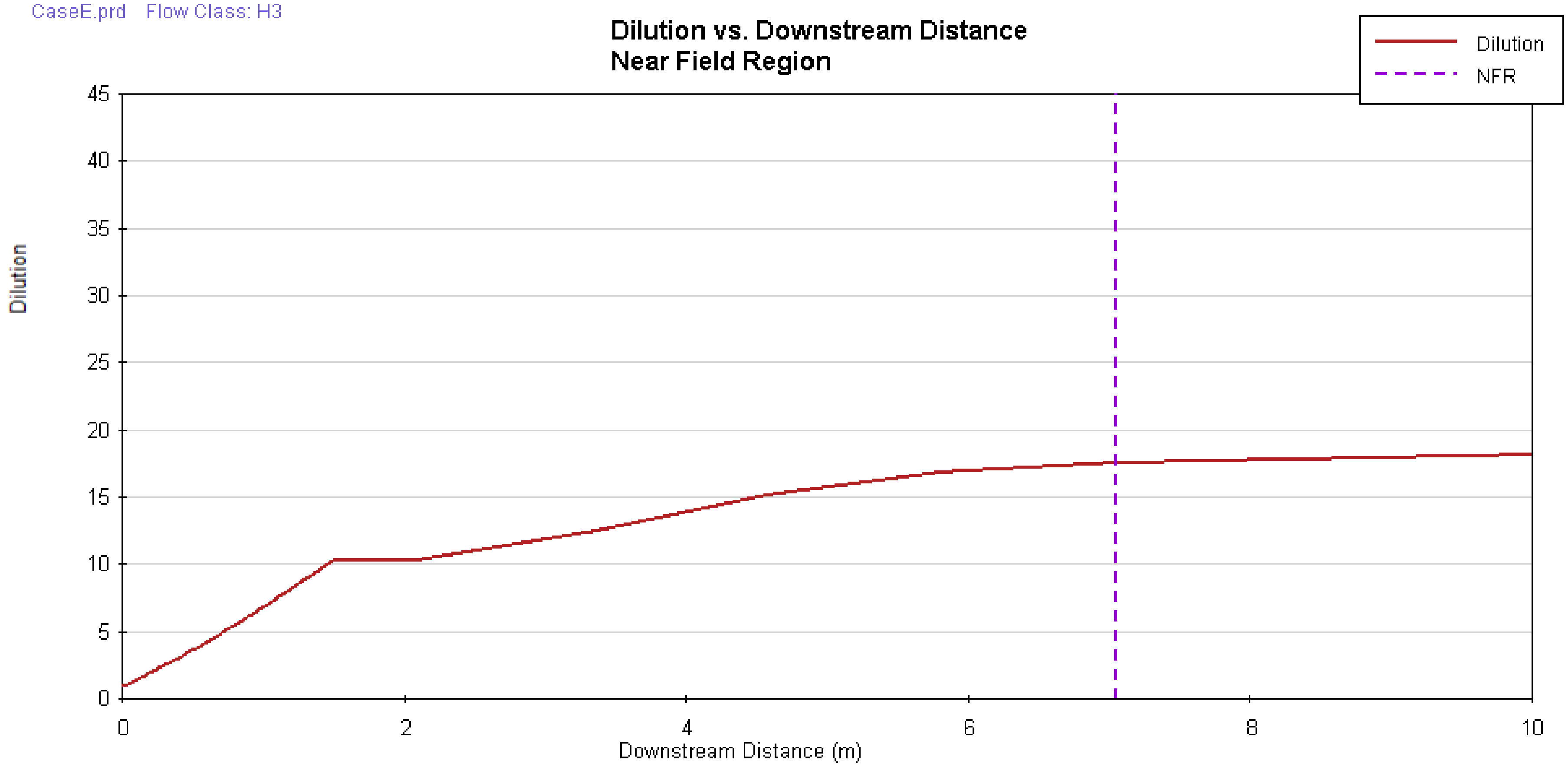


Figure B-15. Dilution versus distance from discharge point (m) in the near field and for Case E ambient conditions – representing conditions just after a spring high tide.

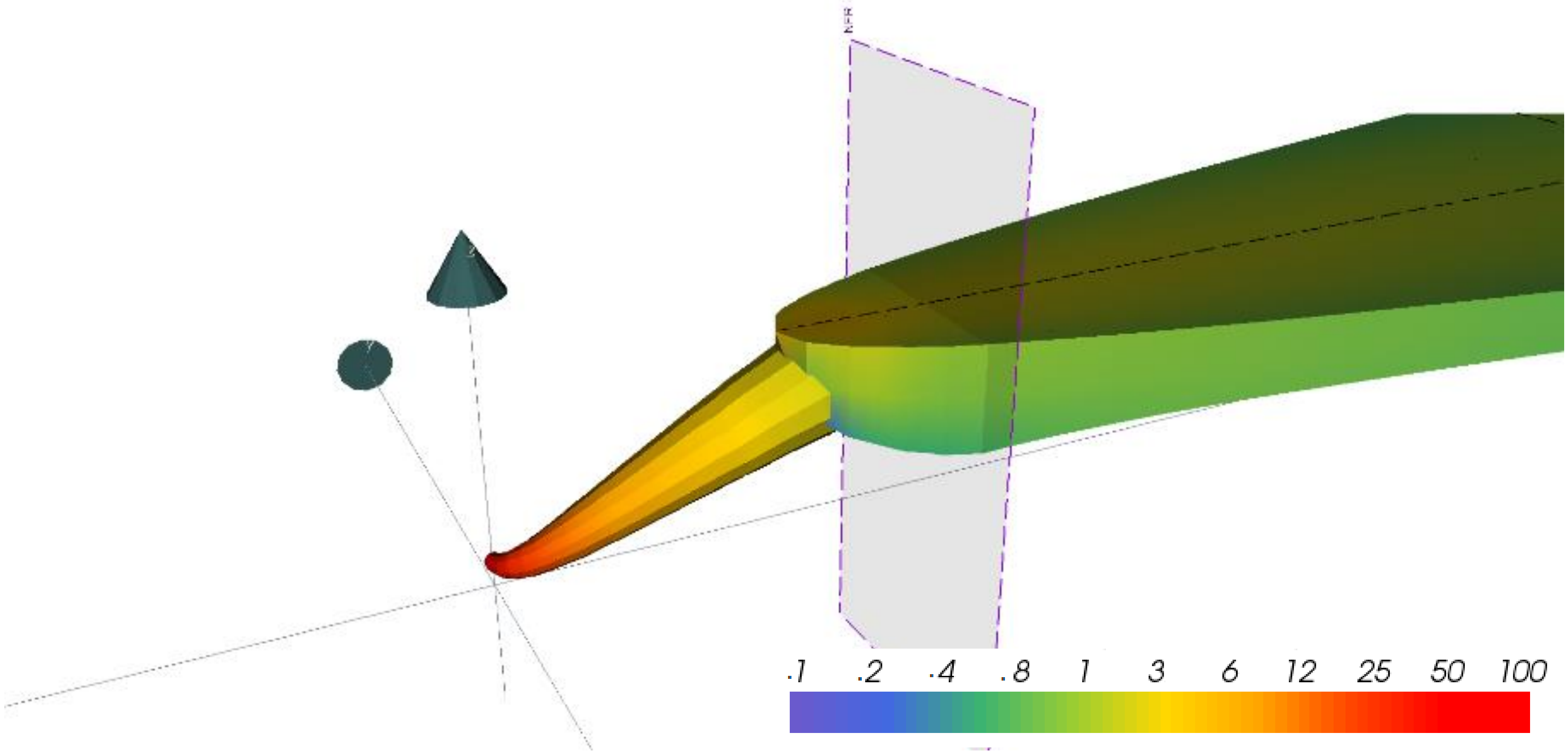


Figure B-16. Dynamics of the treated wastewater plume for Case F ambient conditions – representing conditions at the beginning of a spring ebb tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

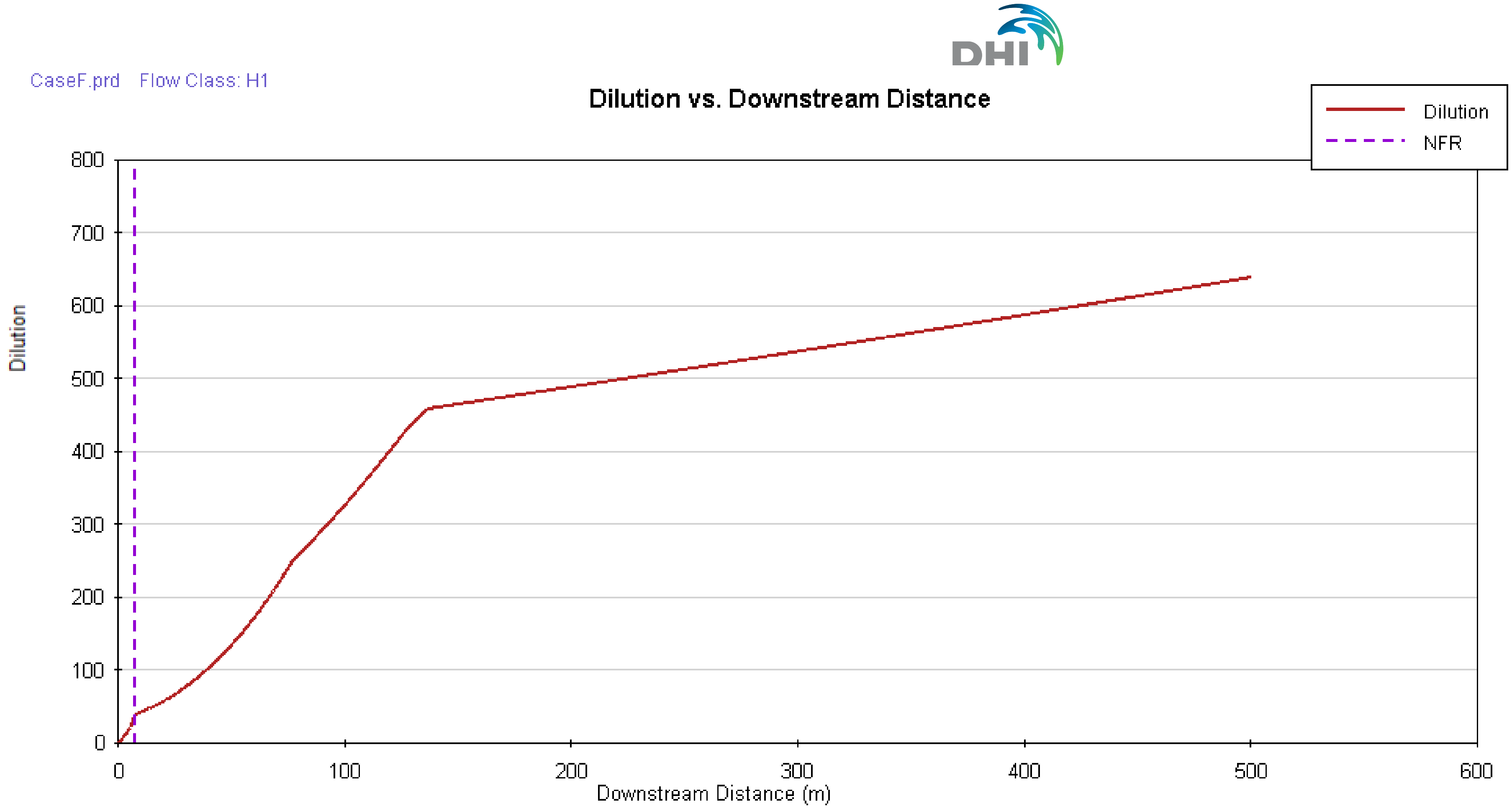


Figure B-17. Dilution versus distance from discharge point (m) in the far field and for Case F ambient conditions – representing conditions at the beginning of a spring ebb tide.

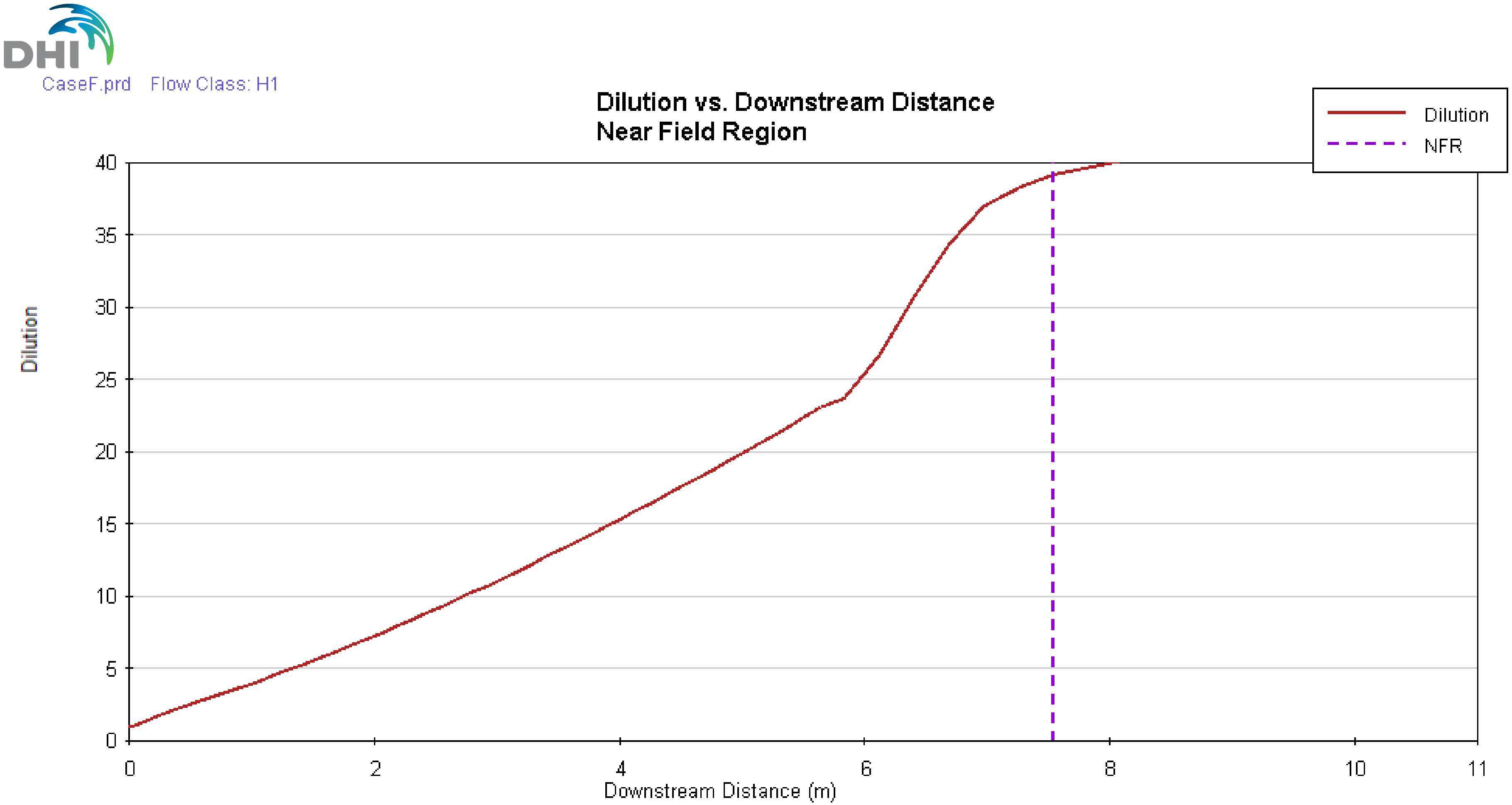


Figure B-18. Dilution versus distance from discharge point (m) in the near field and for Case F ambient conditions – representing conditions at the beginning of a spring ebb tide.

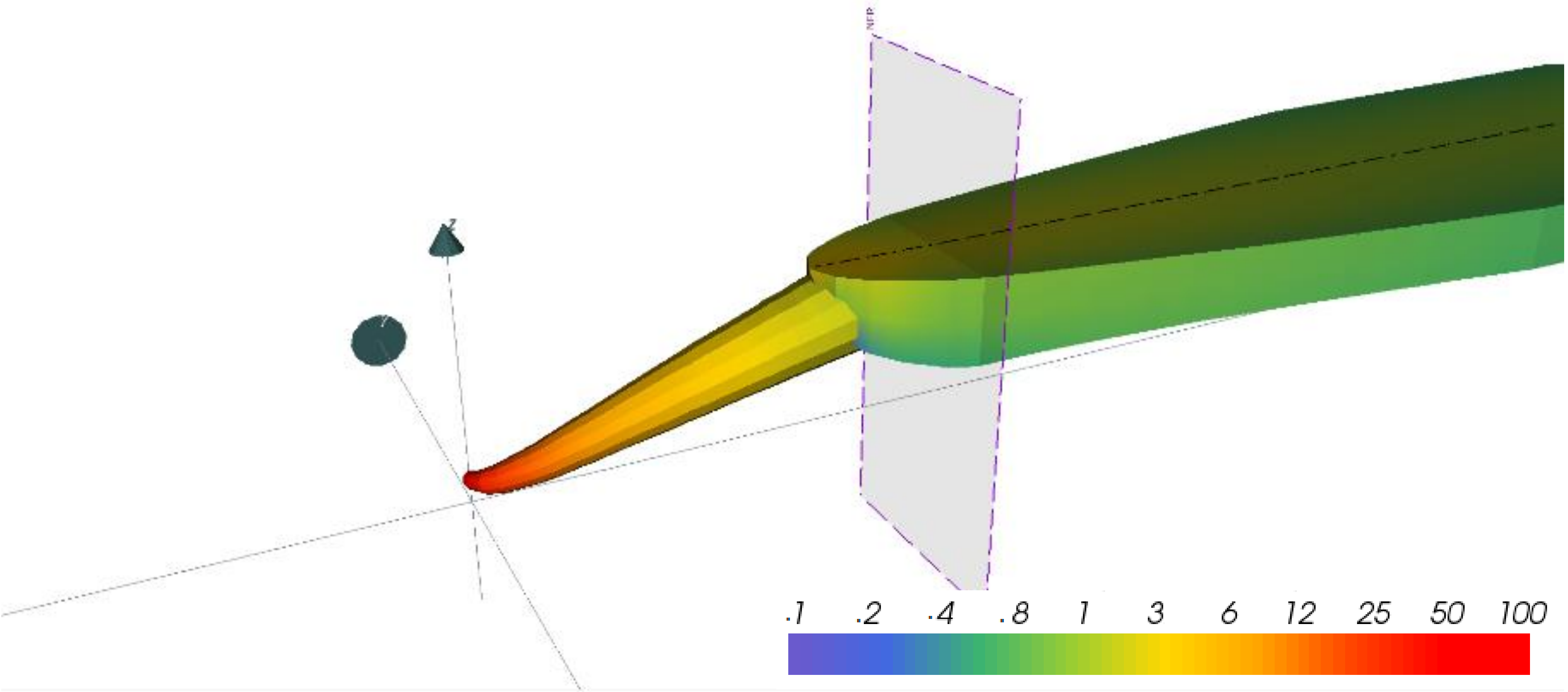


Figure B-19. Dynamics of the treated wastewater plume for Case G ambient conditions – representing conditions at the beginning of a mean ebb tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

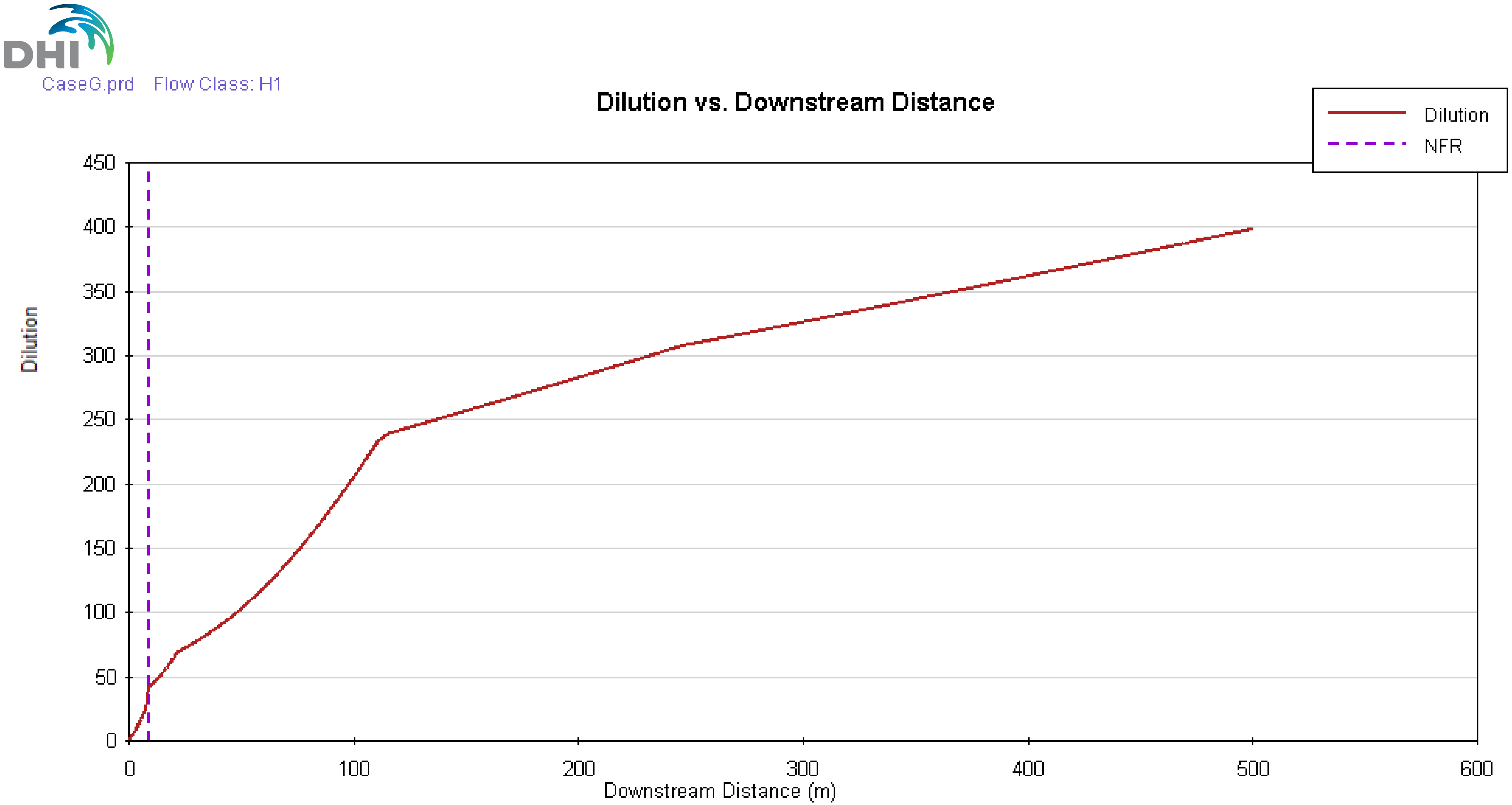


Figure B-20. Dilution versus distance from discharge point (m) in the far field and for Case G ambient conditions – representing conditions at the beginning of a mean ebb tide.

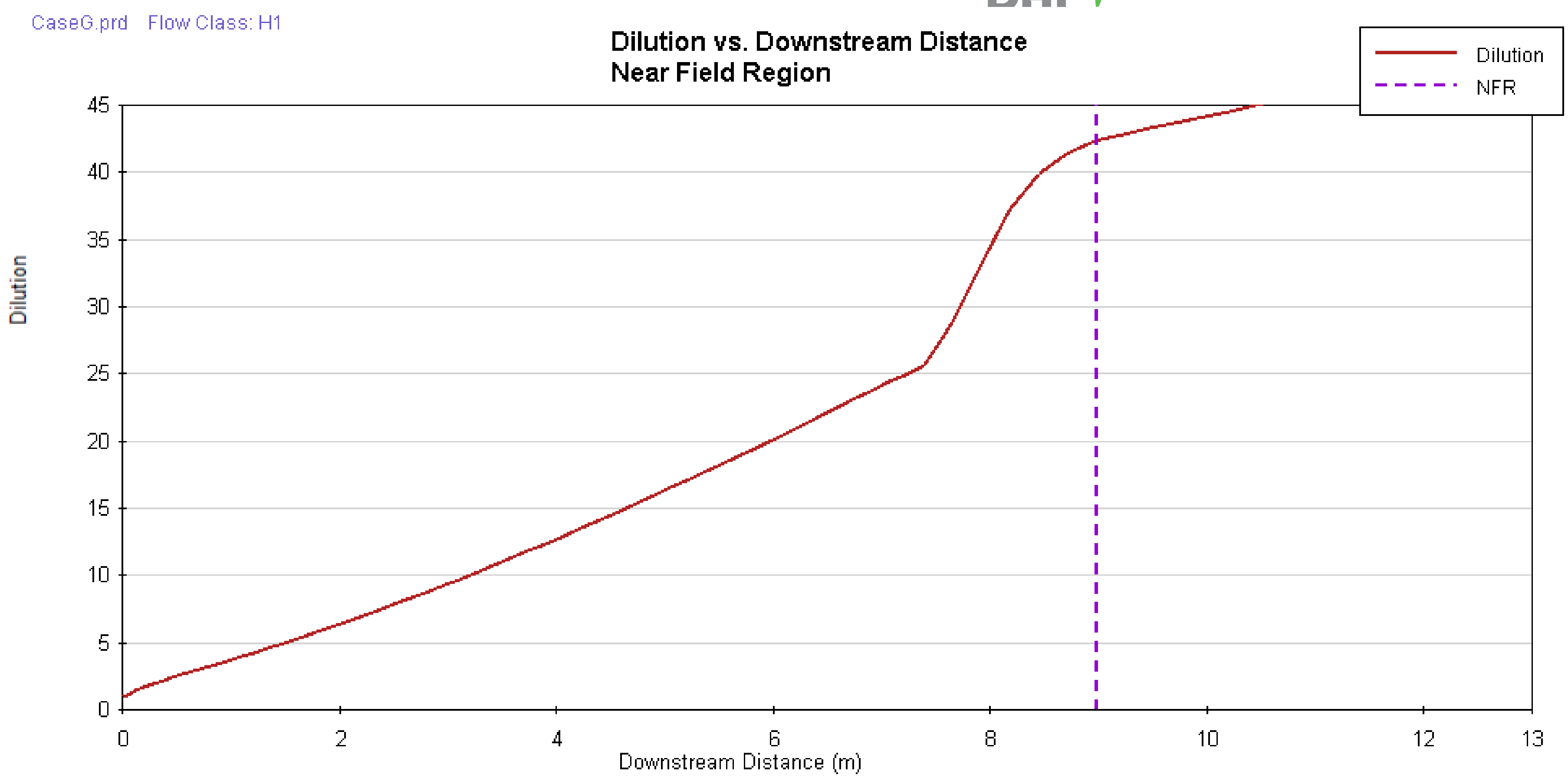


Figure B-21. Dilution versus distance from discharge point (m) in the near field and for Case G ambient conditions – representing conditions at the beginning of a mean ebb tide.

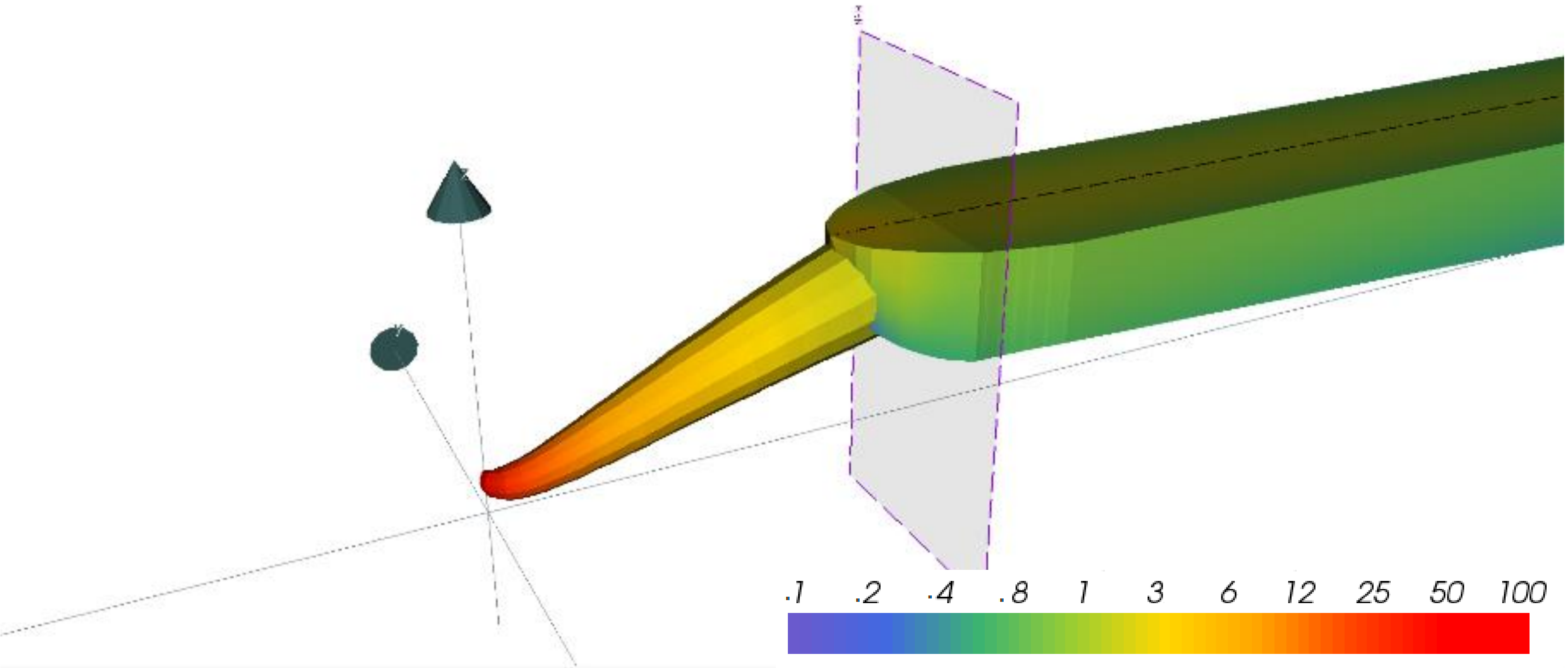


Figure B-22. Dynamics of the treated wastewater plume for Case H ambient conditions – representing conditions at the beginning of a neap ebb tide. Colour coding indicates level of dilution achieved (expressed as percent treated wastewater). The grey shaded area shows the location of the edge of the near-field region.

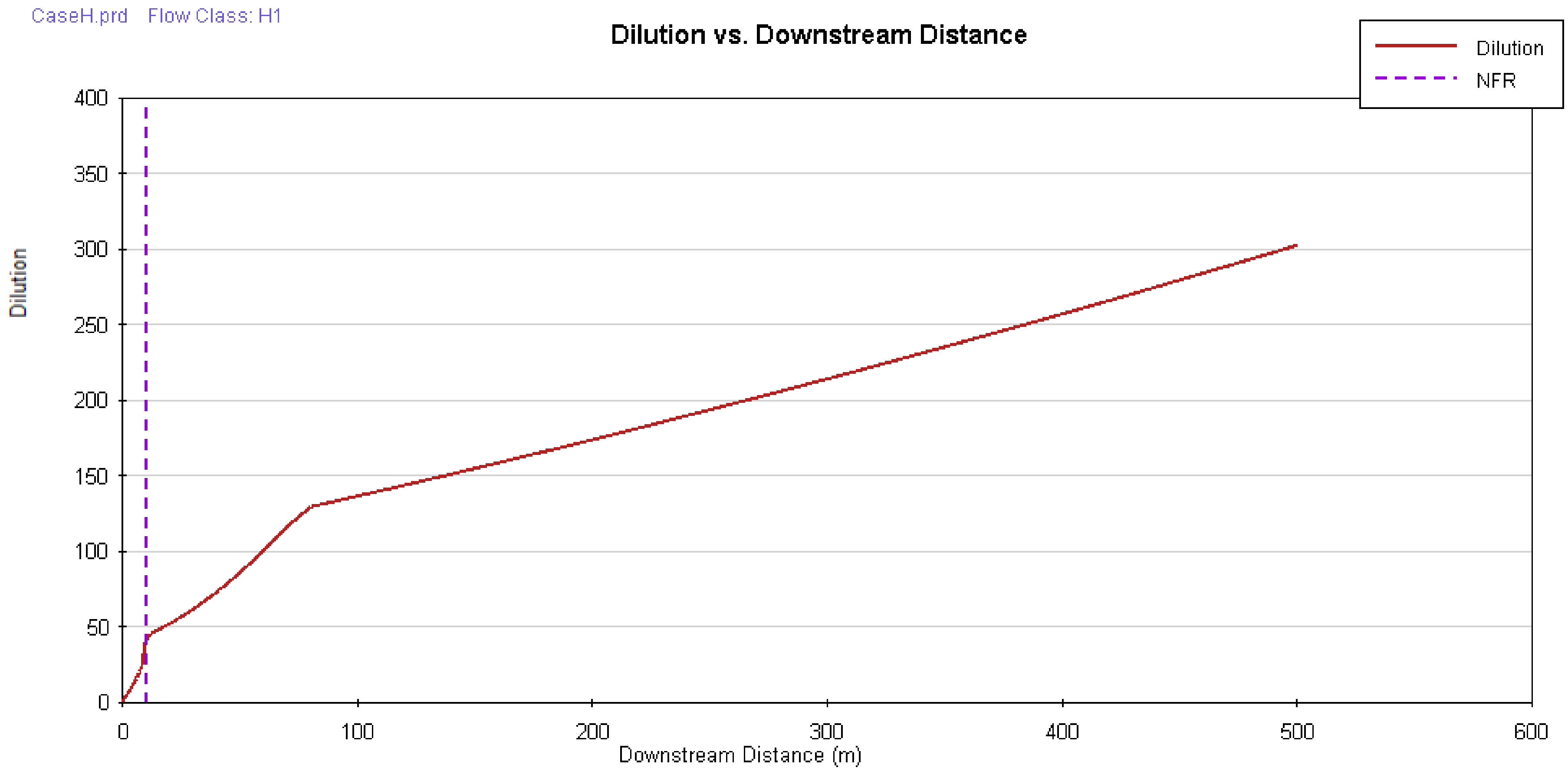


Figure B-23. Dilution versus distance from discharge point (m) in the far field and for Case G ambient conditions – representing conditions at the beginning of a neap ebb tide.

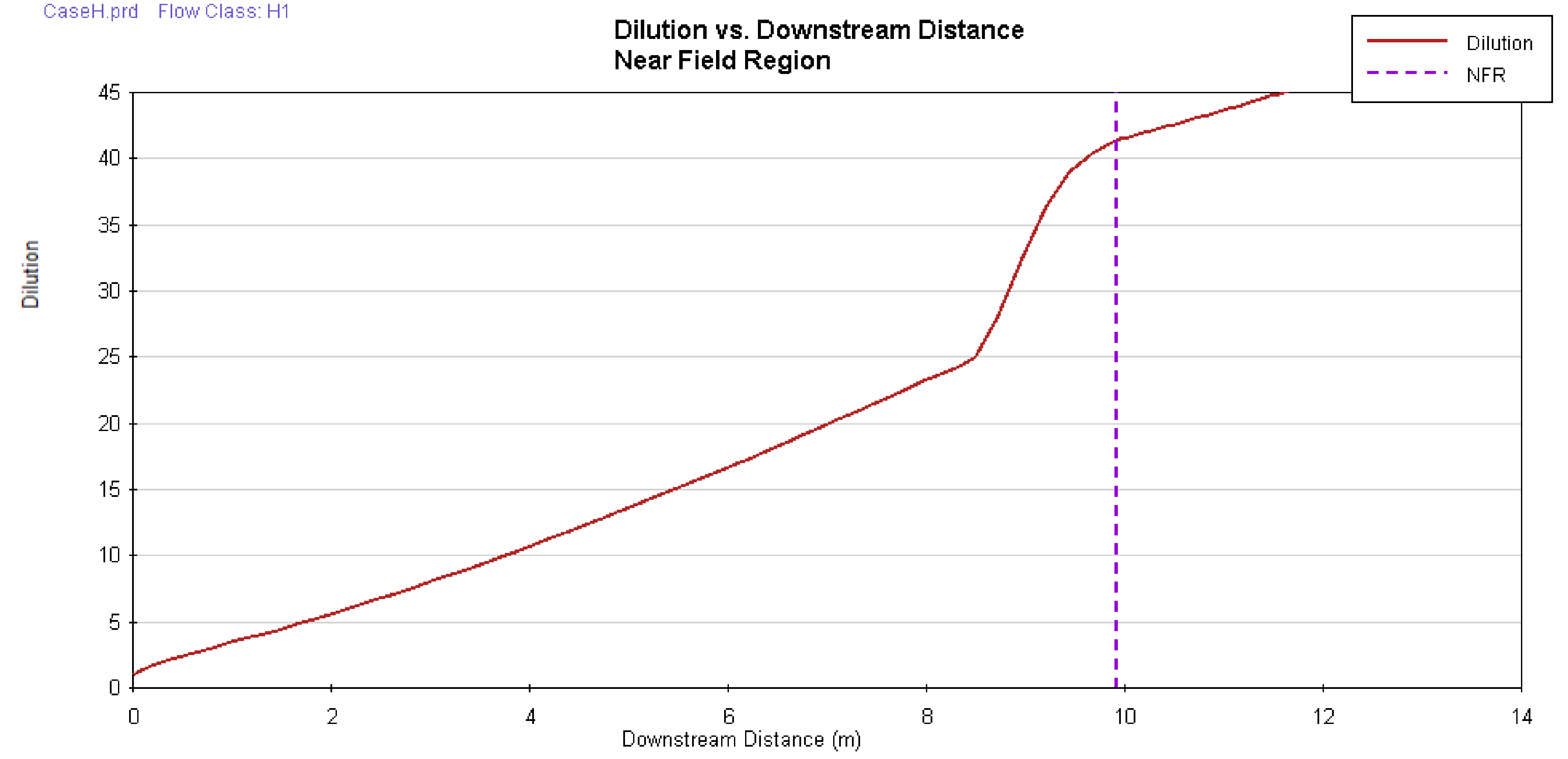


Figure B-24. Dilution versus distance from discharge point (m) in the near field and for Case G ambient conditions – representing conditions at the beginning of a neap ebb tide.

# Appendix C - Empirical Near Field Modelling

Initial dilution estimates were quantified using equations from the Water Research Centre Design Guide for Marine Treatment Schemes (WRC, 1990). These design methods have been incorporated into the regulations and guidelines covering discharges into tidal waters.

The equations assumed a non-stratified water column and a number of equally spaced uniform diffuser ports. The methodology considers buoyancy effects where weak tidal currents occur when mixing is primarily driven by density differences. The effect of increasing ambient currents (when buoyancy effects become negligible) is to create forced entrainment of the treated wastewater in the sea water which leads to increased dilution. The most commonly used predictions for initial dilution are based on early work by Agg (1978a,b) and Cederwall (1968) which are based on results from field experiments where initial dilutions were measured under a variety of tidal conditions. Subsequent work by Bennett (1983) and Bettess and Munro (1981) were used to update the earlier formula of Agg into the standard equations that have been applied for this study.

The flow per port ($P_f$) is defined as the total treated wastewater flow rate divided by the number of ports. The velocity at the port ($P_v$) is defined as the flow per port ($P_f$) divided by the port area $\pi\left(\frac{D}{2}\right)^2$ where *D* is the port diameter.

The densimetric Froude Number (*F*) is defined as

$$F = \frac{P_v}{\sqrt{\left[\frac{(\rho_a - \rho_e)}{\rho_e}\right] gD}} \qquad \text{(Equation 1)}$$

where g is gravity, $\rho_e$ is effluent density and $\rho_a$ is ambient water density.

The Buoyancy Flux (*B*) is defined as

$$B = \left[\frac{(\rho_a - \rho_e)}{\rho_e}\right] gP_f \qquad \text{(Equation 2)}$$

The treated wastewater plume width is defined as 0.76 times $W_d$. The treated wastewater plume depth is defined as 0.37 times $W_d$. The minimum water depth over the diffuser is assumed to occur at Mean Low Water Spring (MLWS). The minimum still water depth over the diffuser ($W_{dmin}$) is the water depth at the outfall site at MLWS minus the Port Height ($P_h$). The minimum still water plume width is defined as 0.26 times $W_{dmin}$. The minimum still water plume depth is defined as 0.13 times $W_{dmin}$.

The still water initial dilution is defined using the Cederwall (1968) formula

$$S_s = 0.54 * F * \left(\left[\frac{0.38 * W_{dmin}}{D * F}\right] + 0.66\right)^{\frac{5}{3}} \qquad \text{(Equation 3)}$$

The minimum port separation ($P_s$) is defined as

$$Ps = W_d\left(0.3 + 0.4\sqrt{\frac{U}{P_v}}\right) \qquad \text{(Equation 4)}$$

The moving water dilution (*ID*) is defined for two cases. If the velocity over the port (*U*) is zero or the water depth ($W_d$) is less than $\left(\frac{5*B}{U^3}\right)$ then

$$ID = \frac{0.27\, B^{\frac{1}{3}}\, W^{\frac{5}{3}}}{P_f} \qquad \text{(Equation 5)}$$

for all other cases

$$ID = \frac{0.27\, U\, W^{\frac{5}{3}}}{P_f} \qquad \text{(Equation 6)}$$


Agg, AR 1978a. Application of Coastal Pollution Research 3 Initial Dilution WRC Technical Report TR 92.

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ASCE 1970. Design and Construction of Sanitary and Storm Sewers. ASCE manuals and reports on engineering practice, No. 37.

Bennett, N.J. 1983. Design of sea outfalls. The lower limit concept of Initial Dilution. Technical Note 354, Proc. Institute of Civil Engineers, 75, Part 2, 113-121.

Bettess, R. and Munro, D 1981. Field Measurement of Initial Dilution at a Sea Outfall, WRC. Technical Report TR 161.

Cederwall, K 1968. Hydraulics of marine waste water disposal. Report 42. Chalmers Institute of Technology. Goteborg.

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# Appendix D – Summary Figures - Hydrodynamics

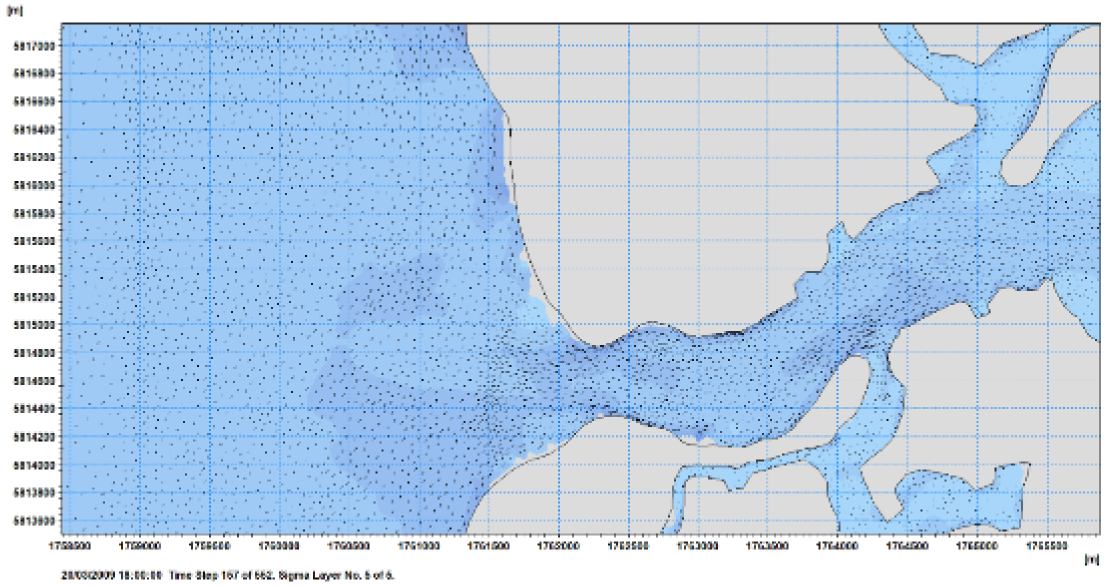


Figure D-1. Typical high water current speed and direction for a neap tide.

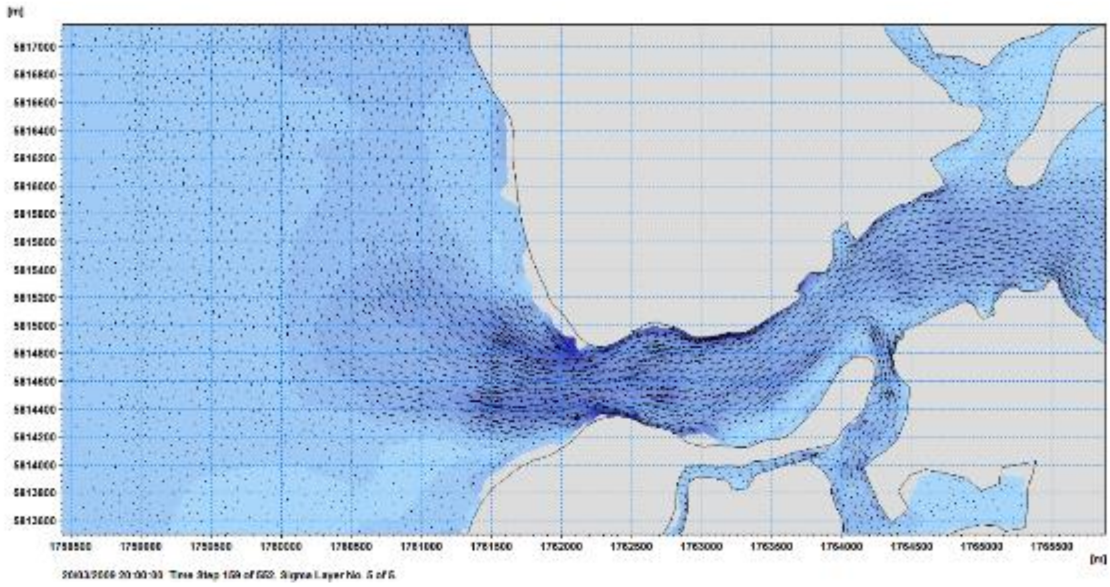


Figure D-2. Typical current speed and direction at the start of an ebb (outgoing) neap tide.

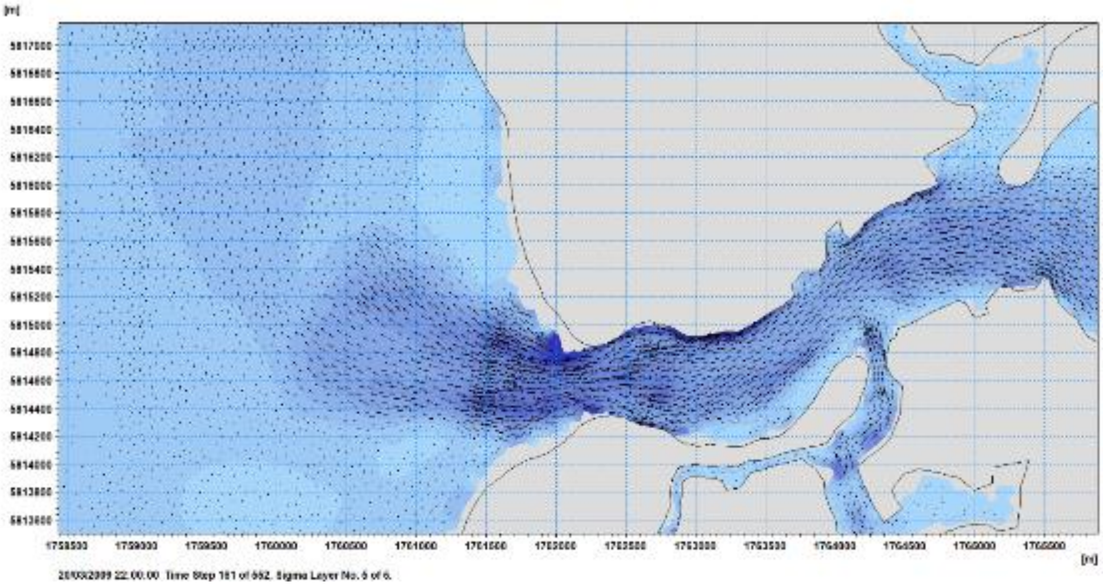


Figure D-3	Typical peak current speed and direction on an ebb (outgoing) neap tide.

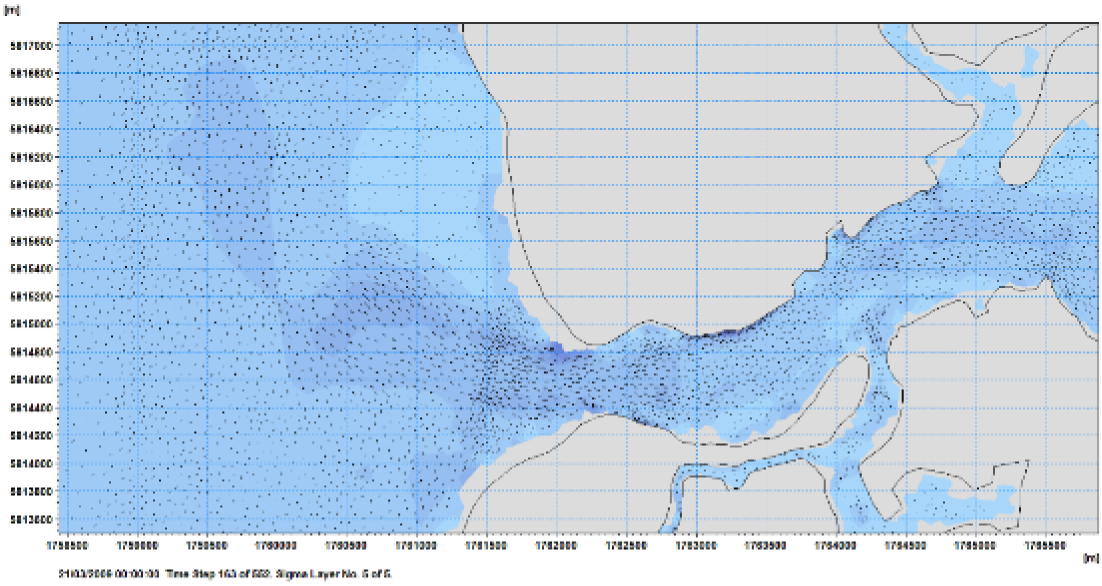


Figure D-4	Typical low water current speed and direction for a neap tide.

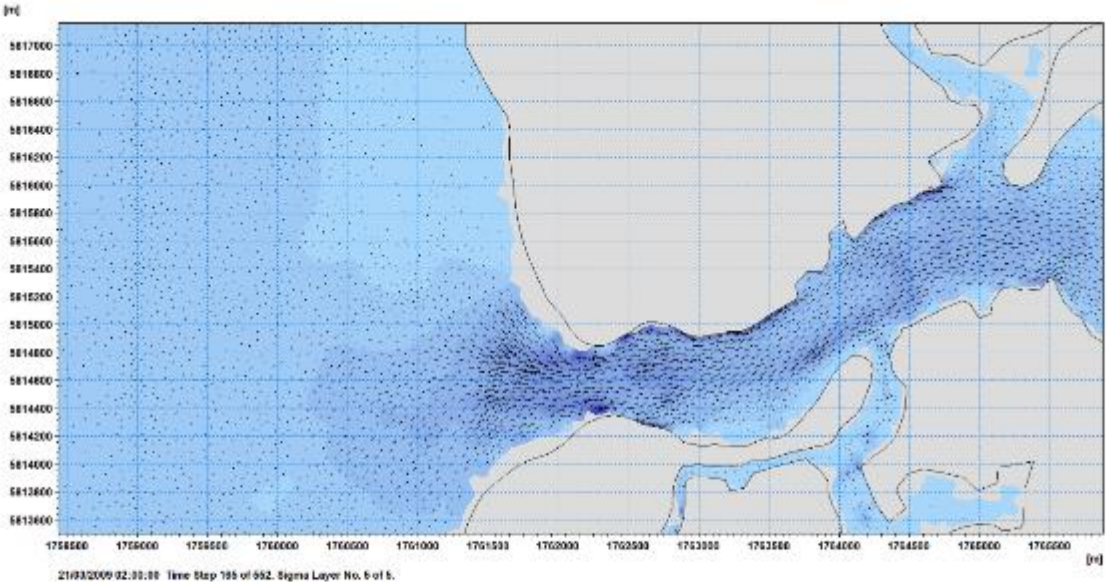


Figure D-5. Typical current speed and direction at the start of a flood (incoming) neap tide.

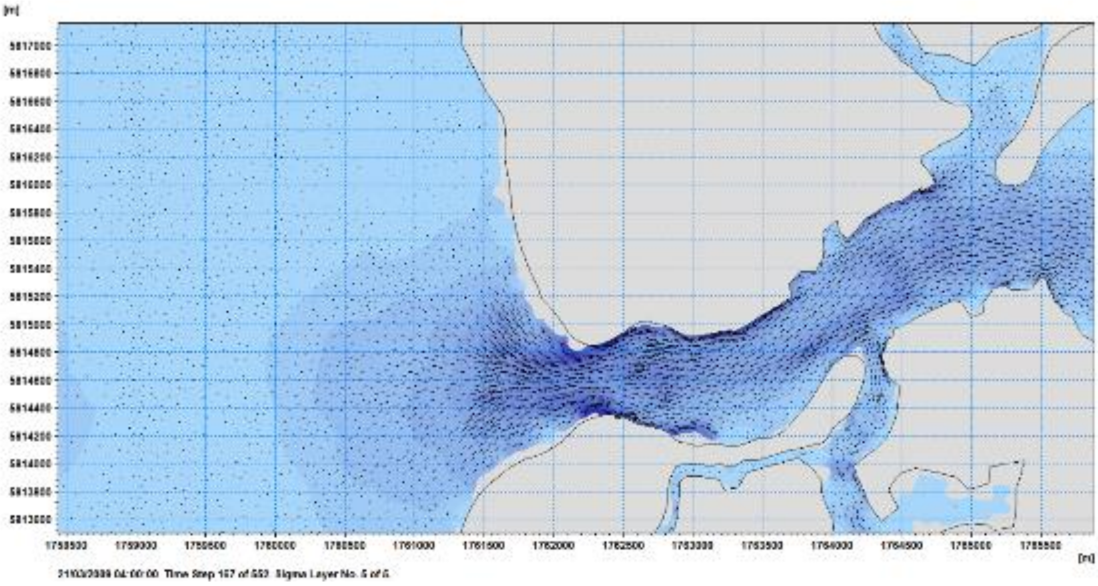


Figure D-6. Typical peak current speed and direction on a flood (incoming) neap tide.

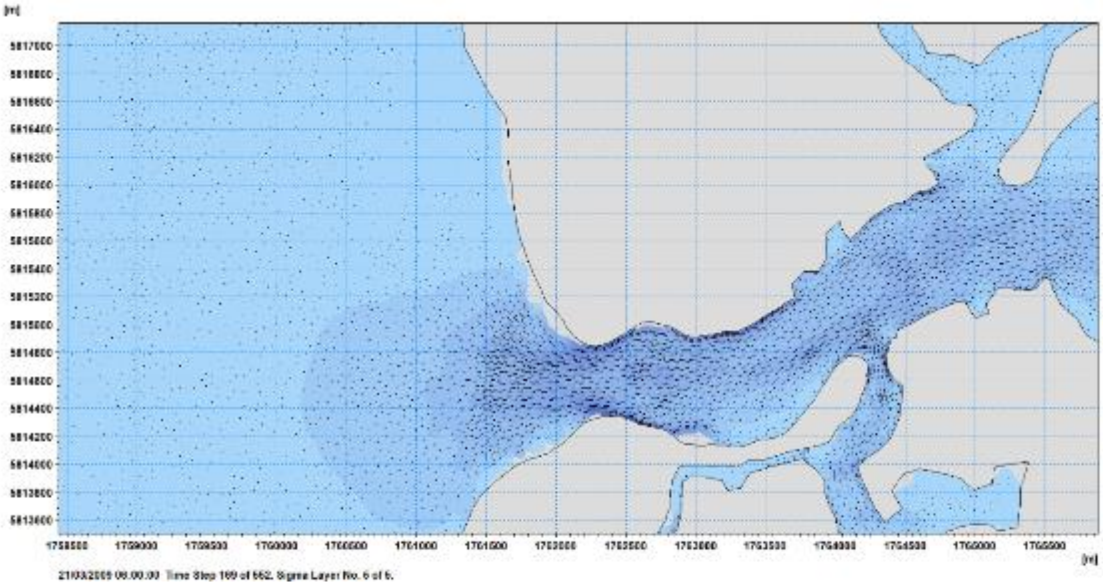


Figure D-7. Typical current speed and direction towards the end of the flood (incoming) neap tide.

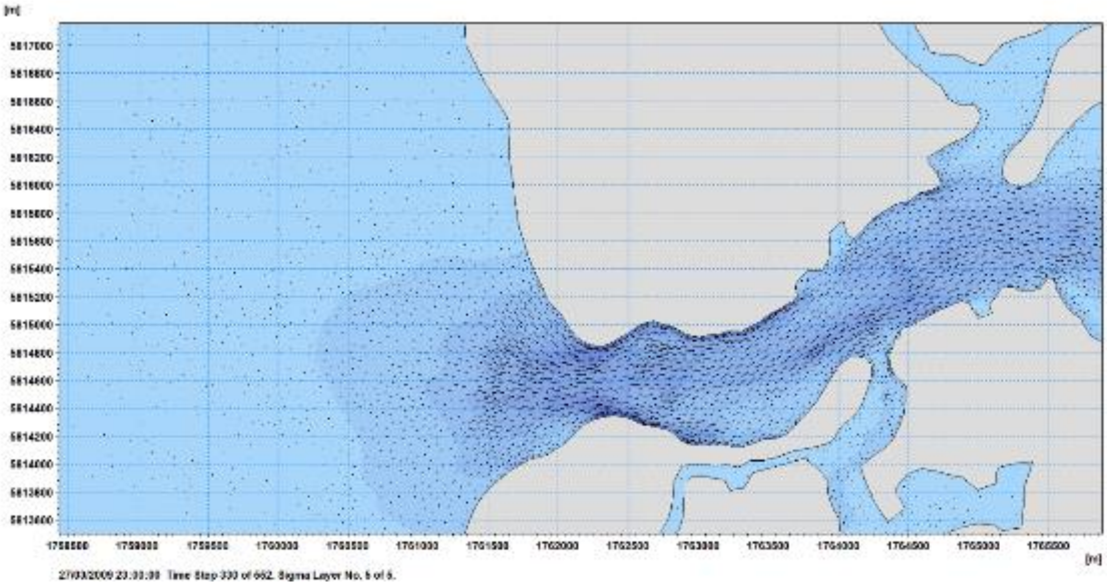


Figure D-8. Typical high water current speed and direction for a spring tide.

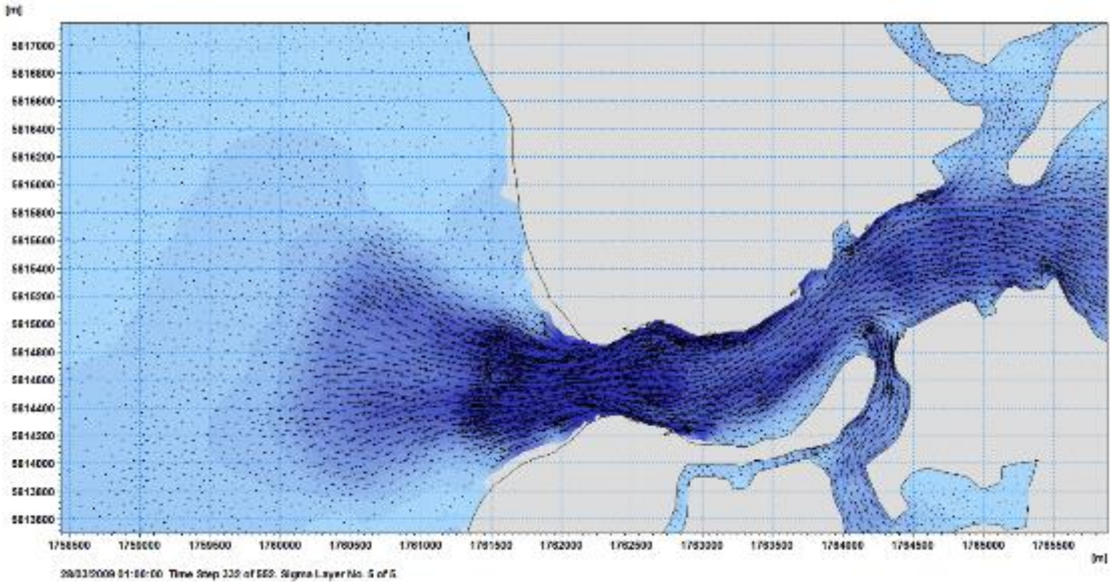


Figure D-9. Typical current speed and direction at the start of an ebb (outgoing) spring tide.

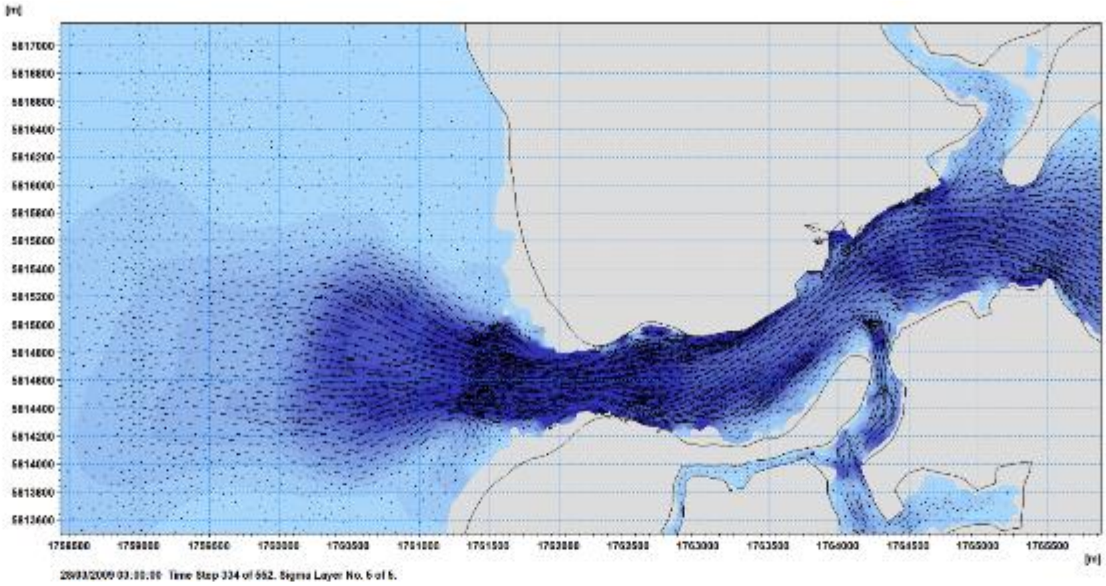


Figure D-10. Typical peak current speed and direction on an ebb (outgoing) spring tide.

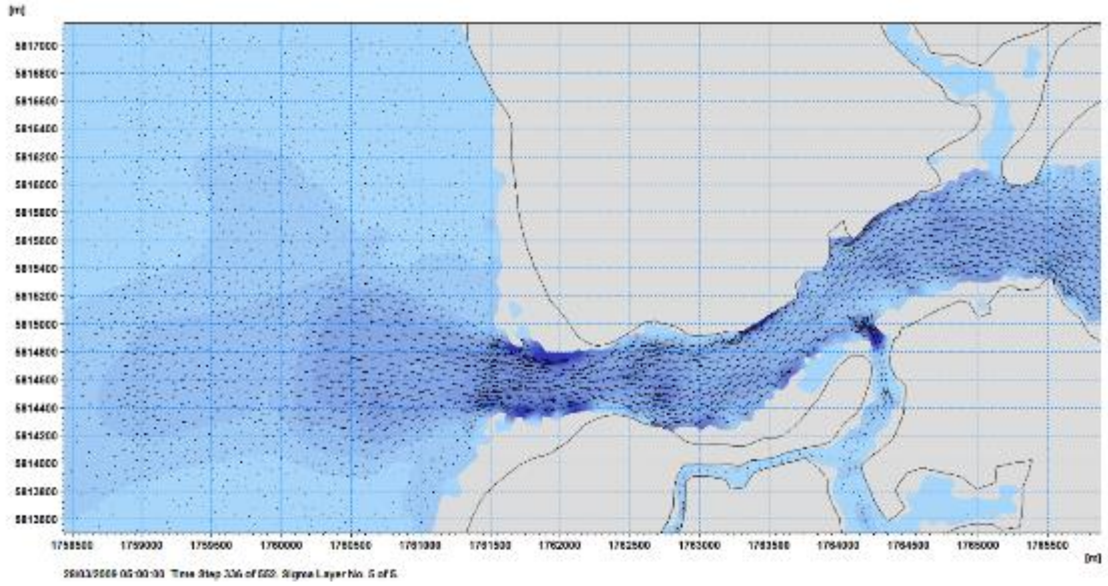
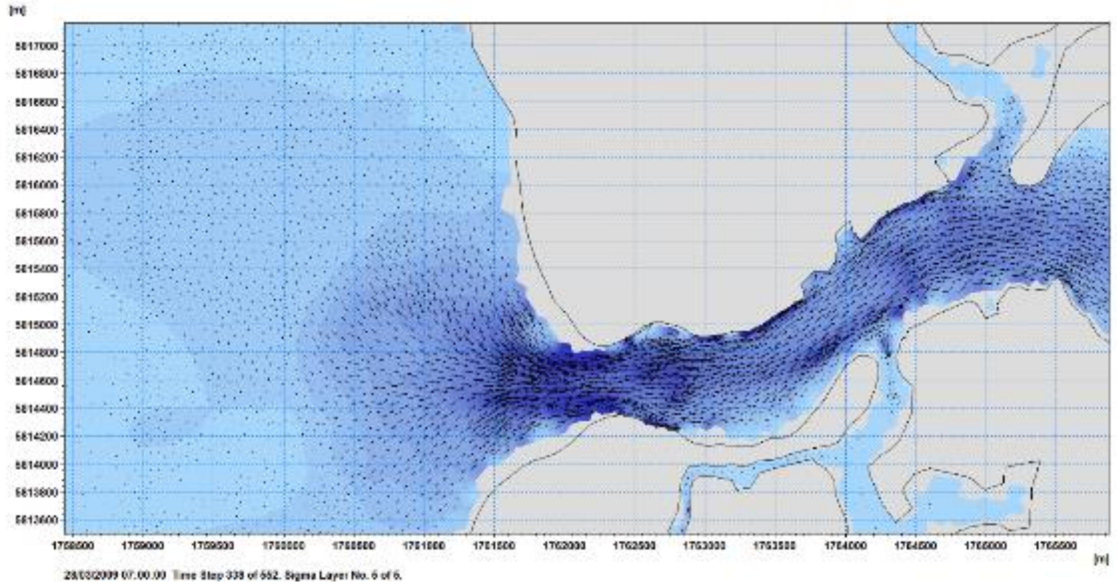


Figure D-11. Typical current speed and direction approaches low water for a spring tide.



FigureD-12. Typical current speed and direction at the start of a flood (incoming) spring tide.

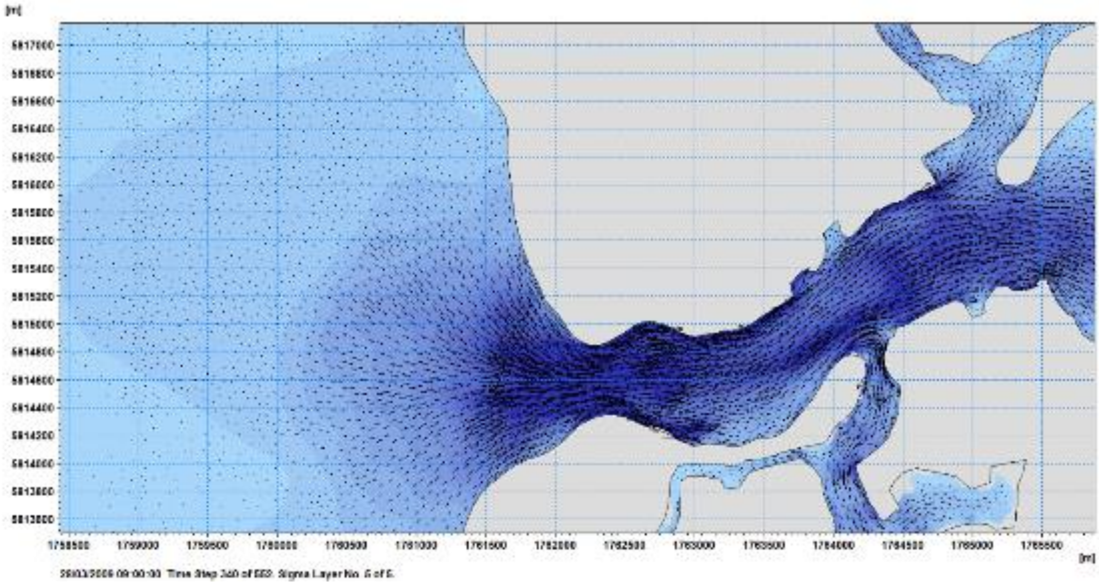


Figure D-13. Typical peak current speed and direction on a flood (incoming) spring tide.

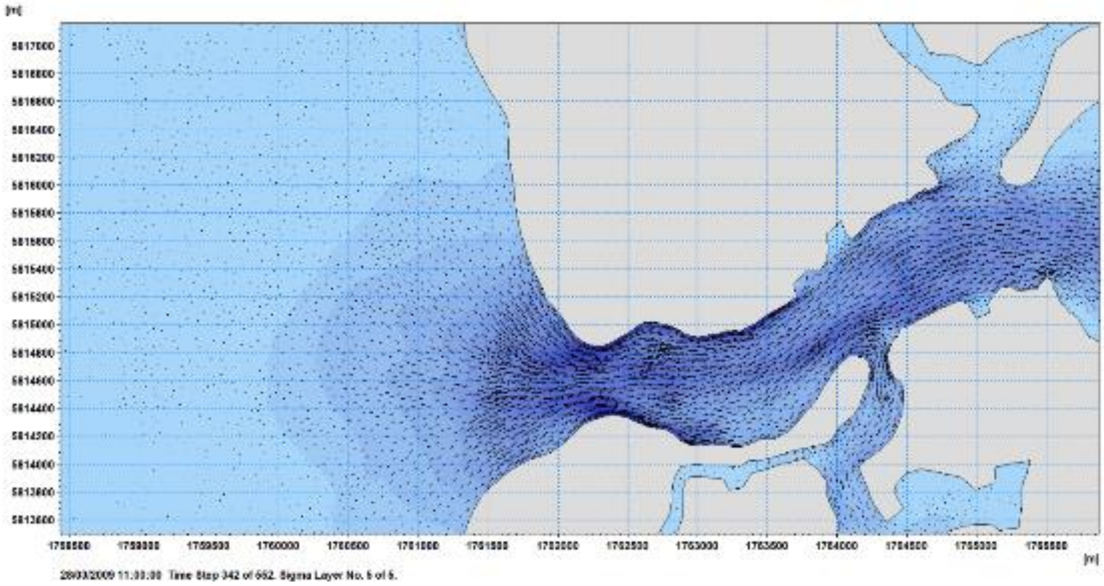


Figure D-14. Typical current speed and direction towards the end of the flood (incoming) spring tide.

# C

Appendix C – NIWA Human health risk assessmet

# Human health risk assessment

## Raglan WWTP

*Prepared for BECA*

*October 2019*

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| Quality Assurance Statement | | |
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# Executive summary

Waikato District Council (WDC) has consent to discharge treated wastewater via a pipeline to the channel that connects Raglan Harbour to the Tasman Sea. This consent is due for renewal, and WDC is evaluating options for future discharge of treated wastewater.

NIWA was engaged to assist with the process of evaluating wastewater treatment options by undertaking the following services:

1. Undertake a quantitative health risk assessment (described in this report).

2. Consideration microbial water quality data derived from routine monitoring programmes (unrelated to the WWTP). This was summarised separately.

The health risks to recreational water users and consumers of shellfish associated with the existing wastewater discharge were assessed using a Quantitative Microbial Risk Assessment processes. Using estimates of dilution derived from a calibrated hydrodynamic model, coupled with Quantitative Microbial Risk Assessment modelling that accounted for a range of WWTP influent virus concentrations, doses of pathogens likely to be ingested by adult and child receptors engaged in contact recreation, we were able to predict infection and illness risks at a range of representative contact recreation and shellfish harvesting sites within Raglan Harbour, and at sites in the Tasman Sea within 3 km of the harbour mouth.

High initial dilution of treated wastewater is achieved, and the concentrations of a conservative tracer are generally low across the model domain. As a consequence, infection and illness risks to both recreational water users and consumers of uncooked shellfish are generally low for all model pathogens selected, and at all sites where exposure to diluted treated wastewater occurred.

Illness risks have been related to the illness risks defined in the New Zealand “Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas” (MfE/MoH 2003). At one $\log_{10}$ removal, illness risks for the most sensitive receptors (children) at all sites but one (the outfall) are below the 1% Gastrointestinal Illness threshold (the “no observable adverse effects level”). The “no observable adverse effects level” is exceeded for acute febrile respiratory illness at several sites at or nearby the outfall (mainly in the channel discharging to the Tasman Sea, where the diluted wastewater is transported out of the harbour on the ebb tide).

Illness risks associated with consuming uncooked shellfish are greatest at two sites along the south shore nearest the discharge outfall but are predicted to be well-below 0.3% illness risk.

We also demonstrate the decrease in illness and infection risk for recreational water-users or consumers of shellfish that may be anticipated as the level of wastewater treatment is increased.

# 1 Introduction

Raglan wastewater treatment plant (WWTP) services the coastal community of Raglan. The influent is primarily domestic sewage derived from the town's population resident population, which fluctuates seasonally in response to tourist influx. There is also a weekly population increase on weekends when visitors from centres such as Hamilton enjoy the local amenity values. The town has grown considerably recently.

In addition to local and international tourist values, Raglan Harbour (Whaingāroa Harbour) and the nearby coast is important to several hapū: Poihākena marae (Ngāti Tahinga of Tainui) is located in Raglan on Wainui Road, Whaingāroa, and Waingaro marae (Ngāti Maahanga and Ngāti Tamainupo) is located approximately 25 kilometres northeast of Raglan, on Waingaro Landing Road.

The WWTP is operated by Waikato District Council (WDC), who currently have a consent to discharge treated wastewater via an outfall pipe located at the mouth of the harbour. The location of the wastewater treatment plant and current discharge in relation to the town and the two marae is indicated in Figure 1-1.


**Figure 1-1: Location of Raglan WWTP and wastewater outfall in Raglan Harbour.**

WDC wishes to renew the resource consent to continue discharge of the treated wastewater. In recognition of the sensitivity of the local community (including impact on the mauri of the harbour), WDC is considering several options for discharge of the treated wastewater. These include continued discharge at the existing site, either using the existing outfall pipe, or using a new outfall pipe (to allow discharge of the wastewater further offshore, nearer the centre of the channel, which would also allow the pipe to be buried), disposal of the total discharge volume to land by irrigation, discharge of the total volume of wastewater via a new outfall located outside of the harbour, or a hybrid option, where land disposal would be favoured when soil moisture conditions permitted, with the existing or one of the new outfall options used when soil conditions where unfavourable for disposal to land.

The WWTP is currently a pond-based system, with limited options for treatment to improve the quality of treated wastewater. As part of the consent renewal process, WDC is also considering increasing the level of wastewater treatment. Options include use of membrane filtration, chemical dosing and increased UV disinfection. Selection of the wastewater treatment option will be determined by several factors, including the location and nature of the future discharge. Another important factor that will direct the level of future wastewater treatment is the health risk associated with the discharge. Despite being well-managed, all treatment systems have the potential to be sources of pathogens – exposure to these pathogens carries the risk of infection and illness.

To assist with the process of evaluating wastewater treatment options, and to facilitate discussion with the community, NIWA was engaged to provide the following services:

1. Undertake a health risk assessment (described in this report).
2. Consideration microbial water quality data derived from routine monitoring programmes (unrelated to the WWTP).

The details for the second task were reported separately (Hudson 2019). Data derived from these other monitoring programmes were not focused on the wastewater discharge. This work was undertaken to provide a microbiological water quality context, within which the detailed health risk assessment could be considered.

### The objectives of the health risk assessment

The primary objectives included:

- Assessing the health risks arising from the discharge of treated wastewater via several modes of exposure of human targets to pathogenic microorganisms via
  - contact recreation (involving ingestion or inhalation of organisms by swallowing highly diluted treated wastewater, or inhalation of aerosols derived from treated wastewater)
  - consumption of raw shellfish that may be exposed to the plume of highly diluted, treated wastewater.
- Identify pathogenic organisms likely to be present in the diluted wastewater, selecting several as candidates for the risk assessment, considering
  - routes whereby the community may ingest these organisms
  - modes of exposure, and
  - the volumes of water likely to be ingested, or meal sizes if the route of exposure is uncooked shellfish.

- Concentration ranges for the model pathogens, and dose response curves for these model pathogens were also to be considered.
- In all cases, ranges of values (e.g., volumes of water ingested doses, concentrations and exposure times etc.,) were to be used rather than single values (such as average, median or 95th percentile values).

Health risk is presented primarily as estimates of Individual Infection Risks (IInR) or Individual Illness Risks (IIR), related to both the level of treatment proposed for the wastewater, and the location at which exposure is likely.

The numbers of scenarios considered in this risk assessment is determined primarily by the number and type of exposure sites, and the range of model pathogens. The location and type of discharge site was not modelled at this stage, because we were uncertain whether an assessment would be required for one or more possible wastewater discharge points. All risks assessed assumed that the treated wastewater would be discharged from the existing site. Although the future efficacy of wastewater treatment is unknown at present, we considered risks associated with discharge of wastewater following four levels of wastewater treatment.

# 2 Assessing human health risks

**Risk assessment** is applied to a diverse range of activities, including workplace health and safety, the design of structures, the planning and operation of space missions. Despite the diversity of these activities, several common factors need to be considered, and are provided here as definitions to guide the reader:

- **Hazard** - anything (e.g., work materials, equipment, methods, practices or activities) that has the potential to cause harm. In this case, the hazard is a wastewater discharge.
- **Risk** - the chance, high or low, that somebody may be harmed by the hazard. Risk is sometimes defined as **chance + hazard + exposure + consequence**, or "the likelihood of identified hazards causing harm in exposed populations in a specified time frame, including the severity of the consequences".[1] By its nature, risk is probabilistic and estimating risk requires the development of quantitative information.
- **Risk assessment** - the process of evaluating risks to individual safety and health arising from the hazards. It is a systematic examination of all aspects of an activity that considers:
  - what could cause injury or harm
  - whether the hazards could be eliminated, and if not
  - what preventive or protective measures are, or should be, in place to control the risks.

Human health risks arising from exposure to microbial contaminants during recreational activities are generally assessed using recreational bathing monitoring programmes. The Ministry for the Environment and Ministry of Health "*Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas"* (MfE/MoH 2003) (MfE/MoH Guidelines) provide guidance regarding establishment and operation of recreational water quality monitoring programmes, and when interpreting the results derived from monitoring. Monitoring recreational water quality generally relies on use of faecal indicator bacteria (FIB) – enterococci is favoured in saline waters.

The MfE/MoH Guidelines are quite clear, however, that they should not be used under several circumstances or for specific purposes:

1. "to directly 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. The relationship between indicator bacteria and disease-causing bacteria, viruses and protozoa in the discharge needs to be established" (p 3).
2. "to assess the microbiological quality of water that is impacted by a nearby point source discharge of treated effluent without first confirming that they are appropriate …… when planning the location and degree of treatment for wastewater treatment plants to recognise that the guideline values are not necessarily a guarantee of safety" (p 3).

[1] http://qmrawiki.canr.msu.edu/index.php/Quantitative_Microbial_Risk_Assessment

3. during periods of “exceptional circumstances”, such as when there is a major outbreak of a potentially waterborne disease in the community, and where that community’s sewage contributes microbiological contaminants to receiving waters (p D9).

When the circumstances or conditions prevail, alternate methods are required to assess human health risks arising from possible exposure to pathogens. These risks may be calculated using Quantitative Microbial Risk Assessment (QMRA) techniques, as explained hereafter.

QMRA is a framework and approach that brings information and data together with mathematical models to describe or predict the spread of microbial agents through environmental exposures and to characterise the nature of the adverse outcomes. Although most microbes are harmless or beneficial, some are extremely dangerous – these are termed pathogens or Biological Agents of Concern (BAC). Although these have the potential to cause serious or fatal illness, they differ greatly in their physical characteristics, movement in the environment, and process of infection. These characteristics and the differences between potential pathogens are considered in the risk assessment process, to ensure that appropriate “model” pathogens are selected to assess human health risks.

This report explains the requirements for undertaking a QMRA, including data regarding receiving environment conditions and the choice of pathogens. In this study, several significant assumptions are required - justification of these choices is presented as well.

# 3 Methodology for conducting a QMRA

As indicated above, risk is probabilistic and estimating risk requires the development of quantitative information. QMRA consists of five basic steps:

A. Selection of the hazard(s), i.e., the pathogen(s) of concern—exposure to which can give rise to illness.

B. Assessment of exposure to the pathogen(s) at key sites (in terms of pathogen concentrations and duration of exposure).

C. Characterisation of human response to pathogen dose (creating suitable dose-response curves) – described in Appendix C.

D. Calculation of the health risk (in terms of infection and/or illness).

E. Communication of health risk, identifying appropriate response and mitigation actions.

Several components associated with or required for steps A-D are described in the schematic in Figure 3-1.

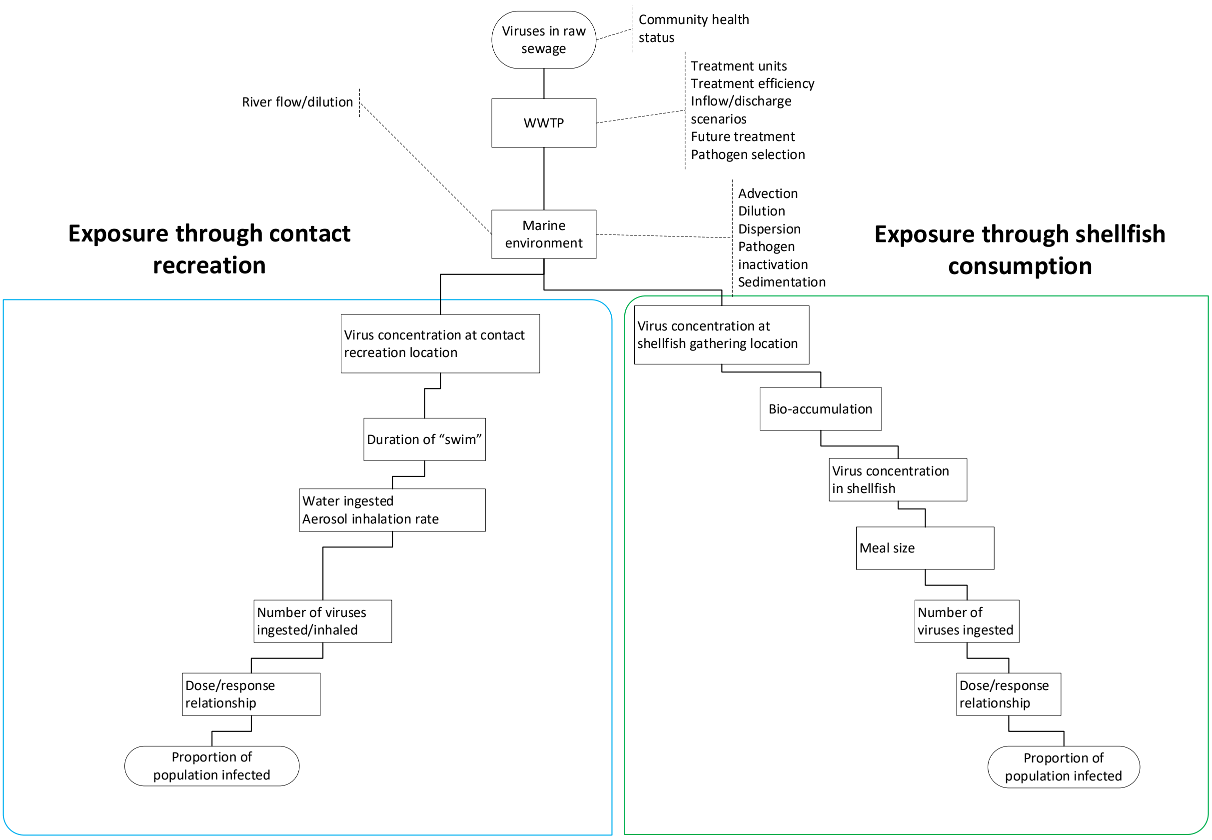


**Figure 3-1: Process followed to relate human health risk to pathogen-contaminated surface waters.**

- The lines and boxes in Figure 3-1 indicate the path followed from source ("Viruses in wastewater"), to the numbers of individuals likely to become infected or ill. Because a large representative "population" is used for the calculation, the results are generally expressed as a proportion.
- Callout boxes indicate the type of information or data required to make the process work.
- Data and information are required for the processes identified in the red boxes as well, but these data and information are less site-specific, and may be accessed from the literature, or values may be assumed (e.g., "Duration of swim or other activity").

In a full QMRA, these data and information are used as follows:

- Distributions of tracer concentrations are created in response to a range of factors, such as river discharge/flow, tidal movements, tidal stage, rainfall, and near-field and far-field mixing and dilution processes. The latter takes into account processes that influence the concentrations of the contaminant or tracer of interest, such as viral inactivation, and flocculation and sedimentation of pathogens.
- These tracer concentrations are then normalised as dilutions, relative to nominal concentrations of the tracer present in the treated wastewater.
- The pathogen concentration is also likely to vary widely according to the health status in the local community, the relative dilution of the wastewater, as well as in response to factors causing virus inactivation or attenuation. A likely range of concentrations based on measured values is used in this work – this is described in detail in Section 3.3.
- Once these distributions are created, "recreational users" are exposed to a large number of likely concentrations, selected randomly using a Monte Carlo procedure.
- When calculating risks, "Monte Carlo" statistical modelling is used, which calls for repetitive sampling from distributions and ranges of key variable concentrations, rather than just using single average concentration values. This approach is particularly important given that much of the risk is caused by combinations of inputs toward the extremes of their concentration ranges, the combined effects of which may not be detected when using average concentration values. Typically 10,000 iterations are considered.
- The concentration of pathogens directly controls the size of infectious doses – the volume of water or size of shellfish meal that needs to be ingested to be exposed to the number of organisms ('dose') required to cause infection.[2]
- This effectively allows the health risk to be estimated following exposure of a hypothetical large population size (typically 100 "individuals"), exposed on any particular "day".

[2] Different individuals have different responses to a given dose, with some becoming infected, others not. Infectivity is therefore characterised by a dose-response curve ('function') and risk calculations need to be made for this range of sensitivity. Using averages to calculate a single risk value is highly inaccurate.

- The output from this modelling process are estimates of **illness** risk, in addition to infection risk, attributable to the discharge of wastewater. These health risks are calculated for individuals engaged in **primary contact recreation** near the discharge. "Primary contact recreation" refers to activities such as swimming and paddling where full immersion is anticipated, i.e., ingestion of contaminated water is likely to be an outcome of recreation.
- The model may be refined to provide risk estimates for adults and children as targets, recognising differences in susceptibility between age groups.

Items A), C) and D) above may be addressed using reported data, values from the scientific literature, or other information that is relatively easily available. Item B) is derived from hydrodynamic modelling, which predicts the likely dispersion and dilution of materials discharged from the WWTP in the receiving environment. The model allows likely concentrations of pathogens to be to predicted anywhere in the model domain. This in turn allows the dose of pathogens to which human receptors may be exposed to be estimated for any location across the model domain.

NIWA has undertaken the human health risk assessment using:

1. Recently published scientific literature that has revisited previously accepted relative risk factors.
2. Estimates of wastewater pathogen concentrations derived from a database of published New Zealand and international concentration data.
3. Estimates of the range of dilution likely to occur in the receiving water at sites of interest – in the current study, these data were provided by an independent agency (DHI, 2019).
4. Estimates of virus ingestion rates derived from other published studies.
5. Available dose-response relationships for a series of representative viral and bacterial pathogens.

We describe these selections below.

## 3.1 Selecting the pathogen(s) of concern

To select appropriate pathogens we first need to consider the water-related diseases that may arise. Microbiological water quality guidelines developed both in New Zealand (MfE/MoH 2003) and internationally (WHO 2003) are based on investigations indicating that risks associated with wastewater-contaminated water comprise two types of infection and illness:

1. Gastrointestinal disease, via ingestion during recreational water-contact, and consumption of raw shellfish flesh.
2. Respiratory ailments, via inhalation of aerosols formed when water-skiing, surfing or by nearby breaking waves.

Table 3-1 lists potential waterborne diseases and their aetiological agents (i.e., pathogens), derived from the ANZECC guidelines (ANZECC & ARMCANZ 2000). It also indicates whether our assessment of the particular pathogen should be based on contact recreation or shellfish consumption exposure routes, and gives a brief rationale for this assessment.

**Table 3-1: Screening of treated wastewater-borne microorganisms of public health significance.**

| Pathogen | Include? | Main disease caused | Rationale |
|---|---|---|---|
| **Bacteria** | | | |
| *Campylobacter spp*. | No | Gastroenteritis | Poor survival in seawater |
| Pathogenic *E. coli* | No | Gastroenteritis | Low concentration expected in treated wastewater |
| *Legionella pneumophila* | No | Legionnaires' disease | No evidence of environmental infection route |
| *Leptospira* sp. | No | Leptospirosis | Low concentration expected in treated wastewater |
| *Salmonella* sp. | No | Gastroenteritis | Low concentration expected in treated wastewater |
| *Salmonella typhi* | No | Typhoid fever | Rare in New Zealand |
| *Shigella* sp. | No | Dysentery | Low concentration expected in treated wastewater |
| *Vibrio cholerae* | No | Cholera | Rare in New Zealand |
| *Yersinia enterolitica* | No | Gastroenteritis | Low concentration expected in treated wastewater |
| **Helminths** | | | |
| *Ascaris lumbricoides* | No | Roundworm | Rare in New Zealand |
| *Enterobius vernicularis* | No | Pinworm | Low concentration expected in treated wastewater |
| *Fasciola hepatica* | No | Liver fluke | Rare in New Zealand |
| *Hymnolepis nana* | No | Dwarf tapeworm | Rare in New Zealand |
| *Taenia* sp. | No | Tapeworm | Rare in New Zealand |
| *Trichuris trichiura* | No | Whipworm | Rare in New Zealand |
| **Protozoa** | | | |
| *Balantidium coli* | No | Dysentery | Low concentration expected in treated wastewater |
| *Cryptosporidium* oocysts | No | Gastroenteritis | Will be removed by proposed wastewater treatment processes |
| *Entamoeba histolytica* | No | Amoebic dysentery | Rare in New Zealand |
| *Giardia* cysts | No | Gastroenteritis | Moderate survival in seawater but will be removed by proposed wastewater treatment processes. |
| **Viruses** | | | |
| Adenoviruses | Yes (SW only)[3] | Respiratory disease[4] | Very infective.  Significant concentrations may be present in wastewater |
| Enteroviruses | Yes (SW and SF) | Gastroenteritis | Less infective, but health consequences can be more severe than for exposure to adenovirus |
| Hepatitis A virus | No | Infectious hepatitis | Minimal concentration in treated wastewater; very infective. Can affect recreational water users in contaminated waters |
| Noroviruses | Yes, exploratory only (SW & SF) | Gastroenteritis | Increasing evidence of its prevalence in treated wastewater. Clinical trials and dose-response now available. However, it hasn't been possible to culture in the laboratory until now.[5] This makes assessment of treatment efficacy problematic. |
| Rotavirus | No | Gastroenteritis | Limited evidence of waterborne infection in NZ; infection in children would be of concern.[6] Difficult to translate units used in clinical trial (Focus Forming Units, FFU, Ward et al. 1986) to those used in culture methods. |

In general terms, for sites impacted by WWTPs processing well-treated human-derived wastewater (e.g., Mangere WWTP), there is widespread agreement that human viruses are the principal aetiological agent causing gastrointestinal disease among water users and consumers of raw shellfish (Lodder and de Roda Husman 2005; Sinclair et al. 2009).[7] Viruses are also more difficult to remove through wastewater treatment processes, and are therefore the focus of this QMRA. More information regarding candidate viruses is included in Table A-1.

### Gastrointestinal illness

Enteroviruses (coxsackie virus and echovirus) are the pathogen-of-choice, for three reasons:

1. Their evaluation is by culture, whereas noroviruses to date have had to be analysed by qPCR methods,[8] and the ratio of infectious/total virus numbers can be expected to vary through the wastewater treatment process.
2. Enteroviruses can cause longer-term illnesses.
3. Clinical trial data and associated infection dose-response relationships based on culture methods are available and have already been used for the health risk assessment associated with the Manukau Wastewater Treatment Plant (DRG 2002; Simpson et al. 2003).

Noroviruses have also been included in an exploratory mode, recognising that while they are often held to be the main aetiological agent for health risk following exposure to waters containing human-derived treated wastewater residues, their enumeration poses difficulties in terms of assessing WWTP removal efficacy and subsequent infectivity ((da Silva et al. 2007; Hewitt et al. 2011; Sima et al. 2011; McBride 2014). QMRAs based on noroviruses have been conducted elsewhere in New Zealand, e.g., Napier and New Plymouth (McBride 2011; McBride 2012).[9] We assume that the removal of noroviruses through the WWTP will be at least as effective as that inferred for enteroviruses.

### Respiratory illness

For this illness category we have only one choice: adenovirus. We are not aware of any other respiratory agents, appropriate to treated wastewater, for which dose-response information is available. Its merits and drawbacks are listed in Appendix B.

### Other pathogens

Wastewater treatment specialists raised concerns regarding *Pseudomonas aeruginosa*, which is resistant to wastewater treatment and disinfection processes, and may survive treatment and be discharged to the environment. In addition, some in the community raised concerns regarding *Staphylococcus aureus*, and there was a belief that there was a potential for infection as a

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[3] "SW" = swimming; "SF" = shellfish.
[4] Adenoviruses can also cause pneumonia, eye infections and gastroenteritis.
[5] A new culture-based method has recently been published—Jones et al. (2014): http://www.ncbi.nlm.nih.gov/pubmed/25378626.
[6] Rose & Sobsey (1993) have documented a rationale for concern about potential contamination of shellfish by rotavirus, but risk appears to have been over-estimated (they equated FFU with actual numbers of virions).
[7] This is not necessarily true for agricultural wastes in rural settings, where bacteria and protozoa predominate—with few exceptions (hepatitis E, some rotaviruses), animal viruses are not pathogenic to humans.
[8] "qPCR" refers to quantitative Polymerase Chain Reaction, a molecular laboratory test that essentially counts the number of virions in a sample, whether infectious or inactivated.
[9] "Norovirus" subsumes the term "Norwalk virus". The clinical trial reported and analysed by Teunis et al. (2008) was for the original Norwalk virus (genotype group GI.1)—it had been stored in a laboratory for some years. Since the time of the first identified norovirus outbreak (in Norwalk, Ohio, 1968) a number of similar caliciviruses have been identified, in genogroups I–V. Current practice is to regard the infectivity of GI.1 norovirus as equivalent to all noroviruses that affect humans (particularly GI and GII).

consequence of physical abrasion and skin damage incurred on rocks in the vicinity of the existing discharge site. As a consequence, these organisms were also considered in the risk assessment.

## 3.2 Selected pathogens

The candidate pathogens selected for the QMRA were: (for which some form of identified dose-response curve is available) are summarised in Appendix A and Appendix B.

- **Adenovirus** – very resistant to disinfection, and a common cause of gastrointestinal illness; it is also able to cause respiratory infections, as a consequence of aerosol inhalation.
- **Enterovirus** – this is highly contagious, and inclusion is warranted given that it can cause more serious longer-term illnesses.
- **Norovirus** is included as a representative "model" virus as well:
  - Norovirus is reported to be the most common aetiological agent in receiving waters (e.g., Sinclair et al. 2009).
  - Infection $ID_{50}$ is in the order of 26 virions (among susceptible people), but the dose-response curve indicates that ~20% of people may become infected after ingestion of just one virion.
- *Pseudomonas aeruginosa.*
- *Staphylococcus aureus*.

## 3.3 Assessing exposure - predicting doses

To turn concentrations into doses we need:

1. Wastewater virus concentrations.
2. Ingestion or inhalation rates for recreational users exposed to contaminated waters.
3. Bioaccumulation factors for shellfish.

Details on how these factors have been modelled and enumerated are given in Table 3-2. Water ingestion rates by swimmers—a key component of dose-calculation—were derived from a clinical trial involving 53 volunteers involved in recreational swimming in an outdoor community swimming pool (Dufour et al. 2006). The volume of water ingested during swimming events lasting at least 45 minutes was calculated for each swimmer using a chemical tracer. It has become standard practice to apply these ingestion rates to water recreation.[10]

[10] Personal communication: Jeff Soller, Soller Environmental, California (http://www.sollerenvironmental.com/env/main/Home.html).

The focus on "primary contact recreation" does not imply that exposure through other forms of recreation does not create risk.  The health risks associated with paddle-boarding or canoeing are likely to be lower (there is little opportunity for ingestion), unless the individual capsizes or falls into the water.  At such time, similar ingestion rates are likely as for a swimmer, but for shorter exposure periods.  If the individual remains in the water for a longer period, then both ingestion rate and duration of exposure are likely to be similar to those of the swimmer.  The swimming health risk is therefore a reasonable (and conservative) surrogate for other recreational users.

**Table 3-2: Distributions and inputs for the QMRA.** Plain numbers in the Statistics column are for typical health conditions in the local community; italicised numbers are for the rare case where there is a norovirus illness outbreak in that community. [a]

| Component | Statistics | Distributions/comments |
|---|---|---|
| Influent virus concentration | | Bounded “hockey stick” distribution (McBride 2005), strongly right-skewed with a hinge at the 95%ile. |
| Influent enterovirus concentration, per litre | Minimum = 500<br>Median = 4,000<br>Maximum = $5 \times 10^7$ | Mimicking high values found for Mangere influent in a “Scoping study” in May-July 1999 (Table B1, DRG 2002, where missing values for 24 & 26 May were advised by Mr Peter Loughran, MWH, on 7 11/2003—these values are plotted on Figure 3.3.5 of the DRG report). Most usually the concentrations are 1,000–10,000 per litre (DRG 2002, Table B6).[a] |
| Influent adenovirus concentration, per litre | Minimum = 2,000<br>Median = 5,000<br>Maximum = $3 \times 10^7$ | Rationale as above. Most usually the concentrations are 1,000–10,000 per litre (DRG 2002, Table B6): 10% of these concentrations are assumed infectious for respiratory illness effects (Kundu et al. 2013 have noted that a minority of adenovirus strains cause respiratory illness). |
| Influent norovirus concentration, genome copies per litre | Minimum = $10^2$<br>Median = $10^4$<br>Maximum = $10^7$ | Typical range found for New Zealand cities (e.g., Napier, New Plymouth—McBride 2011, 2012, (McBride 2016)). |
| Duration of swim (hours) | Minimum = 0.1<br>Median = 0.25<br>Maximum = 2 | Child or adult. |
| Swimmers water ingestion rate, mL per hour | Minimum = 20<br>Median = 50<br>Maximum = 100 | PERT distribution, for a child (adult rate is half this rate). For a review on this see Wood et al. (2015, sec. 6.2.1). |
| Water inhalation rate, mL per hour | Minimum = 10<br>Median = 25<br>Maximum = 50 | PERT distribution. Assumed to be half the child ingestion rate. |
| Dose-response parameters | – | ▪ Adenoviruses, simple binomial [eq. (4)]; $r$ = 0.4142 (so $ID_{50,infection} \approx 2$), Pr(ill \| Infection) = 0.5 (Soller et al. 2010),<br>▪ Enterovirus, beta-binomial [eq. (5)]: $\alpha$ = 1.3, $\beta$ = 75 (so $ID_{50,infection}$ = 53); Pr(ill \| Infection) = 1.<br>▪ Norovirus, beta-binomial [eq. (5)]: $\alpha$ = 0.04, $\beta$ = 0.055 (so $ID_{50,infection}$ = 26); Pr(susceptible) = 0.74 (Teunis et al. 2008); Pr(ill \| Infection) = 0.60 (Soller et al. 2010). |
| Shellfish meal size | $\alpha$ = 2.2046<br>$\beta$ = 75.072<br>$\gamma$ = –0.903 | Using a log logistic distribution, truncated below at 5 g and above at 800 g, obtained by fitting distributions to estimates of daily intake of 98 consumers of mussels, oysters, scallops, pipi and tuatua in the 1997 National Nutrition Survey (Russell et al. 1999, (McBride 2005)). |
| Bioaccumulation factor | Mean = 49.9<br>Std. dev. = 20.93 | Using normal distributions, truncated at 1 and 100. The pathogen dose ingested on eating 100 grams of shellfish is BAF x the number of pathogens in the equivalent volume of water (Burkhardt and Calci 2000). The chosen factors are for $F^+$ coliphage in winter. The use of a normal distribution for BAFs allows half of these factors to be below 50 yet retain a precautionary approach. |

## 3.4 Characterising dose-response

These relationships are mostly inferred from data reported by "volunteer studies" (i.e., clinical trials). These have been done for a restricted number of viruses. In these studies, healthy adult volunteers (typically between 50 and 100, in groups of 10 or so) are individually challenged with a pathogen dose and their infection and illness states are monitored for a few days thereafter. Such a study has been conducted for noroviruses (Teunis et al. 2008). Occasionally data from viral illness outbreaks become available, from which dose-response information can be inferred.[11] Note that to perform QMRA calculations, comparability between the definition of "dose" used in the clinical trial or outbreak study and the methods used in assessing virus concentrations in the wastewater of concern is required. For example, when assessing pathogens in treated wastewater, noroviruses cannot be cultured, so a quantitative molecular-based laboratory procedure (Reverse Transcription Polymerase Chain Reaction "RT-qPCR") is used to detect the norovirus genome. Since RT-qPCR detects genetic material, the method picks up both viable and non-viable viruses. Since there are variants of the qPCR procedure, some harmonisation between the methods used in the clinical trial and wastewater Norovirus enumeration methods may be required (and is so in this study).

## 3.5 Conducting the health risk assessment

To adequately reflect limits to knowledge on key features of the risk assessment, Monte Carlo statistical modelling is used (Haas et al. 1999; McBride 2005). In simpler models, key inputs are described by a single number (e.g., wastewater treatment plant (WWTP) influent pathogen concentration). However, such inputs are known to be variable and some are uncertain. The way this variability and uncertainty has been addressed is shown in Table 3-2. The proprietary Excel plug-in product "@RISK" was used to perform the calculations, incorporating factors that reflect these distributions and inputs (Palisade Corp 2013).[12] The models were run for 10,000 iterations for the selected virus, for the proposed virus concentration distribution, and for each of four dilution scenarios. During each iteration, 100 individuals were 'exposed', by taking a random sample from statistical distributions covering the range of possible doses received by individuals ingesting water possibly containing pathogen.

It can be appropriate to report the results in terms of infection (which is the approach taken for the freshwater component of the MfE/MoH Guidelines), rather than illness. For the present study where Norovirus is the model pathogen, we take standard values of the probability of illness, given that infection has occurred. The output metric is an individual's illness risks, to facilitate comparison with relevant guidelines. [13,14] We do however account for "aggregation" – clumping together of viral particles to form a single infectious mass, rather than existing as several or many discrete particles. The extent and likelihood of aggregation is determined by the presence and amount of organic matter able to facilitate attraction between and binding of these infectious agents.

[11] An example is a study by Thebault et al. (2013) of norovirus illness outbreaks among consumers of oysters in southern France.

[12] The @RISK models use named cells as much as possible, to facilitate checking and readability.

[13] There is insufficient time and information to also compute DALY metrics (Disability-Adjusted Life Years) as often used when assessing health risks associated with drinking-water (WHO 2011, chapter 7).

[14] The individual's illness risk (IIR) is calculated as the total number of predicted illness cases divided by the total number of exposures to potentially contaminated water or shellfish flesh. It represents the risk to an individual swimmer or shellfish consumer on any day, having no prior knowledge of any contamination from the outfall. It is calculated via the Monte Carlo modelling, for which 100 individuals are exposed on each of 1,000 separate days, i.e., $10^5$ exposures. The total number of cases is $1{,}000m$ where $m$ is the mean infection case rate over 100 people (readily calculated by the Monte Carlo software—@RISK, Palisade Corp. 2013). So the individual's infection risk, expressed as a proportion, is $1{,}000m/10^5 = m/100$. When expressed as a percentage, IIR = $m$%.

## 3.6 Exposure assessment sites

A recreational survey was undertaken (Reference), which provided information regarding the various activities undertaken in and along the shoreline of Raglan Harbour, as well as in waters and along the Tasman Sea shoreline within a radius of approximately 3,000 m of the harbour mouth. These activities included varied recreational water uses (swimming, surfing, windsurfing, kitesurfing, kayaking, paddle-boarding and sailing), as well as shellfish gathering.

Based on the outcomes of the surveys, individual sites were selected as candidates for quantitative health assessment. These were separated into contact recreation (15 sites) and shellfish gathering sites (14 sites). Several recreation and shellfish gathering sites coincide. Site descriptions are provided in Table 3-3 (contact recreation), Table 3-4 (shellfish gathering/ consumption sites), and the location of these sites is indicated in Figure 3-2 and Figure 3-3 respectively.

**Table 3-3: Description of sites used for contact recreation health risk assessment.** The location of the sites is indicated in Figure 3-2.

| Label | Site and use | Grid reference (NZTM) | |
|---|---|---|---|
| | | East | North |
| **R1** | Kite Surf Inner | 1762419 | 5814417 |
| **R2** | Inshore Kite surf | 1762288 | 5814526 |
| **R3** | Entrance kite surf | 1761619 | 5814656 |
| **R4** | Northern swimming | 1761055 | 5813536 |
| **R5** | Northern surfing | 1760986 | 5813623 |
| **R6** | Bar surf | 1760317 | 5813961 |
| **R7** | Offshore kite surf/Maui | 1761064 | 5814864 |
| **R8** | Western Swimming and Shellfish in Harbour | 1763494 | 5814317 |
| **R9** | Domain Recreation/Shellfish | 1764336 | 5814604 |
| **R10** | Maui North | 1761081 | 5815498 |
| **R11** | Kite Surf B | 1762418 | 5814413 |
| **R12** | Kite Surf C | 1762262 | 5814482 |
| **R13** | Kite Surf D | 1761914 | 5814560 |
| **R14** | Kite Surf E | 1761368 | 5814656 |
| **R15** | Kite Surf F | 1761098 | 5814864 |

**Table 3-4: Description of sites used for shellfish consumption health risk assessment.** The location of the sites is indicated in Figure 3-3.

| Label | Site and use | Grid reference (NZTM) | |
|---|---|---|---|
| | | East | North |
| **S1** | Eastern end of tuatua | 1762114 | 5814352 |
| **S2** | Mid-point of tuatua | 1761888 | 5814178 |
| **S3** | Mussel Rocks | 1761368 | 5816861 |
| **S4** | Western Cockle/Pipi in Harbour | 1762947 | 5814230 |
| **S5** | Western Swimming and Shellfish in Harbour | 1763494 | 5814317 |
| **S6** | Western Shellfish in Harbour A | 1763538 | 5814638 |
| **S7** | Western Shellfish in Harbour B | 1763512 | 5815020 |
| **S8** | Mid Harbour Shellfish | 1764024 | 5815628 |
| **S9** | Inner Harbour Shellfish A | 1764831 | 5816262 |
| **S10** | Inner Harbour Shellfish B | 1765126 | 5816036 |
| **S11** | Inner Harbour Shellfish C | 1764996 | 5815750 |
| **S12** | Inner Harbour Shellfish D | 1764336 | 5815194 |
| **S13** | Domain Recreation/Shellfish | 1764336 | 5814604 |
| **S14** | Inner Harbour Shellfish | 1764206 | 5814968 |

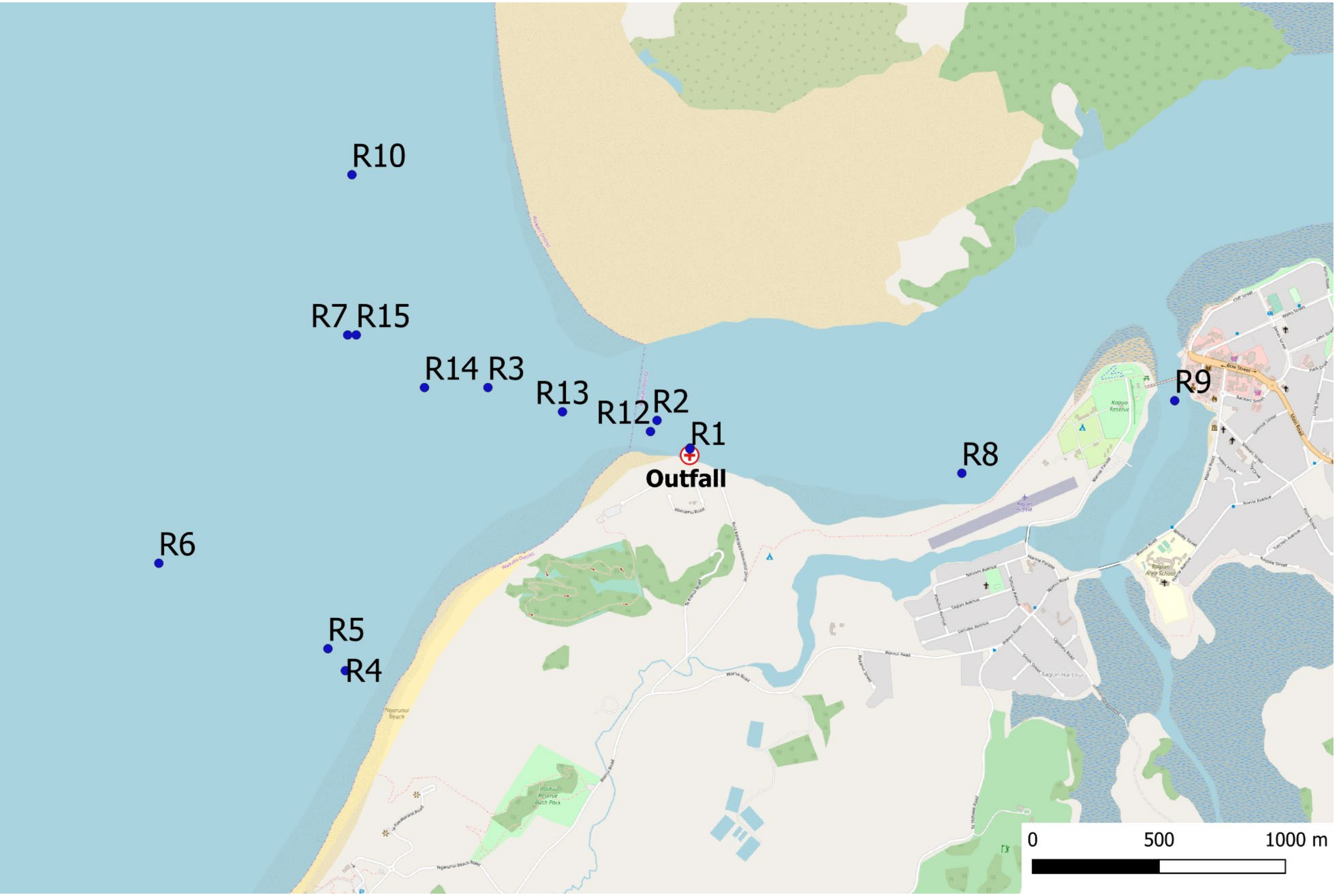


**Figure 3-2: Location of contact recreation sites at which human health risks were estimated in Raglan Harbour.** The location of site R11 is partially overlain by that of site R1 and the Outfall site. The site details are listed in Table 3-3.

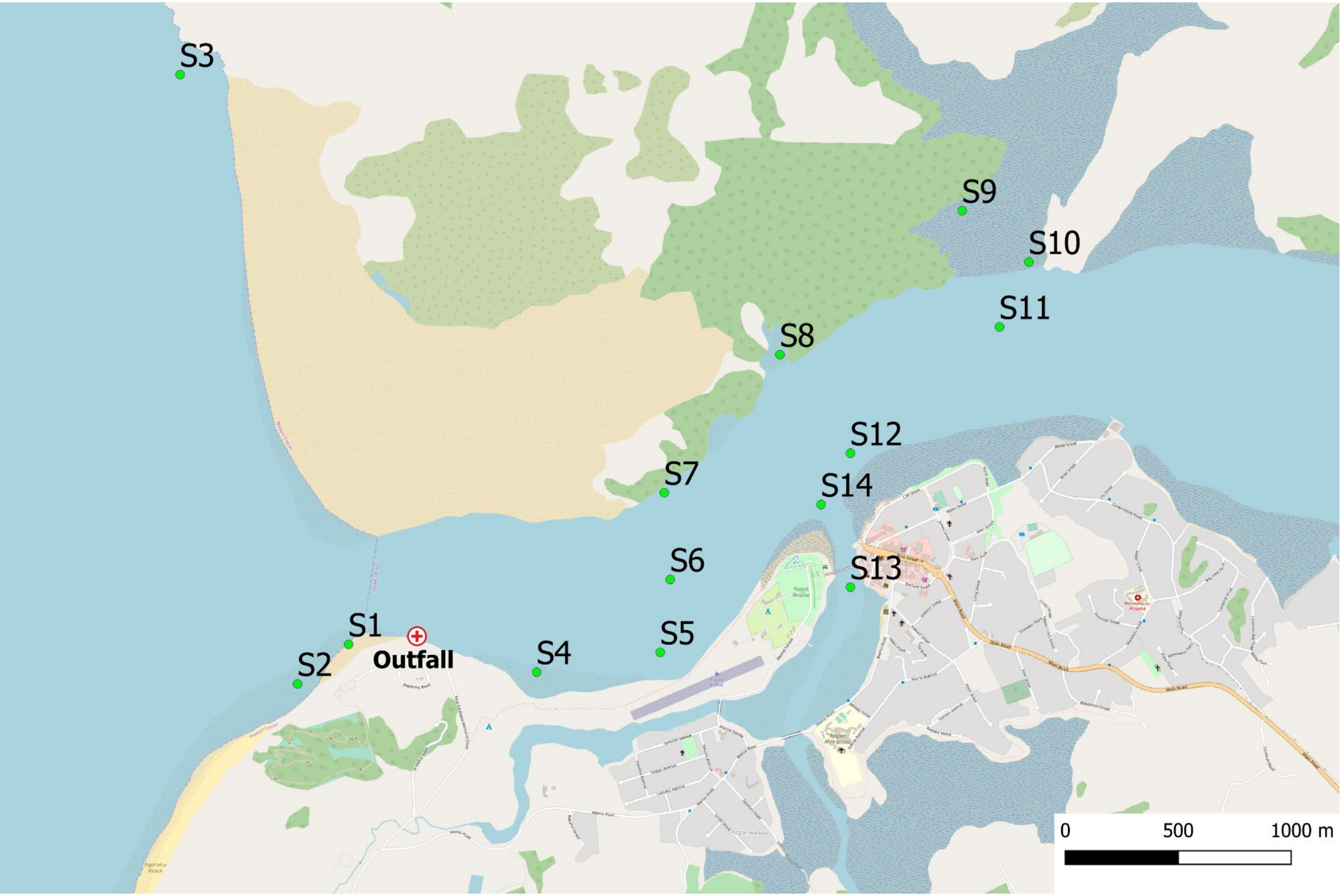


**Figure 3-3: Location of shellfish gathering sites at which human health risks were estimated in Raglan Harbour.** The site details are listed in Table 3-4.

# 4 Public health risk estimates

## 4.1 Description of risk estimation process

The QMRA process generates statistical distributions of risk estimates from which the probability of infection or illness may be estimated. These risks are generalised in the sense that they represent the risks applicable to any random group of water users or shellfish consumers engaged in contact recreation or consuming shellfish on any day of the year. These models do not represent the illness or infection risk on any specific day, or for a specific set of discharge, meteorological or environmental conditions. The risks generated are termed incremental risks – risks associated only with pathogens or microbiological hazards associated with the treated wastewater discharge, and for the specific model pathogens selected. They do not account for pathogens derived from other sources, which in the case of Raglan Harbour may include stormwater or discharges from overloaded sewers in urban areas, as well as runoff from agricultural lands. The latter may include pathogens such as *Campylobacter* sp. (Davies-Colley and Nagels 2002; McKergow and Davies-Colley 2010; Stott et al. 2017).

As part of the risk estimation process, levels of virus removal are assumed for wastewater treatment processes, and risks are calculated for each of them. For this study, four levels of wastewater treatment were considered: 10-fold, 100-fold, 1,000-fold, and 10,000-fold virus removal. In engineering and science studies these levels are commonly denoted as "log removals" (log being shorthand for logarithms to base ten, or $\log_{10}$). In this case, the four levels of removal or treatment efficacy correspond to the $\log_{10}$ removals of 1, 2, 3, and 4. Essentially, the log number is the number of zeroes in the removal efficacy figures. For example 1,000-fold removal is "log 3". For the sake of convenience (and convention), we may omit the superscript indicating $\log_{10}$, and refer to "log 3" or similar.

## 4.2 Inclusion of tracers in the hydrodynamic modelling

The hydrodynamic modelling was described separately in DHI (2019). Briefly, the process accounts for nearfield turbulence, winds, tides, large-scale currents and significant water discharges (rivers, streams and the wastewater discharge) to represent the mixing and dispersion of treated wastewater in the model domain (which includes the entire Raglan Harbour and substantial area within the Tasman Sea off shore the harbour mouth.

During the modelling process, a constant "load" of tracer is input with the wastewater, and the fate of these tracers is used to represent what is likely to happen to actual pathogens. This allows estimation of the concentration of tracers at exposure sites (where contact recreation or shellfish gathering is likely to occur). In the risk assessment process these shoreline or site-specific concentrations are related to the initial discharge concentrations and the ratio (the effective dilution predicated at each site) is used in the risk assessment process.

Three tracers were incorporated in the hydrodynamic modelling. These tracers included:

- a conservative tracer (which assumes that no inactivation or sedimentation of microbial species occurs)
- an enterococci tracer (which represent the dominant processes and rates at which these processes impact on these organisms), and

- a representative virus (with appropriate inactivation processes and rates).

The relative magnitude of dilutions derived from these three tracers are summarised in Figure 4-1.

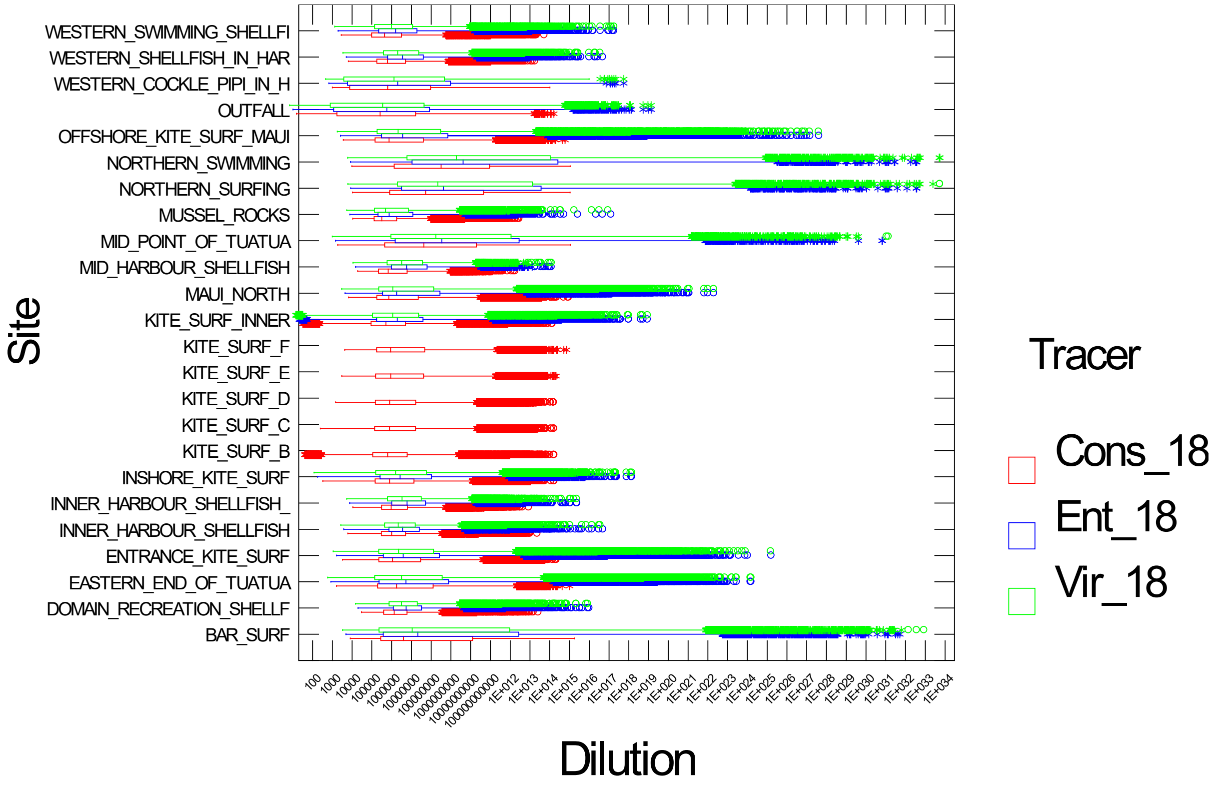


**Figure 4-1: Dilutions of pathogens predicted by three tracers at recreational and shellfish gathering sites.** Full site details are listed in Table 3-3 and Table 3-4. Large dilutions indicate low concentrations. “Cons_18”, “Ent_18” and “Vir_18” indicate conservative, enterococci and viral tracers for 2018 wastewater discharge conditions respectively. The median dilution is indicated by the line in the box of the box plot. A full description of a box plot is provided in Appendix D.

Use of tracers allow the effect of processes such as UV inactivation, and flocculation and sedimentation to be represented. Figure 4-1 indicates that median dilutions are generally Enterococci > Virus > Conservative tracer, with a smaller number of very large dilution values for the conservative tracer at all sites. In hydrodynamic modelling exercises for relatively poorly flushed, shallow environments such as harbours and tidally dominated inlets, use of conservative tracers sometimes leads to “accumulation” of contaminants, evident as increasing concentrations over time. This generally occurs because the model does not include or removal processes, and the mass of contaminant ins not flushed from the harbour during each tidal cycle. This does not appear to be the case for the Raglan Harbour model, where little evidence of contaminant accumulation is evident either at the harbour mouth, or at a relatively poorly diluted site in the inner harbour (Figure 4-2).

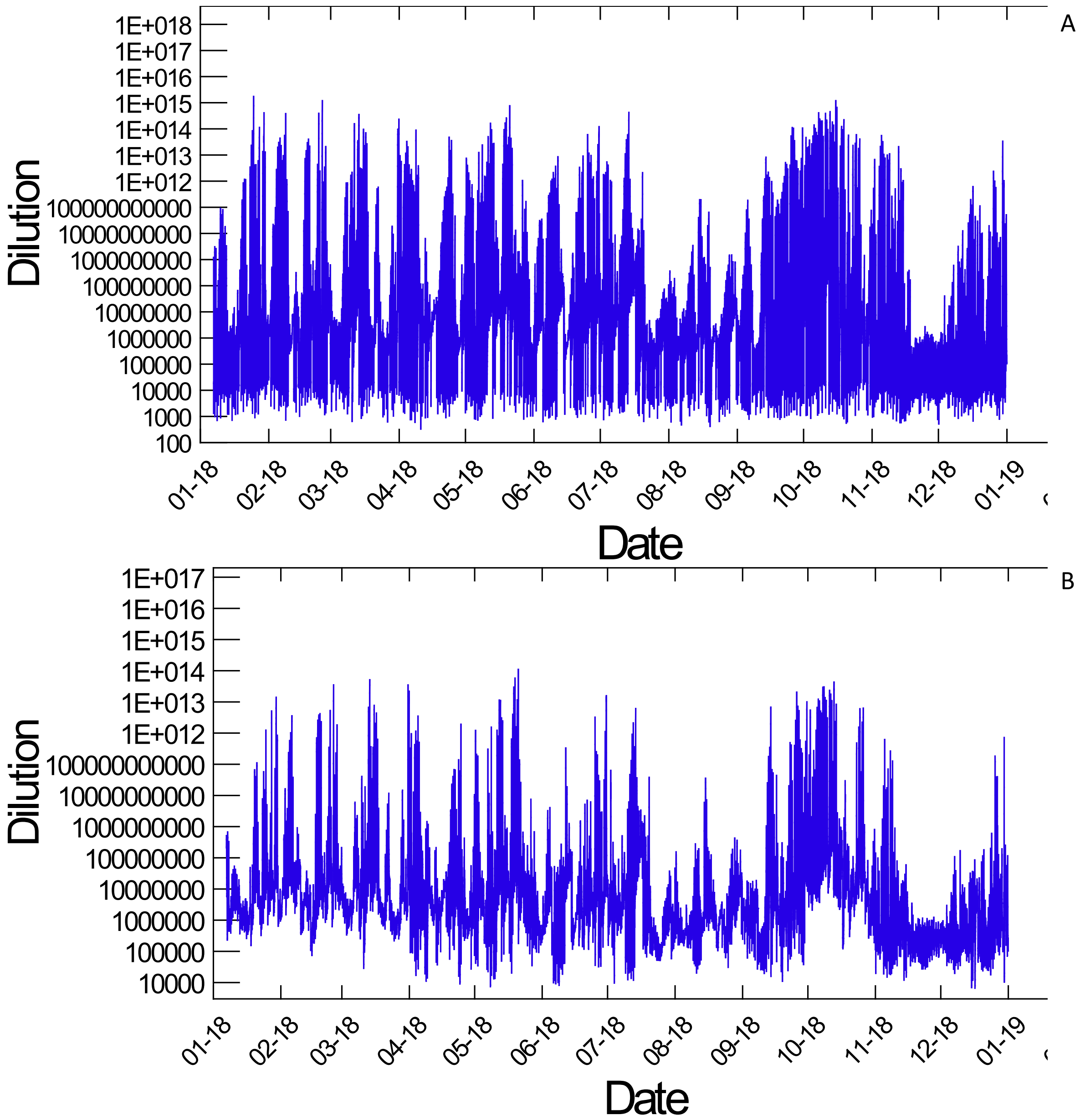


**Figure 4-2: Dilutions of pathogens predicted by a conservative tracer at a recreational and shellfish gathering site.** A) shows dilutions for site R12, a recreational site in the Harbour mouth on the seaward side of the wastewater discharge, and B) is for a mid-inner harbour shellfish site (S11). Full site details are listed in Table 3-3 and Table 3-4. Large dilutions indicate low concentrations.

In this optioneering exercise, we estimated risks using the conservative tracer dilution values only. These provided conservative risk estimates (worst case risks) at all sites, for both contact recreation and shellfish gathering, but with assurance that the model was not accumulating pathogens in the harbour over time.

## 4.3 Presentation of results

The risk estimation process generates a large volume of data.

**For swimming risks** (mean IIR or IInR only), the following scenarios were modelled:

> 16 sites x 4 treatment efficacies x 2 receptors x 3 pathogen x 2 risk types = 768 results

**For shellfish consumption risks** (mean IIR or IInR only), modelled scenarios included:

> 14 sites x 4 treatment efficacies x 1 receptor x 2 pathogen x 2 risk types= 224 results

Potentially the number of results could be three times greater because three tracers were used to represent the mixing and advection of microbial species across the model domain. We focussed on the results derived from use of a conservative tracer (discussed earlier in Section 4.2).

Selected illness and infection risks arising from contract recreation at 15 exposure sites and the Outfall site are displayed in Table 4-1; selected illness and infection risks arising from consuming shellfish gathered at 14 sites and the Outfall site are displayed in Table 4-2. These results assume one log wastewater treatment efficacy. These represent risks remaining following a very modest standard of wastewater treatment. This was appropriate because of the generally very low levels of risk that prevailed at most sites. These results were also summarised graphically. For recreational health risks (arising from contact recreation):

- Figure 4-3 - Figure 4-8 show the magnitude of child and adult illness and infection risks for three viral pathogens spatially, representing relative risk using bar symbols at the site where contact recreation is known to occur.
- Figure 4-11 - Figure 4-14 show the magnitude of child and adult illness and infection risks for three viral pathogens according to treatment efficacy (1-4 log removal).
- Figure 4-9 - Figure 4-10 show the magnitude of adult illness and infection risks for shellfish consumers for two viral pathogens spatially, representing relative risk using bar symbols at the site where shellfish gathering is known to occur.
- Figure 4-15 - Figure 4-16 show the magnitude of adult illness and infection risks for shellfish consumers for two viral pathogens according to treatment efficacy (1-4 log removal).

These data may also be considered in terms of relative illness or infection risk presented by various pathogens at each exposure site, as well as the impact that efficacy of wastewater treatment has on human health risk. In Figure 4-17, the relative illness and infection risk presented by the three model pathogens is summarised at each exposure site for child and adult targets for wastewater treated to one log efficacy. In Figure 4-18, the effect that increasing levels of wastewater treatment has on child infection risk is summarised for the three target pathogens.

These results are briefly discussed in Section 5.

**Table 4-1: Infection and illness health risk for child and adult receptors at recreation sites for three viral pathogens given one-log treatment efficacy.** NoV = Norovirus, AdV = Adenovirus, EnV = Enterovirus. The location of the sites is indicated in Figure 3-2.

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk according to virus (%) | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Infection | Child | R1 | Kite Surf Inner | 1 | 0.8045 | 1.2879 | 0.3958 |
| Infection | Child | R2 | Inshore Kite surf | 1 | 0.2138 | 0.6003 | 0.0561 |
| Infection | Child | R3 | Entrance kite surf | 1 | 0.0794 | 0.2749 | 0.0183 |
| Infection | Child | R4 | Northern swimming | 1 | 0.0261 | 0.0948 | 0.0048 |
| Infection | Child | R5 | Northern surfing | 1 | 0.0221 | 0.0887 | 0.0045 |
| Infection | Child | R6 | Bar surf | 1 | 0.0824 | 0.3058 | 0.0178 |
| Infection | Child | R7 | Offshore kite surf/Maui | 1 | 0.0830 | 0.3192 | 0.0157 |
| Infection | Child | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.1190 | 0.4257 | 0.0244 |
| Infection | Child | R9 | Domain Recreation/ Shellfish | 1 | 0.0445 | 0.1956 | 0.0067 |
| Infection | Child | R10 | Maui North | 1 | 0.0560 | 0.2268 | 0.0098 |
| Infection | Child | R11 | Kite Surf B | 1 | 0.7322 | 1.1353 | 0.3386 |
| Infection | Child | R12 | Kite Surf C | 1 | 0.3455 | 0.8269 | 0.1335 |
| Infection | Child | R13 | Kite Surf D | 1 | 0.0901 | 0.3253 | 0.0203 |
| Infection | Child | R14 | Kite Surf E | 1 | 0.0836 | 0.3090 | 0.0178 |
| Infection | Child | R15 | Kite Surf F | 1 | 0.0830 | 0.3192 | 0.0157 |
| Infection | Child | Outfall | Outfall | 1 | 1.5511 | 2.2693 | 0.7213 |
| Infection | Adult | R1 | Kite Surf Inner | 1 | 0.5735 | 1.0393 | 0.2854 |
| Infection | Adult | R2 | Inshore Kite surf | 1 | 0.1330 | 0.4086 | 0.0301 |
| Infection | Adult | R3 | Entrance kite surf | 1 | 0.0492 | 0.1645 | 0.0095 |
| Infection | Adult | R4 | Northern swimming | 1 | 0.0134 | 0.0529 | 0.0022 |
| Infection | Adult | R5 | Northern surfing | 1 | 0.0097 | 0.0475 | 0.0022 |
| Infection | Adult | R6 | Bar surf | 1 | 0.0423 | 0.1747 | 0.0079 |
| Infection | Adult | R7 | Offshore kite surf/Maui | 1 | 0.0460 | 0.1870 | 0.0095 |
| Infection | Adult | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.0627 | 0.2485 | 0.0124 |
| Infection | Adult | R9 | Domain Recreation/ Shellfish | 1 | 0.0227 | 0.0991 | 0.0030 |
| Infection | Adult | R10 | Maui North | 1 | 0.0288 | 0.1209 | 0.0039 |
| Infection | Adult | R11 | Kite Surf B | 1 | 0.5123 | 0.9009 | 0.2446 |
| Infection | Adult | R12 | Kite Surf C | 1 | 0.2418 | 0.6370 | 0.0810 |
| Infection | Adult | R13 | Kite Surf D | 1 | 0.0514 | 0.1940 | 0.0111 |
| Infection | Adult | R14 | Kite Surf E | 1 | 0.0435 | 0.1725 | 0.0084 |
| Infection | Adult | R15 | Kite Surf F | 1 | 0.0460 | 0.1870 | 0.0095 |
| Infection | Adult | Outfall | Outfall | 1 | 1.0626 | 1.7788 | 0.5450 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk according to virus (%) | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Illness | Child | R1 | Kite Surf Inner | 1 | 0.3002 | 0.6418 | 0.3958 |
| Illness | Child | R2 | Inshore Kite surf | 1 | 0.0767 | 0.2965 | 0.0561 |
| Illness | Child | R3 | Entrance kite surf | 1 | 0.0310 | 0.1414 | 0.0183 |
| Illness | Child | R4 | Northern swimming | 1 | 0.0090 | 0.0453 | 0.0048 |
| Illness | Child | R5 | Northern surfing | 1 | 0.0079 | 0.0444 | 0.0045 |
| Illness | Child | R6 | Bar surf | 1 | 0.0329 | 0.1558 | 0.0178 |
| Illness | Child | R7 | Offshore kite surf/Maui | 1 | 0.0307 | 0.1640 | 0.0157 |
| Illness | Child | R8 | Western Swimming & Shellfish in Harbour | 1 | 0.0451 | 0.2142 | 0.0244 |
| Illness | Child | R9 | Domain Recreation/ Shellfish | 1 | 0.0184 | 0.1042 | 0.0067 |
| Illness | Child | R10 | Maui North | 1 | 0.0202 | 0.1125 | 0.0098 |
| Illness | Child | R11 | Kite Surf B | 1 | 0.2736 | 0.5757 | 0.3386 |
| Illness | Child | R12 | Kite Surf C | 1 | 0.1226 | 0.4123 | 0.1335 |
| Illness | Child | R13 | Kite Surf D | 1 | 0.0341 | 0.1605 | 0.0203 |
| Illness | Child | R14 | Kite Surf E | 1 | 0.0332 | 0.1566 | 0.0178 |
| Illness | Child | R15 | Kite Surf F | 1 | 0.0307 | 0.1640 | 0.0157 |
| Illness | Child | Outfall | Outfall | 1 | 0.5738 | 1.1367 | 0.7213 |
| Illness | Adult | R1 | Kite Surf Inner | 1 | 0.4224 | 0.5178 | 0.2854 |
| Illness | Adult | R2 | Inshore Kite surf | 1 | 0.0951 | 0.2026 | 0.0301 |
| Illness | Adult | R3 | Entrance kite surf | 1 | 0.0365 | 0.0861 | 0.0095 |
| Illness | Adult | R4 | Northern swimming | 1 | 0.0095 | 0.0249 | 0.0022 |
| Illness | Adult | R5 | Northern surfing | 1 | 0.0072 | 0.0242 | 0.0022 |
| Illness | Adult | R6 | Bar surf | 1 | 0.0333 | 0.0901 | 0.0079 |
| Illness | Adult | R7 | Offshore kite surf/Maui | 1 | 0.0349 | 0.0963 | 0.0095 |
| Illness | Adult | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.0247 | 0.1251 | 0.0124 |
| Illness | Adult | R9 | Domain Recreation/ Shellfish | 1 | 0.0100 | 0.0545 | 0.0030 |
| Illness | Adult | R10 | Maui North | 1 | 0.0101 | 0.0606 | 0.0039 |
| Illness | Adult | R11 | Kite Surf B | 1 | 0.1913 | 0.4567 | 0.2446 |
| Illness | Adult | R12 | Kite Surf C | 1 | 0.0872 | 0.3143 | 0.0810 |
| Illness | Adult | R13 | Kite Surf D | 1 | 0.0186 | 0.0997 | 0.0111 |
| Illness | Adult | R14 | Kite Surf E | 1 | 0.0173 | 0.0878 | 0.0084 |
| Illness | Adult | R15 | Kite Surf F | 1 | 0.0180 | 0.0963 | 0.0095 |
| Illness | Adult | Outfall | Outfall | 1 | 0.7854 | 0.8963 | 0.5450 |

**Table 4-2: Infection and illness health risk at shellfish harvest sites for two viral pathogens given one-log treatment efficacy.** NoV = Norovirus, EnV = Enterovirus. The location of the sites is indicated in Figure 3-3.

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk according to virus (%) | |
|---|---|---|---|---|---|---|
| | | | | | NoV | EnV |
| Illness | Adult | S1 | Eastern end of tuatua | 1 | 0.0864 | 0.0297 |
| Illness | Adult | S2 | Mid-point of tuatua | 1 | 0.0665 | 0.0197 |
| Illness | Adult | S3 | Mussel Rocks | 1 | 0.0148 | 0.0036 |
| Illness | Adult | S4 | Western Cockle/Pipi In Harbour | 1 | 0.1687 | 0.0573 |
| Illness | Adult | S5 | Western Swimming & Shellfish In Harbour | 1 | 0.0495 | 0.0139 |
| Illness | Adult | S6 | Western Shellfish In Harbour A | 1 | 0.0231 | 0.0059 |
| Illness | Adult | S7 | Western Shellfish In Harbour B | 1 | 0.0111 | 0.0025 |
| Illness | Adult | S8 | Mid Harbour Shellfish | 1 | 0.0078 | 0.0021 |
| Illness | Adult | S9 | Inner Harbour Shellfish A | 1 | 0.0082 | 0.0018 |
| Illness | Adult | S10 | Inner Harbour Shellfish B | 1 | 0.0097 | 0.0013 |
| Illness | Adult | S11 | Inner Harbour Shellfish C | 1 | 0.0078 | 0.0020 |
| Illness | Adult | S12 | Inner Harbour Shellfish D | 1 | 0.0129 | 0.0024 |
| Illness | Adult | S13 | Domain Recreation/Shellfish | 1 | 0.0090 | 0.0022 |
| Illness | Adult | S14 | Inner Harbour Shellfish | 1 | 0.0134 | 0.0029 |
| Infection | Adult | S1 | Eastern end of tuatua | 1 | 0.1166 | 0.0297 |
| Infection | Adult | S2 | Mid-point of tuatua | 1 | 0.0901 | 0.0197 |
| Infection | Adult | S3 | Mussel Rocks | 1 | 0.0191 | 0.0036 |
| Infection | Adult | S4 | Western Cockle/Pipi In Harbour | 1 | 0.2385 | 0.0573 |
| Infection | Adult | S5 | Western Swimming & Shellfish In Harbour | 1 | 0.0693 | 0.0139 |
| Infection | Adult | S6 | Western Shellfish In Harbour A | 1 | 0.0294 | 0.0059 |
| Infection | Adult | S7 | Western Shellfish In Harbour B | 1 | 0.0145 | 0.0025 |
| Infection | Adult | S8 | Mid Harbour Shellfish | 1 | 0.0107 | 0.0021 |
| Infection | Adult | S9 | Inner Harbour Shellfish A | 1 | 0.0082 | 0.0018 |
| Infection | Adult | S10 | Inner Harbour Shellfish B | 1 | 0.0097 | 0.0013 |
| Infection | Adult | S11 | Inner Harbour Shellfish C | 1 | 0.0078 | 0.0020 |
| Infection | Adult | S12 | Inner Harbour Shellfish D | 1 | 0.0129 | 0.0024 |
| Infection | Adult | S13 | Domain Recreation/Shellfish | 1 | 0.0090 | 0.0022 |
| Infection | Adult | S14 | Inner Harbour Shellfish | 1 | 0.0134 | 0.0029 |


**Figure 4-3: Spatial distribution of adult Norovirus infection risk (bold text) associated with contact recreation assuming one log treatment efficacy.** The site details are listed in Table 3-3 and the infection risk values are summarised in Table 4-1. The bars are scaled relative to the largest risk estimated given one log treatment.

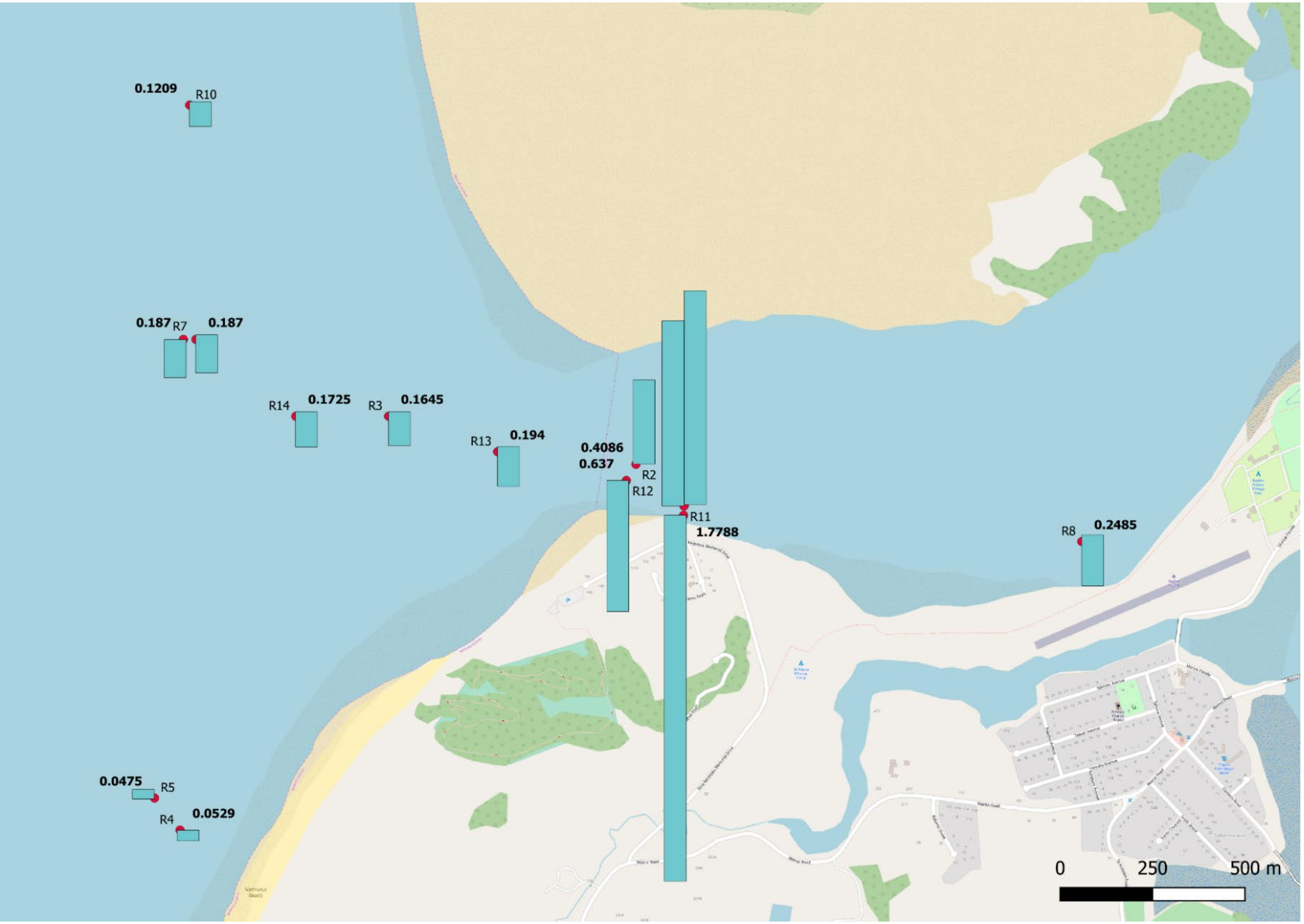


**Figure 4-4: Spatial distribution of adult Adenovirus infection risk (bold text) associated with contact recreation assuming one log treatment efficacy assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.

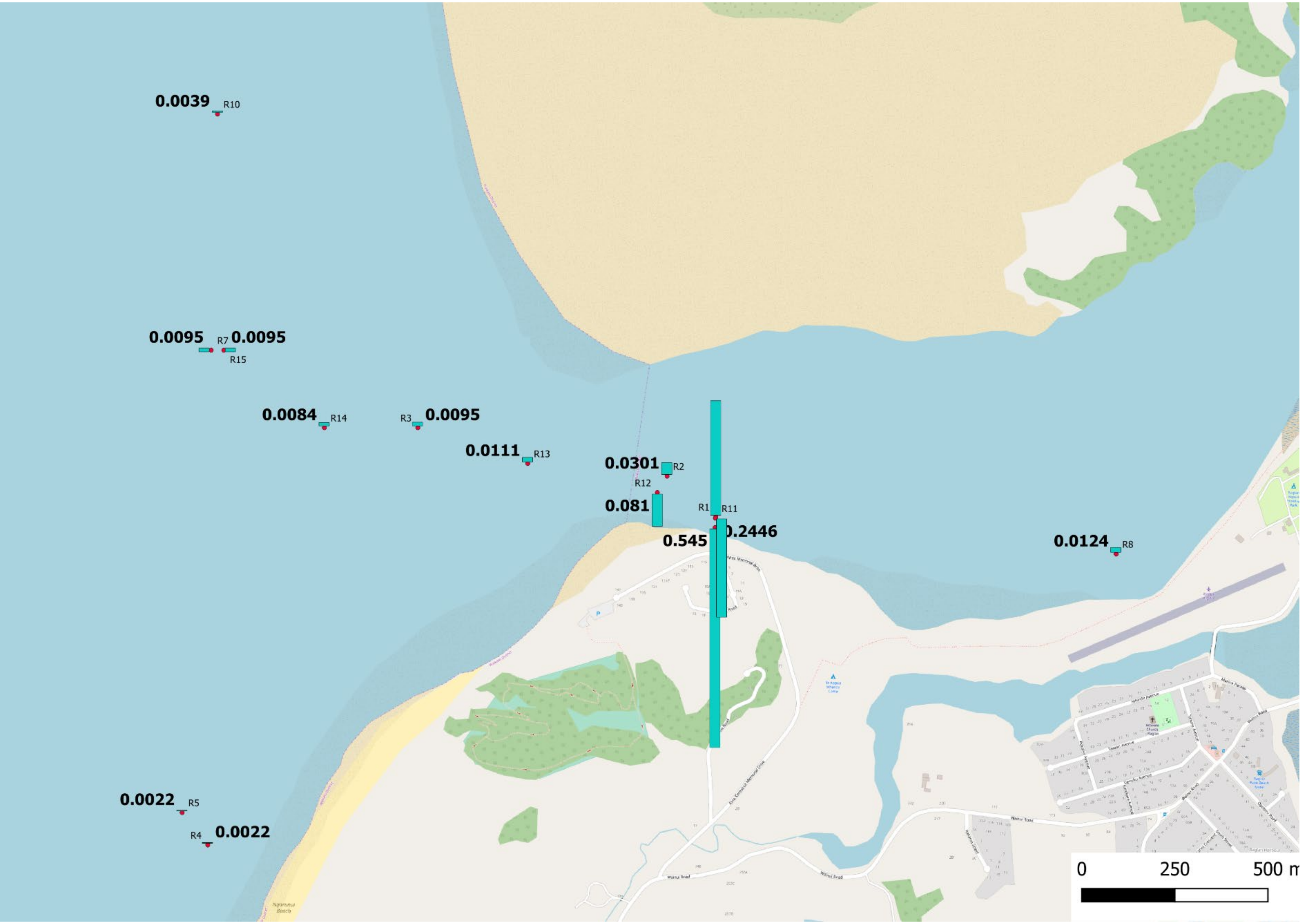


**Figure 4-5: Spatial distribution of child Enterovirus infection risk (bold text) associated with consumption of shellfish assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.

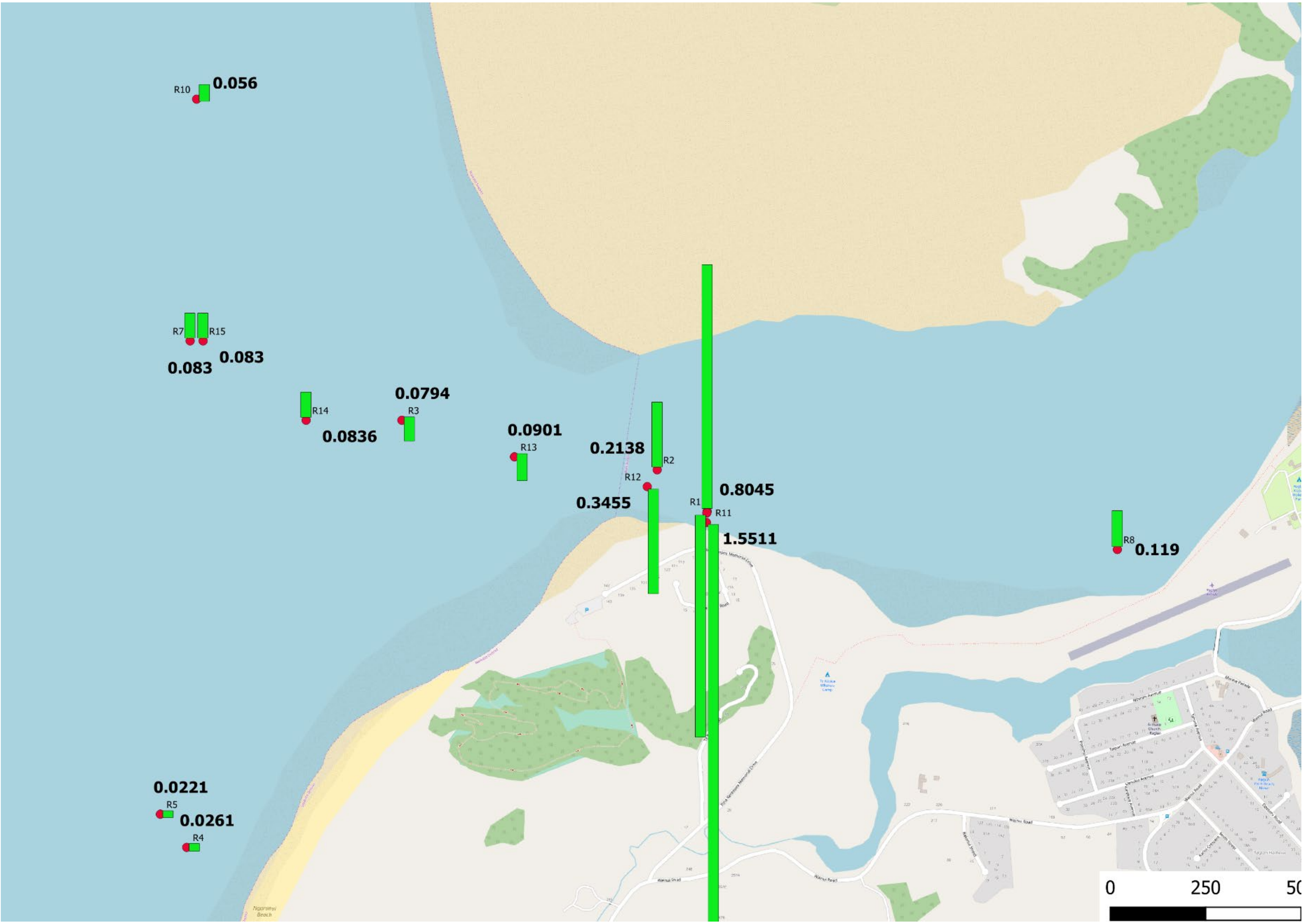


**Figure 4-6: Spatial distribution of child Norovirus infection risk (bold text) associated with contact recreation assuming one log treatment efficacy.** The site details are listed in Table 3-3 and the infection risk values are summarised in Table 4-1. The bars are scaled relative to the largest risk estimated given one log treatment.

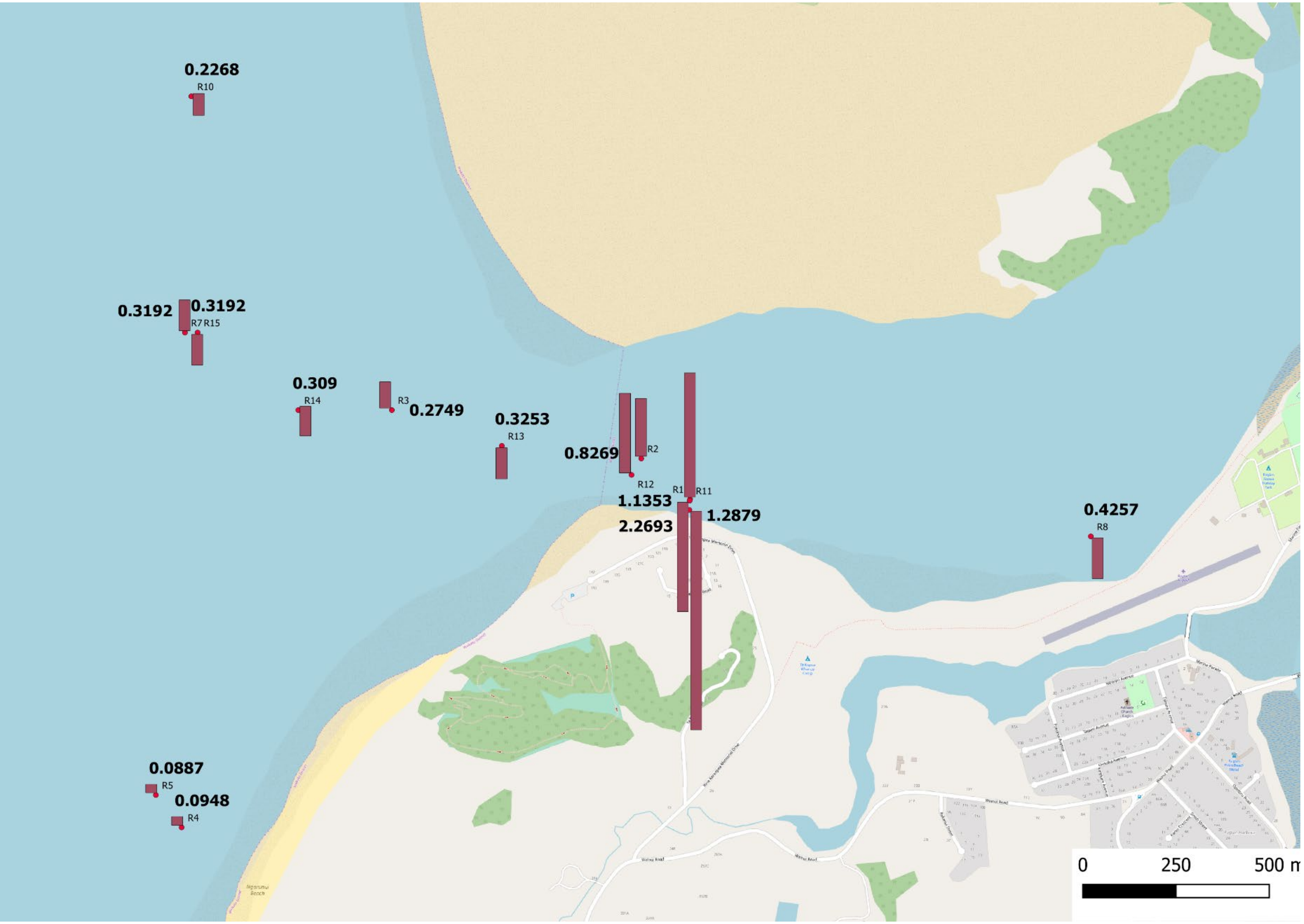


**Figure 4-7: Spatial distribution of child Adenovirus infection risk (bold text) associated with contact recreation assuming one log treatment efficacy assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.

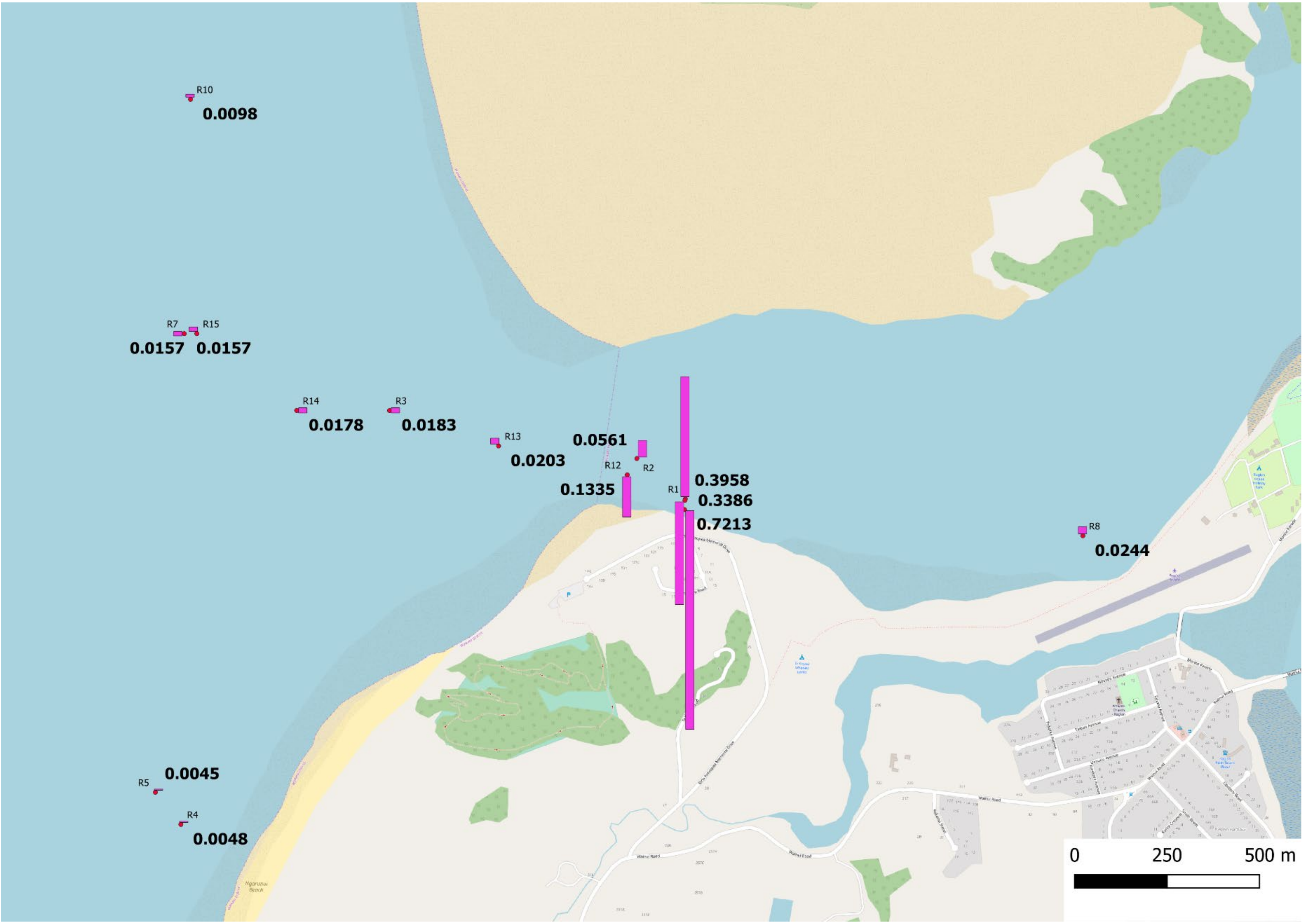


**Figure 4-8: Spatial distribution of Enterovirus child infection risk (bold text) associated with consumption of shellfish assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.


**Figure 4-9: Spatial distribution of Norovirus infection risk (bold text) associated with shellfish consumption assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.


**Figure 4-10: Spatial distribution of Enterovirus infection risk (bold text) associated with shellfish consumption assuming one log treatment efficacy.** The site details are listed in Table 3-4 and the infection risk values are summarised in Table 4-2. The bars are scaled relative to the largest risk estimated given one log treatment.

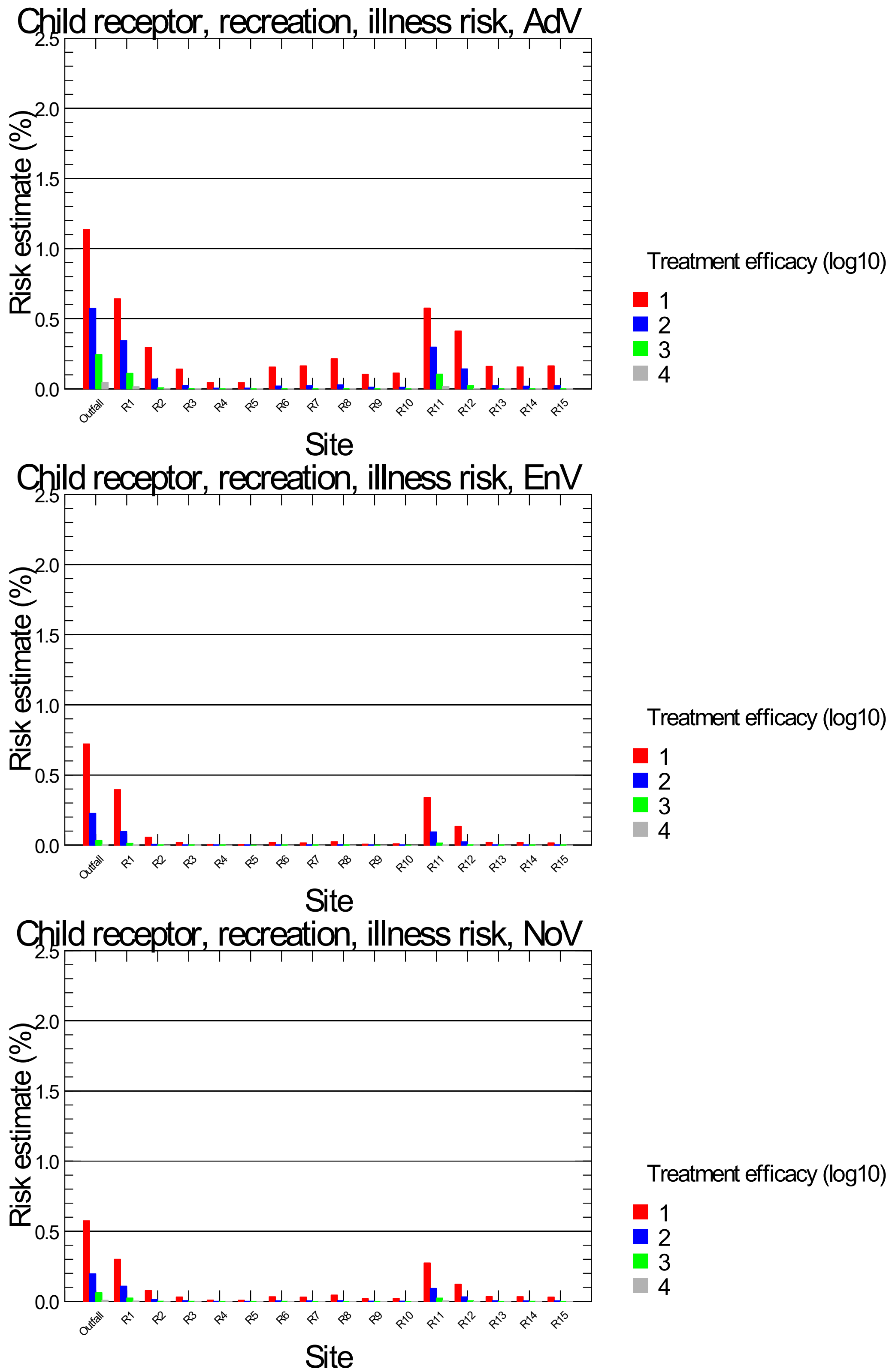


**Figure 4-11: Summary of illness risk estimates for three pathogens for child receptors as a consequence of contact recreation at sites potentially impacted by treated wastewater to four levels of efficacy.** AdV = Adenovirus, EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-2.

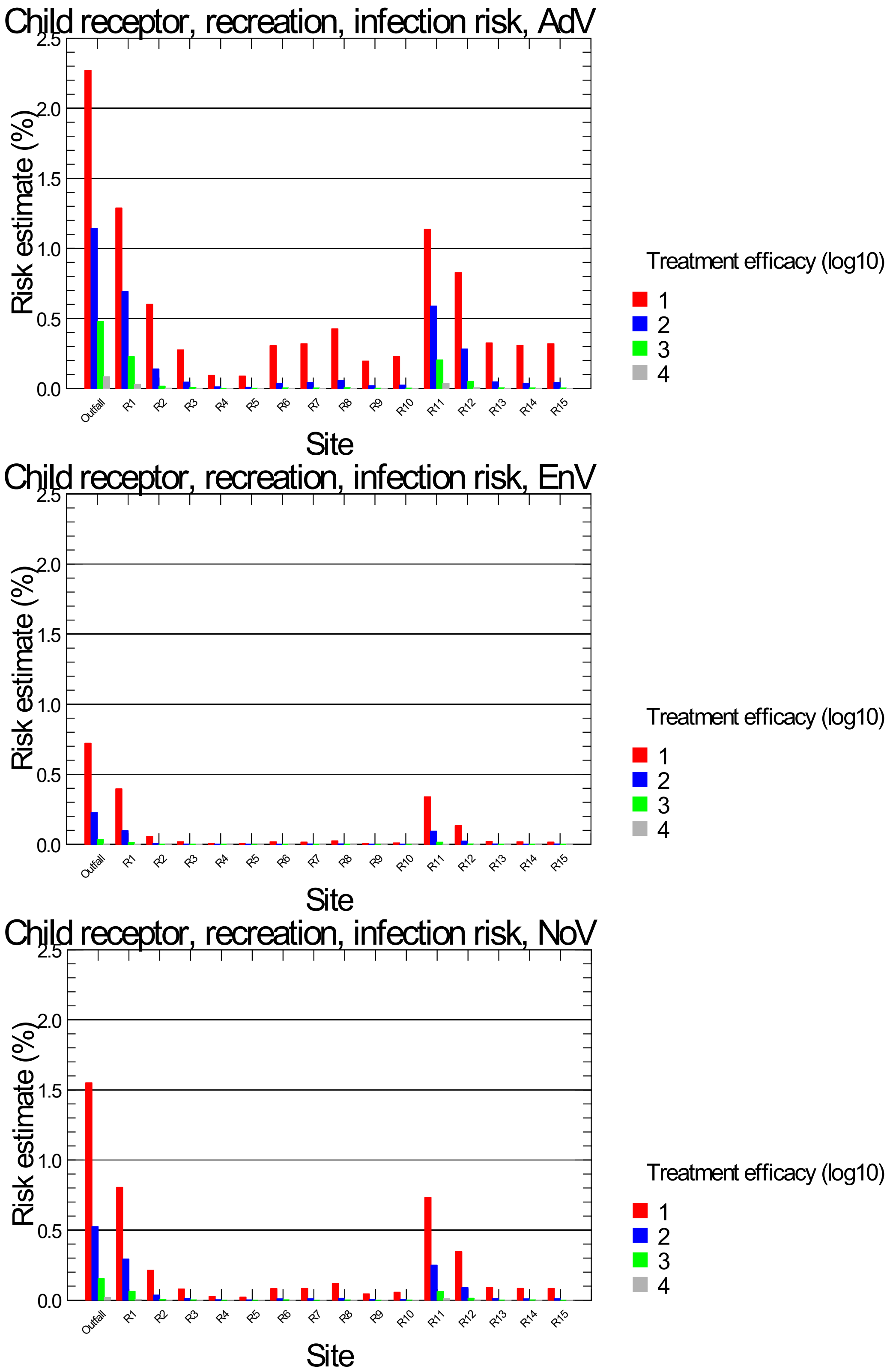


**Figure 4-12: Summary of infection risk estimates for three pathogens for child receptors as a consequence of contact recreation at sites potentially impacted by treated wastewater to four levels of efficacy.** AdV = Adenovirus, EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-2.

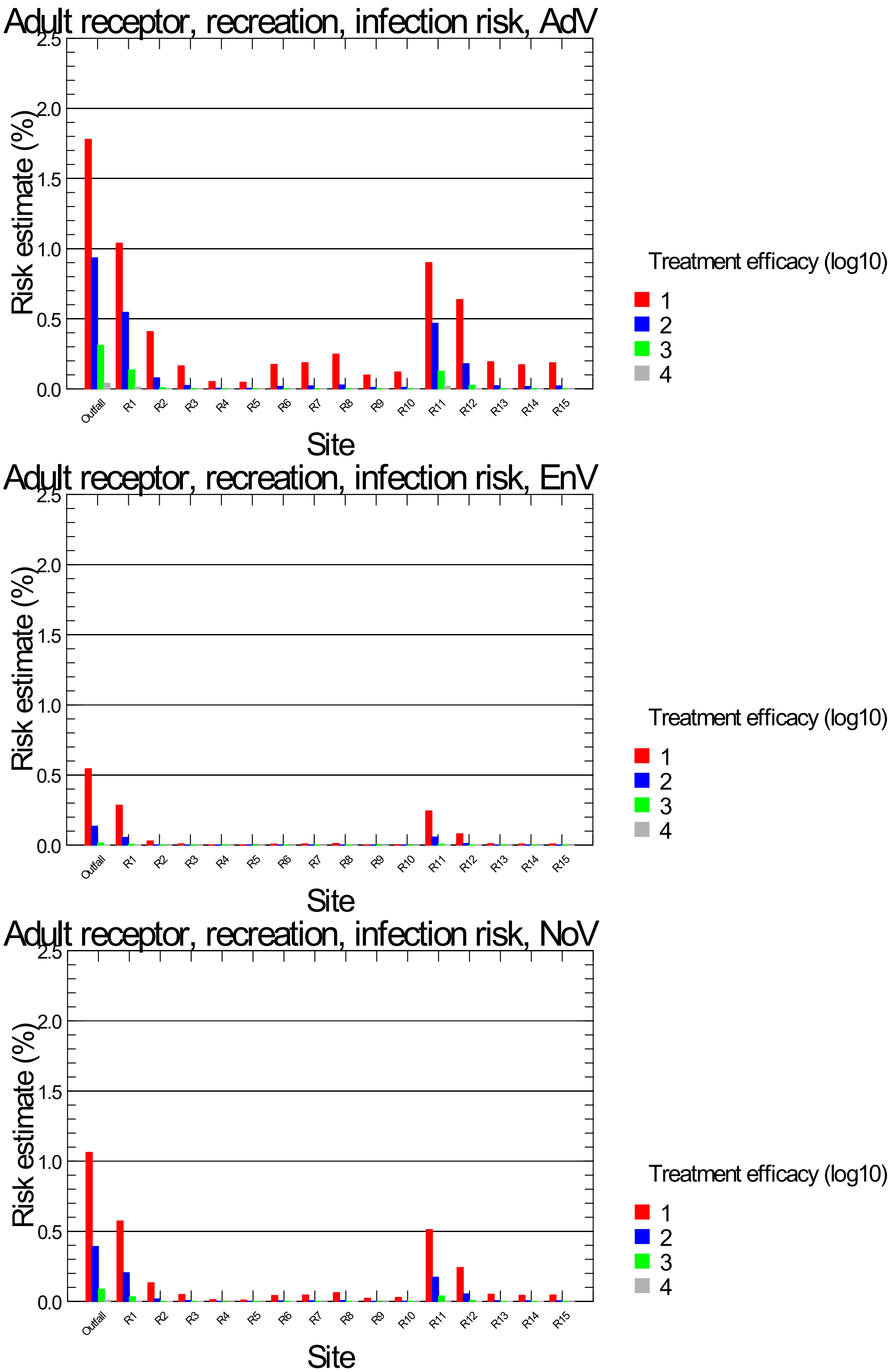


**Figure 4-13: Summary of infection risk estimates for three pathogens for adult receptors as a consequence of contact recreation at sites potentially impacted by treated wastewater to four levels of efficacy.** AdV = Adenovirus, EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-2.

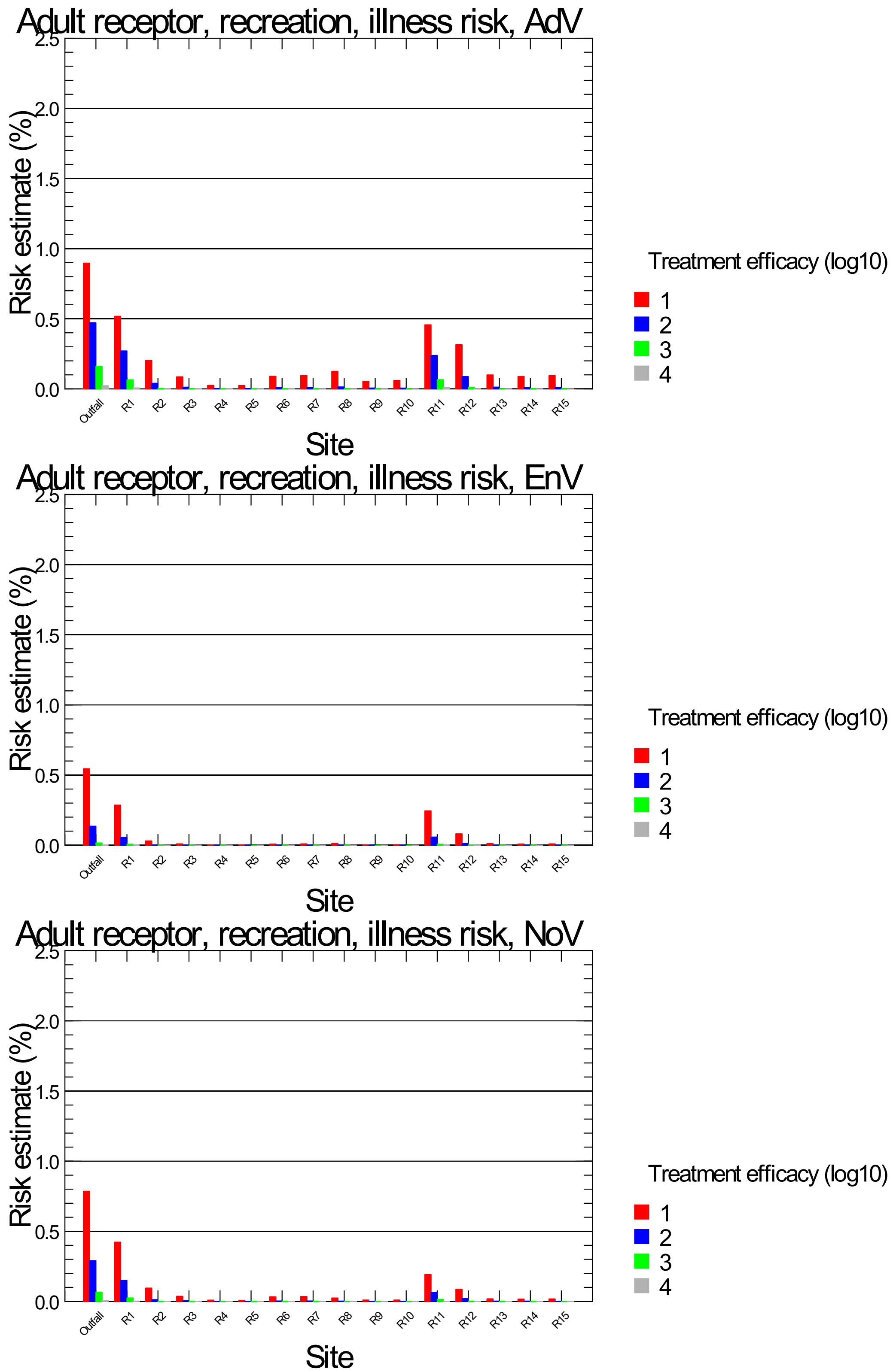


**Figure 4-14: Summary of illness risk estimates for three pathogens for adult receptors as a consequence of contact recreation at sites potentially impacted by treated wastewater treated to four levels of efficacy.** AdV = Adenovirus, EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-2.

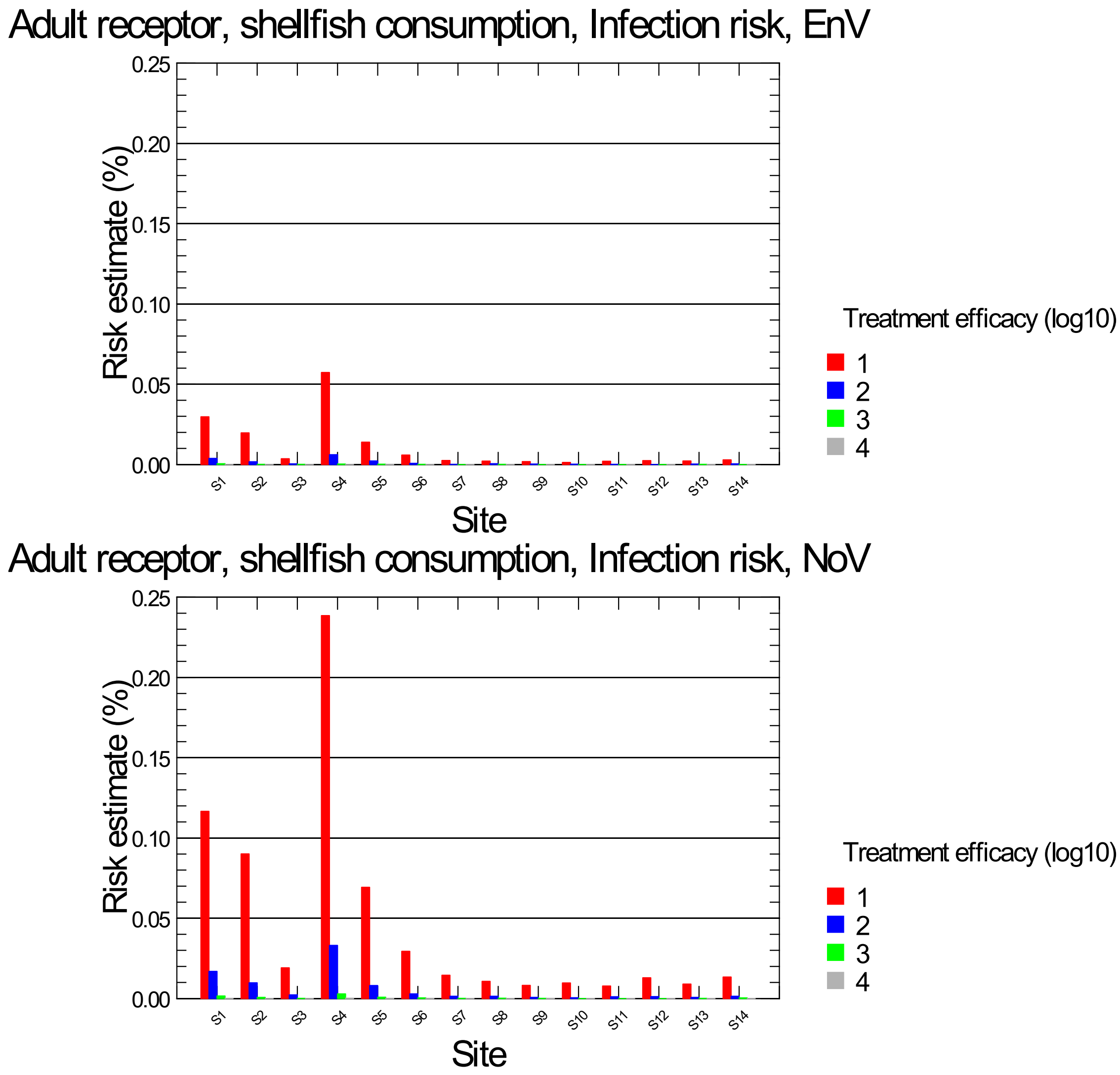


**Figure 4-15: Summary of infection risk estimates for two pathogens as a consequence of consumption of shellfish gathered at sites potentially impacted by treated wastewater to four levels of efficacy.** EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-3.

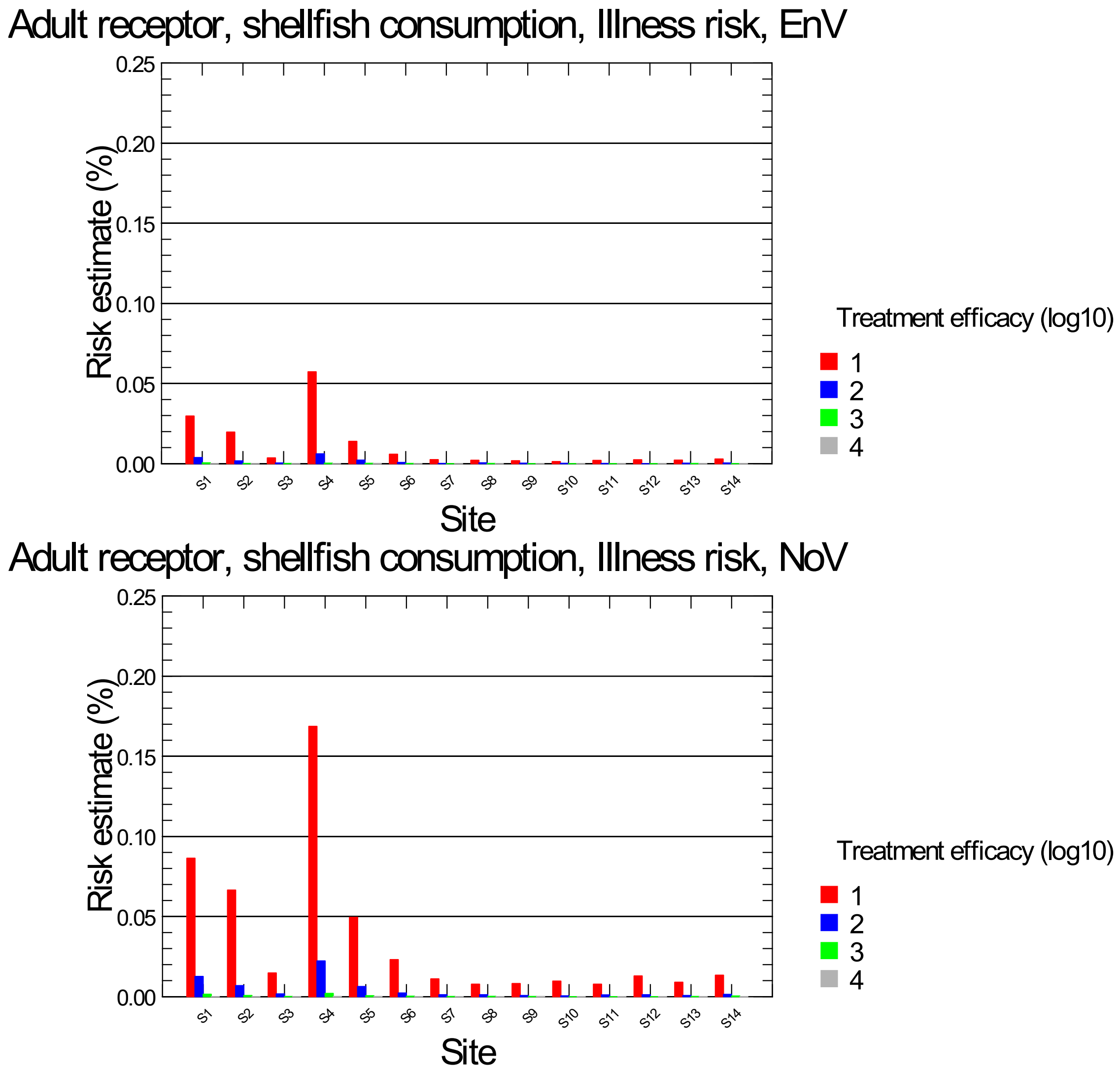


**Figure 4-16: Summary of illness risk estimates for two pathogens as a consequence of consumption of shellfish gathered at sites potentially impacted by treated wastewater to four levels of efficacy.** EnV = Enterovirus, NoV = Norovirus. The location of the sites is indicated in Figure 3-3.

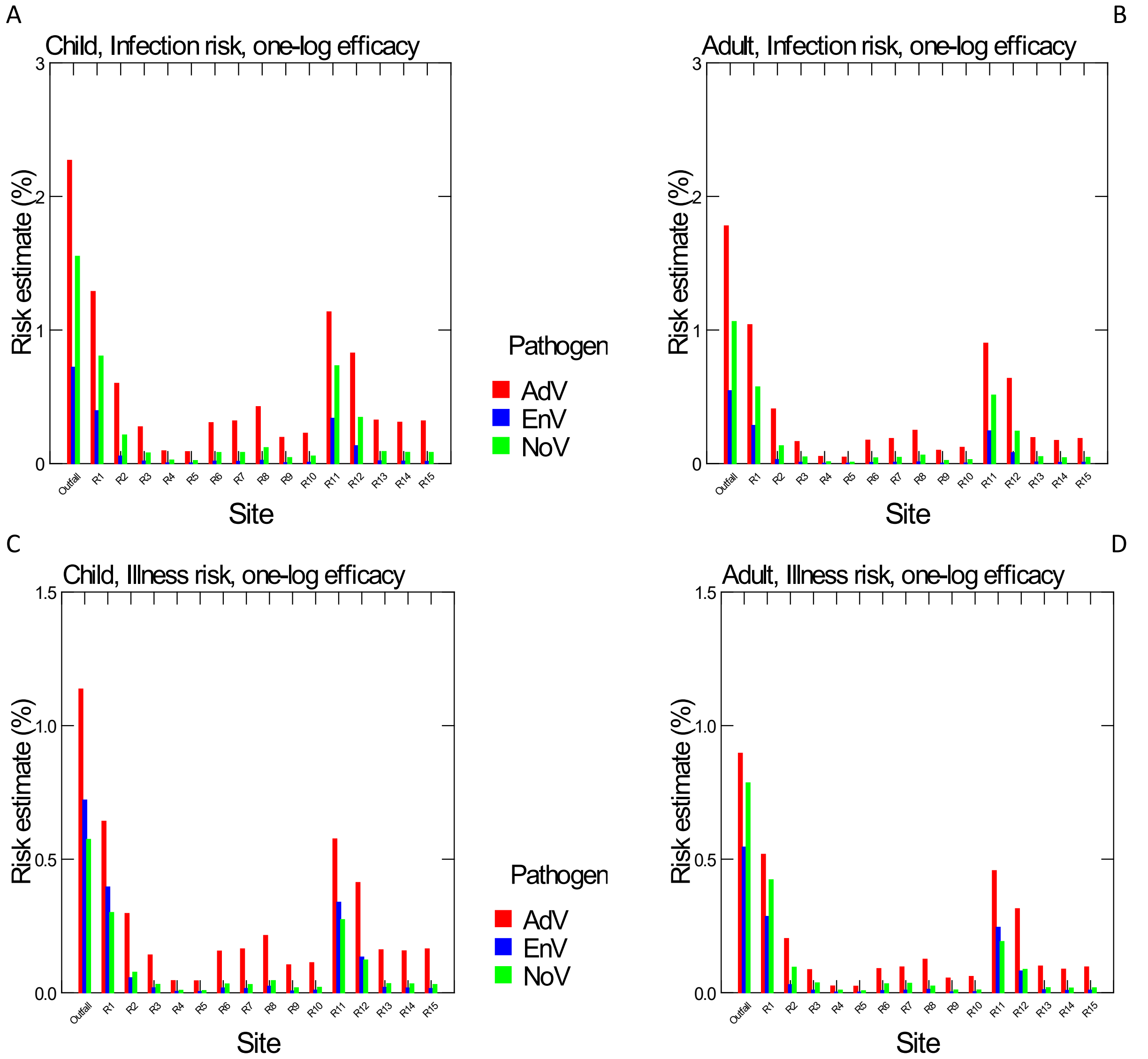


**Figure 4-17: Comparison of illness and infection risk across sites, receptors and pathogen.**

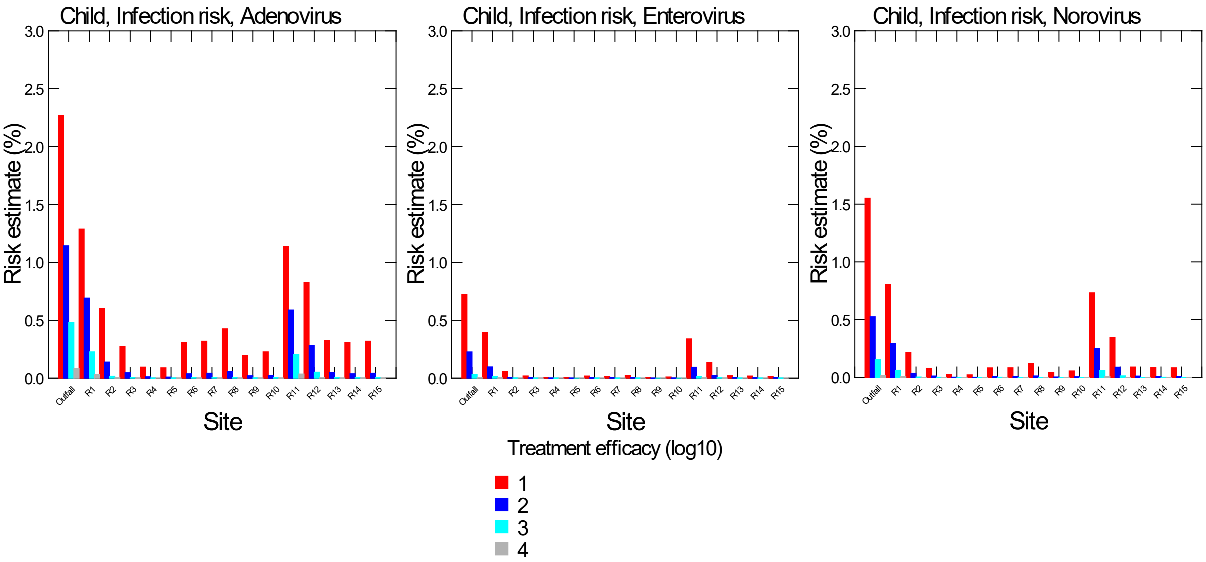


**Figure 4-18: Comparison of infection risk across sites by pathogen and efficacy of wastewater treatment.**

# 5 Discussion

In general, predicted infection and illness risks are generally low. The principal reason is the large dilution achieved at the existing discharge site, coupled with the tidally staged timing of discharge (discharge is timed to occur on ebb tide only). Wastewater is discharged to a relatively narrow harbour mouth, and currents and ebb-tide water velocities are quite large – these characteristics favour high immediate dilution, and dispersion continues as the diluted wastewater is discharged to the Tasman Sea. The west coast of the North Island is also quite dynamic, and the highly diluted wastewater is further dispersed and diluted once it leaves the Harbour mouth. As a consequence, relatively little diluted wastewater is returned to the harbour on the following flood tide. This is evident from the dilutions predicted with all three tracers, but particularly the conservative tracer. Should appreciable mass of wastewater contaminants return to the Harbour, concentrations of a conservative tracer may be expected to increase at times. This is not observed (see Section 4.2).

- For both adults and children and for all pathogens, infection risks are greater than risks of illness – this is because illness requires infection, whereas infection does not always result in illness (Figure 4-11 vs Figure 4-12 for children, and Figure 4-13 vs Figure 4-14 for adults).
- Children have higher risk of infection and illness than their adult counterparts (e.g., Figure 4-12 vs. Figure 4-13).
- For both adults and children, at a given level of treatment efficacy, the risk of infection and illness may be ranked AdV > NoV >EnV (Figure 4-17).
- When risk estimates are viewed spatially, highest illness and infection risks are evident at or near the wastewater outfall. This is consistent with the relatively lower dilution at these sites (Figure 4-3 - Figure 4-10).
- For contact recreation water users:
  - Highest risks for all pathogens and receptors exist at the outfall site, followed by “Kite surf inner” (R1), “Kite surf B” (R11) and “Kite surf C” (R12).
  - The sequence R1, R11, R12 is consistent with the movement of the increasingly diluted wastewater plume as it leaves the harbour.
- For consumers of shellfish:
  - Highest risk exists at site S1, followed by S4 and S5.
  - These sites represent the site nearest to the outfall on the seaward side of the discharge (S1), and the site closest to the outfall on the harbour side of the discharge.
  - Risks are approximately twice as large on the seaward side of the discharge pint as the harbour side of the discharge.
  - Risks are approximately half as low again from site S4 to S5, indicating that the diluted wastewater plume does not intrude substantially into the inner harbour.

- It is likely that the low illness risks at the two sites closest to the outfall arise because these sites are on the shoreline, whereas the diluted wastewater is more or less confined to a narrow plume extending along the well-flushed channel that connects the harbour to the Tasman Sea.

Whether these risk estimates are tolerable or not is a matter that the community must decide, with guidance from health experts. To assist with this evaluation, it is worth considering the health risks associated with the current New Zealand "Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas" (MfE/MoH 2003). The grading system applied to marine waters identify the categories summarised in Table 5-1, which are defined in terms of measured 95$^{th}$ percentile enterococci concentrations observed over a bathing season; the latter have been related to pathogen concentrations through epidemiological studies:

**Table 5-1: Guideline values for microbiological marine water quality.** (MfE/MoH 2003).

| 95$^{th}$ percentile enterococci concentration (N/100 ml) | Basis of derivation | Estimated risk |
|---|---|---|
| ≤40 | Below "no observable adverse effects level" (NOAEL) | <1% gastrointestinal illness<br><0.3% acute febrile respiratory illness |
| 41-200 | Exceeds a "lowest observed adverse effects level" (LOAEL) | 1-5% gastrointestinal illness risk<br>0.3-1.9% acute febrile respiratory illness |
| 201-500 | Substantial elevation in probability of all adverse health outcomes for which dose-response data are available | 5-10% gastrointestinal illness<br>1.9-3.9% acute febrile respiratory illness |
| >500 | Significant risk of high levels of minor illness transmission | >10% gastrointestinal illness<br>>3.9% acute febrile respiratory illness |

In Figure 5-1 we have related the illness risks predicted for the most sensitive receptors (child) and viruses posing greatest illness risk to the Guideline values in Table 5-1 for all sites, at various levels of wastewater treatment. These results indicate that illness risks are generally low ("acceptable" in terms of expectations for recreational water users at one-log treatment level), and decrease substantially as treatment efficacy is increased.

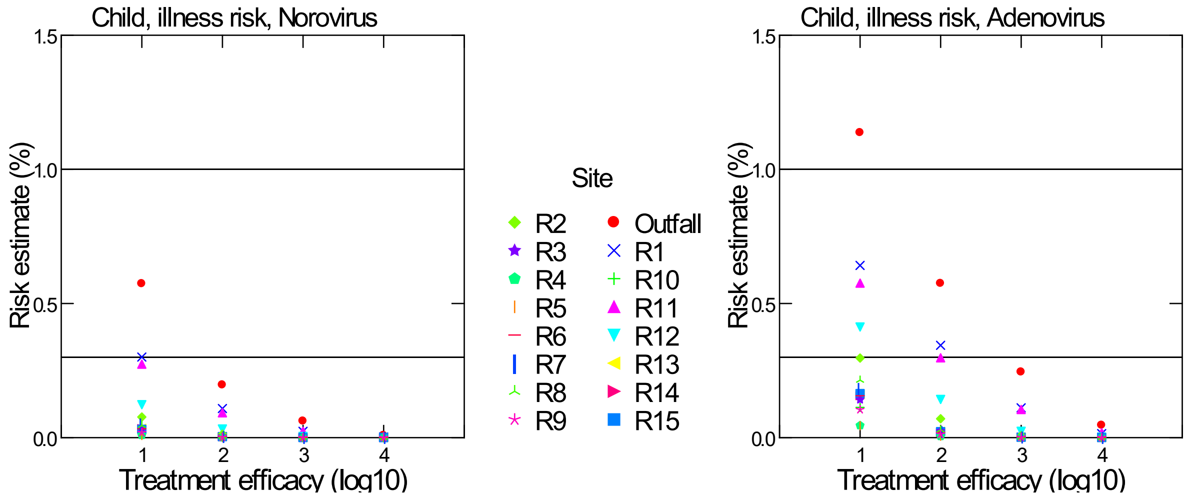


**Figure 5-1: Comparison of estimated illness risk for four levels of treatment efficacy for child receptors and two viruses.** The broken horizontal lines represent the 0.3% AFRI risk and 1% GI risks from Table 5-1.

We are advised that the existing pond treatment system currently provides 0.5-1 log removal, and a membrane treatment system is likely to provide at least 2 log removal for viruses. [15]

[15] Dr John Crawford, BECA, pers. comm.

# 6 Conclusions

Using estimates of dilution derived from a calibrated hydrodynamic model, coupled with Quantitative Microbial Risk Assessment modelling that accounted for a range of WWTP influent virus concentrations, doses of pathogens likely to be ingested by adult and child receptors engaged in contact recreation, we were able to predict infection and illness risks at a range of representative contact recreation and shellfish harvesting sites within Raglan Harbour, and at sites in the Tasman Sea within 3 km of the harbour mouth.

High initial dilution of treated wastewater is achieved, and the concentrations of a conservative tracer are generally low across the model domain. As a consequence, infection and illness risks are generally low for all model pathogens selected.

Illness risks have been related to the illness risks defined in the New Zealand “Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas” (MfE/MoH 2003). At one log order treatment standard, illness risks for the most sensitive receptors (children) at all sites but one (the outfall) are below the 1% Gastrointestinal Illness threshold (the “no observable adverse effects level”). The “no observable adverse effects level” is exceeded for acute febrile respiratory illness at several sites at or nearby the outfall (mainly in the channel discharging to the Tasman Sea, where the diluted wastewater is transported out of the harbour on the ebb tide).

Illness risks associated with consuming uncooked shellfish are greatest at two sites along the south shore nearest the discharge outfall, but are predicted to be well-below 0.3% illness risk.

We also demonstrate the decrease in illness and infection risk for recreational water-users or consumers of shellfish that may be anticipated as the level of wastewater treatment is increased.

# 7 Acknowledgements

I gratefully acknowledge:

- expert assistance with development of the models implemented in Microsoft Excel used to estimate risk by Mr Graham McBride, and
- provision of dilution data derived from hydrodynamic modelling undertaken by Mr John Oldman of DHI in exactly the required format.

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# Appendix A Information to assist with the selection of viruses

**Table A-1: Comparison of the merits and limitations of viruses for which dose-response information is available.**

| Virus | Advantages | Disadvantages |
|---|---|---|
| <u>Gastrointestinal</u> | | |
| **Enterovirus** | Can induce more serious long-term effects compared to other viruses (Haas et al. 1999, DRG 2002, Simpson et al. 2003). Its inclusion is warranted given that it can cause more serious longer-term illnesses.[16] | Restricted to echovirus 12, the only enterovirus for which an infection dose-response relationship is available. Nevertheless, enterovirus by culture captures more than just echovirus, so, for example, would also capture Coxsackie virus.<br>Meaning of "dose" not clear, giving rise to two quite different infection $ID_{50}$ values (54 and 1052).[17] See **Appendix** D. |
| **Norovirus** | Reported to be the most common aetiological agent in receiving waters (e.g., Sinclair et al. 2009). Infection $ID_{50}$ is in the order of 20 virions (among susceptible people), but the dose-response curve rises steeply from the origin, such that ~20% of people may become infected after ingestion of just one virion—see Figure C-1(b), emphasising that a precautionary approach should be taken when modelling this virus. | Efficacy of wastewater treatment in removing infectious noroviruses is difficult to establish.<br>Restricted to Norwalk virus—norovirus genotype I.1. But note that an outbreak study ((Thebault et al. 2013)) identified other genotypes to be, if anything, at least as virulent.<br>In the absence of results to the contrary, and taking an appropriate precautionary approach, noroviruses in treated wastewater are assumed to be not aggregated - were they to be aggregated, health risks would be lessened.<br>May require a conversion from the PCR method used in the clinical trial (Lindesmith et al. 2003; Teunis et al. 2008), as described in (McBride et al. 2013). |
| **Rotavirus** | Particularly affects children. The most infective virus for which published dose-response data is available. Has been used as a "model virus" in earlier QMRAs, for Warkworth (Stott & McBride 2009), Army Bay (Palliser 2011), Snell's Beach (Palliser & Pritchard 2012). | Not as prevalent in treated wastewater as noroviruses.<br>Doses in the one available clinical trial (Ward et al. 1986) were measured in terms of "Focus Forming Units" (FFU), with the lowest "dose" set at 0.009 FFU. So FFU numbers need to be multiplied by an unknown factor to index doses of discrete virions (see the approach taken in a USA-wide study, (McBride et al. 2013). |
| **Hepatitis A** | A serious illness. Dose-response function indicates virulence (infection $ID_{50}$ = <2). | Present in very low numbers in treated wastewater relative to noroviruses. |
| **Coxsackie (an enterovirus)** | May particularly affect children (Suptel 1963). | Studied by Couch et al. (Couch et al. 1965) for coxsackie A21 so restricted to respiratory illness response. Present in low numbers in treated wastewater. Dose-response function indicates moderate virulence (infection $ID_{50}$ = 48). |
| <u>Respiratory</u> | | |
| **Adenovirus** | Found routinely in treated wastewater (DRG 2002, Simpson et al. 2003, (Thompson et al. 2003), Hewitt et al. 2011).Very resistant to disinfection (is double-stranded DNA). A common cause of gastrointestinal illness (especially the 40/41 complex). Can be applied to respiratory infections, and therefore be relevant for surfers and/or water-skiers. | Dose-response only for adenovirus 4, a respiratory aetiological agent. Haas .. (1999) report fitting a single-parameter exponential model to data reported by Couch et al. (1966a) giving rise to an infection $ID_{50}$ less than 2 virions. However, most adenoviruses are not respiratory agents. Applying the adenovirus 4 dose-response model to all adenoviruses for gastrointestinal illness appears to over-estimate the dose-response for that form of illness (we can expect more substantial response of the human body's defences to gastrointestinal infection compared to respiratory infection). Applying the model to only the respiratory portion of total adenoviruses requires assumptions about their proportional presence in treated wastewater (Kundu et al. 2013). The latter authors also considered other studies by (Couch et al. 1966a; Couch et al. 1966b; Couch et al. 1969). |

[16] For example, coxsackievirus type B (an enterovirus) is now recognised as the most common viral aetiological agent associated with heart disease (Haas et al. 1999).
[17] Infection $ID_{50}$ is a quantity derived from clinical trials of pathogen infectivity. It is the pathogen dose that would result in 50% of an exposed population becoming infected.

# Appendix B Adenoviruses

Respiratory viruses, particularly some adenoviruses, may also need to be considered within a QMRA. Respiratory symptoms (via inhalation of contaminated water when water skiing, or inhaling surf-generated aerosols) are sometimes associated with contact with wastewater-impacted coastal waters (WHO 2003). In particular, a New Zealand epidemiological study at seven coastal beaches found a respiratory effect associated with the faecal indicator bacterium enterococci (McBride et al. 1998). Respiratory-associated viruses are probably the commonest causes of acute respiratory infections, for example reportedly causing around 70% of acute sore throats (Mims et al. 2004). They can be particularly resistant to disinfection (Gerba et al. 2003; Thompson et al. 2003). However, while adenoviruses are commonly found in water (Horwitz 2001), including wastewater, many strains give rise to gastrointestinal illness (e.g., the 40/41 strain complex), with a rather smaller proportion associated with respiratory symptoms. So we should note that we have clinical trial information available only for the respiratory-illness-causing adenovirus 4 (Couch et al. 1966a; Couch et al. 1966b; Couch et al. 1969) for which a dose-response model has been developed (Haas et al. 1999). We can expect that people are more vulnerable to respiratory agents than to gastrointestinal agents, because the human body's defences to the latter are more formidable. Fong et al. ((Fong et al. 2010)) found only 3% of wastewater adenoviruses were type 4. So QMRA studies that apply the adenovirus 4 infection dose-response model to all adenoviruses (Gerba et al. 1996; Crabtree et al. 1997) have over-estimated health risk.

Other QMRA studies in New Zealand have predicted illness via ingestion among recreational water users near marine outfalls to be rather higher than illness-via-inhalation (Stott and McBride 2011). A recent study of wet weather bypass flows at Moa Point, Wellington, has included consideration of respiratory effects, using Fong's results (Crawford et al. 2014).

## Enteroviruses

Enterovirus (EV) is a single-stranded member of the picornavirus family, containing over 70 serotypes.[18] It was originally classified into 4 groups, polioviruses, coxsackie A viruses, coxsackie B viruses, and echoviruses but molecular characterisation has led to their reclassification into an enterovirus genus that includes 12 species: enterovirus A-H, J and Rhinovirus A-C. Human species of enterovirus are grouped into the four EV species A-D and the three Rhinovirus groups A-C.

Enteroviruses are often found in respiratory secretions (e.g., saliva, nasal mucus) and stools of infected persons. Poliovirus, coxsackie and echovirus can be spread through faecal-oral route. Infection can result in a wide variety of symptoms ranging from mild respiratory illness (common cold), hand, foot and mouth disease, acute hemorrhagic conjunctivitis, aseptic meningitis, myocarditis, severe neonatal sepsis-like disease, and acute flaccid paralysis. Enteroviruses are distributed worldwide and are influenced by season and climate. Infections can show a seasonal pattern with enterovirus prevalence peaking in summer and early fall in temperate areas, while tropical and semitropical areas showing no discernible seasonal trend.

A comparison with literature data found that E-30 (echovirus 30) was the most prevalent type detected internationally (Janes et al. 2014). Generally, enterovirus B viruses (in particular echoviruses) were the most frequently detected. Age distribution patterns were observed with 30–74% of all isolates detected in young children (< 5 years).

Surveillance and monitoring of enteroviruses has traditionally been based on culturing and serotyping. However, it is likely that concentrations may be under-reported due to differences in cell culture sensitivities (Schiff 1984; Schiff et al. 1984). Current advances in molecular techniques using RT-PCR for detection followed by sequencing of the capsid genes for typing is now the method typically used (Benschop et al. 2010).

[18] http://www.picornastudygroup.com/types/enterovirus/enterovirus.htm

## Noroviruses

Noroviruses are a principal cause of viral gastroenteritis. They are single-stranded RNA viruses that have been classified into 5 genogroups (GI to GV). Strains I, II and IV can infect humans (particularly strain GII, see Matthews et al. 2012), while GIII infects bovine species and GV has recently been identified in mice. The GI viruses are highly infectious for a proportion of the population (Teunis et al. 2008) and spread easily by direct person-to-person or person-surface-person contact. By analogy, the GII genogroup exhibits the same behaviour. They also can be associated with waterborne gastroenteritis (Parshionikar et al. 2003) or shellfish-associated gastroenteritis (Lees et al. 1995; Thebault et al. 2013)[19] and are therefore a hazard to recreational water users (Gray et al. 1997). They have been detected in both raw and treated wastewaters (Nordgren et al. 2009), with strains of GI and GII predominating in human-derived wastewater that are typically very similar to human strains circulating in the population (van den Berg et al. 2005). Therefore, the public may be at appreciable risk whenever there is exposure to human wastes (animal viruses are generally thought to be not infectious to humans, and so other animal pathogens—bacteria and protozoa—come into play). For the purposes of the QMRA, noroviruses therefore represent the primary potential risk of infection from human-derived wastewaters via ingestion for primary contact users, such as swimmers, surfers and bodyboarders.

[19] These authors considered both infection and illness.

# Appendix C Dose-response functions

## For infection

Standard clinical trial procedures involve challenging groups of volunteers with aliquots taken from serially-diluted preparations whose well-mixed concentrations are measured. Doses in individuals' challenges are not measured. Consequently only the average dose given to each member of a group is known. Nevertheless, by making two simple assumptions the mathematical form of the infection dose-response equation can be obtained (Haas et al. 1999, McBride 2005a):

1. The "single-hit" hypothesis: That a single pathogen, surviving the body's barriers (e.g., acidic digestion system) and reaching a potential infection site, is sufficient to cause infection.
2. Poisson distribution of pathogens in the preparation—as is appropriate for a random well-mixed population.

The mathematical result, after averaging across each group's individual Poisson-distributed doses, is the single-parameter "simple exponential" equation

$$\mathrm{Pr}_{\mathrm{inf}}(d) = 1 - \mathrm{e}^{-rd} \qquad (1)$$

where *d* is the average doses given to each group, "e" is the standard exponential number (the base of natural logarithms, e = 2.7183…), and *r* is the probability that a pathogen survives the body's defences and reaches an infection site.

Sometimes host-pathogen interactions are such that a constant value of *r* is implausible (e.g., because of differential immunity, or varying pathogen virulence, as indicated by lack of fit to the single-parameter model). In that case *r* is replaced by a standard two-parameter beta distribution with shape parameter α and location parameter β. The mathematical result is the much-more-difficult-to-evaluate[20] Kummer hypergeometric function (denoted as ${}_1F_1$):

$$\mathrm{Pr}_{\mathrm{inf}}(d) = 1 - {}_1F_1(\alpha, \alpha + \beta, -d) \qquad (2)$$

For obvious reasons this can be called the "beta-Poisson" equation.[21] Fortunately in many cases we find that β >> 1 and α << β, in which case this equation can be well-approximated by the following equation (confusingly, also called "beta-Poisson")

$$\mathrm{Pr}_{\mathrm{inf}} = 1 - \left(1 + \frac{d}{\beta}\right)^{-\alpha} \qquad (3)$$

However this approximation is inadequate for noroviruses because the fitted parameter doublet (α = 0.04 and β = 0.055, Teunis et al. 2008) constitute a serious breach of the approximation-validity criteria (α << β, β>>1). Analysis of clinical trial data for noroviruses therefore calls for specialist software that can evaluate (2), as reported by Teunis et al. 2008, Thebault et al. (2013).

[20] Equation (2) can't be evaluated in Excel.

[21] Because a two-parameter (α and β) beta distribution is used instead of the single parameter *r* and the doses are assumed random, i.e., Poisson-distributed. Strictly, β is not properly a location parameter for equation (2), but it is for its approximation equation (3) (because *d* is simply divided by β in that equation: increasing the value of β shifts the curve to the right).

Good QMRA practice, especially for virulent pathogens, is to "expose" *multiple* people on each exposure occasion.[22] In that case the individual doses are known (i.e., are calculated and assigned to individuals by the model) so that there is no need for Poisson-averaging. This somewhat simplifies the mathematical development of the infection dose-response formulae such that for constant r the simple one-parameter exponential model is replaced by the simple binomial model

$$\mathrm{Pr_{inf}} = 1 - (1 - r)^{i} \tag{4}$$

where *i* is the individual's dose.

Also, the two-parameter beta-Poisson model (the $_1F_1$ functional form) is replaced by the "beta-binomial" model

$$\mathrm{Pr_{inf}} = 1 - \frac{\mathrm{B}(\alpha,\beta + i)}{\mathrm{B}(\alpha,\beta)} \tag{5}$$

where B is the standard beta function (Abramowitz and Stegun 1972) and α and β are as defined previously. This equation can be simply evaluated in Excel.[23]

These two equations have been described by Haas (Haas 2002) as conditional infection dose-response models, the condition being that individual doses are known.

The following figures (Figure C-1a&b) give examples of these functions for adenovirus 4 and for Norwalk virus, for both conditional and unconditional infection dose-response models.

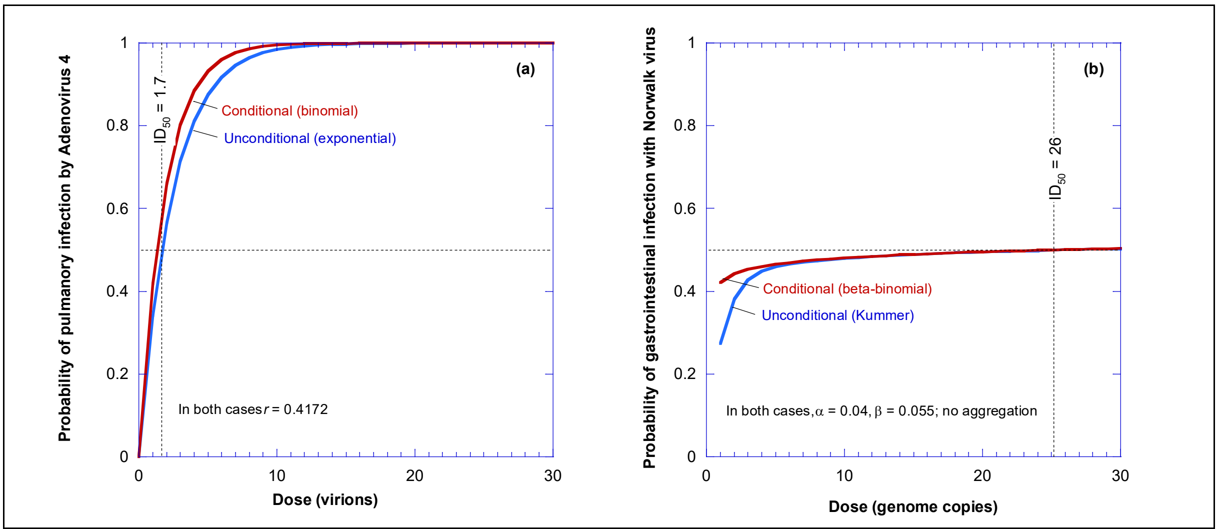


Figure C-1: Conditional and unconditional infection dose-response curves for: (a) single-parameter models for adenovirus 4, and (b) double-parameter models for Norwalk virus (only for susceptible individuals).

These graphs highlight some important features of infection dose-response curves:

- The single-parameter models (e.g., Figure B-1a) rise inexorably to unit probability, precisely because their common parameter (*r*) is constant.

[22] To not do so gives rise to implausible risk profiles. For example if only one individual is exposed per exposure occasion—as a representative of a group visiting a contaminated beach—and if the probability of infection given ingestion of one pathogen is high (say, 20%), then probabilities of infection *between* 0% and 20% are impossible. The resulting risk profile becomes extremely jagged (McBride 2005b). In such cases exposing a group of people per exposure occasion (say, 100), each with different doses (some swim for a few minutes, others for an hour or so), allows many values between 0 and 20% to be calculated.

[23] To do so we note that B(α,β) = Γ(α)Γ(α)/Γ(α+β), where Γ is the standard Gamma function (Abramowitz & Stegun 1972). Standard Excel includes the natural logarithm of the gamma function (as the function 'GAMMALN'), so that we can derive : Pr = 1 – EXP{GAMMALN(β+*i*) + GAMMALN(α+β) – [GAMMALN(α+β+*i*) + GAMMLN(β)]}.

- The double-parameter models (e.g., Figure B-1b) "flatten out" well before reaching unit probability.[24]
- Whilst the relatively high infection $ID_{50}$ for Norwalk virus (26 genome copies among susceptible individuals) occurs on the flattened top of its dose-response curve, infection probabilities are still appreciable at much lower doses.[25]
- The unconditional curves have a jagged profile around the conditional forms, yet deploying the latter in a QMRA gives rise to the same averaged risk.[26]
- Whilst the adenovirus 4 infection dose-response curve is in all respects more severe than that for Norwalk virus, for two reasons that doesn't mean that it is the most severe pathogen:
    i. Adenoviruses that can cause respiratory ailments are a minor part of the total adenovirus population in sewage,[27] with most causing gastro-intestinal illness.
    ii. Exposure to respiratory adenoviruses (via inhalation, e.g., whilst surfing) tends to be lower than ingestion of water whilst swimming.[28]

However, having double-stranded DNA, adenoviruses are more resistant to disinfection processes.

## For illness

Some individuals who become infected (e.g., as measured by serological response, or by evidence of pathogen shedding) may not go on to exhibit symptoms, i.e., they are asymptomatic. In that case, to obtain the unconditional probability of illness (given dose) we first need to calculate the conditional probability of illness given infection for each dose, denoted as $\Pr_{ill|inf}$. The probability of illness is calculated as:

$$\Pr_{ill} = \Pr_{ill|inf} \Pr_{inf} \tag{6}$$

Two common approaches are used for the conditional illness function:

*Hazards model*

Teunis et al. (1999) developed hazard models for the illness given infection, with two forms

*Decreasing hazard*
$$\Pr_{ill|inf}(d) = 1 - \left(1 + \frac{\eta}{d}\right)^{-r} \tag{7}$$

and

*Increasing hazard*
$$\Pr_{ill|inf}(d) = 1 - (1 + \eta d)^{-r} \tag{8}$$

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[24] In fact these models approach unit probability only for enormous doses.

[25] The "flat top" is caused by the variable host-pathogen interactions, including a proportion of exposed population who high (but incomplete) immune. There is also another group who are completely immune.

[26] That's because applying the unconditional form to a single individual representing a group of people, as is common practice, doesn't capture the fact that, by good luck, some people at a beach will avoid exposure whilst the averaged dose is above zero (McBride 2005b).

[27] Typically respiratory serotypes are detected less frequently than adenovirus F serotypes and so the gastro-intestinal (GI) disease-causing seroptyes tend to predominate in sewage studies Osuolale, O., Okoh, A. (2015) Incidence of human adenoviruses and hepatitis A virus in the final effluent of selected wastewater treatment plants in Eastern Cape Province, South Africa. *Virology Journal*, 12: 98. . However, a proportion of respiratory versus GI serotypes detected will depend on the cell line used for culture assays and the target primers for molecular methods. For example, Hewitt et al. (2011) used cell line 594 and reported that culturable adenoviruses were mainly A-E types (which are respiratory and conjunctivitis serotypes) and there was still around 3 log presence in effluents.

[28] Water-contact-related respiratory illness is an area worthy of further research, particularly in the light of the respiratory illness rates reported in the one New Zealand epidemiological study on this matter—McBride et al. (1998). In that study (at seven New Zealand beaches) those rates were generally more prominent than gastrointestinal rates, a phenomenon that is not fully understood.

where η is a location parameter, and *r* is a shape parameter.[29]

*Dose independence*

Existing models of the conditional probabilities of illness (the condition being that infection has already occurred) are held in some doubt internationally. For example, the norovirus model (Teunis et al. 2008) predicts substantial infection probabilities at very low doses, but predicts substantial illness probabilities (among the infected) only at very high doses. A large body of work has taken the view that the conditional probability of illness-given-infection should be independent of dose—(Schoen and Ashbolt 2010), (Soller et al. 2010; Soller et al. 2015), (Viau et al. 2011) and (Boehm et al. 2015). Indeed, that approach is endorsed by WHO (WHO 2011), with the result that for the pathogens considered here the conditional illness probabilities are on the order of ½.

[29] The decreasing hazards model has only been reported for a clinical trial on adults exposed to *Campylobacter* (Teunis et al. 1999): All other conditional illness models that I am aware of infer an increasing hazards model, including a *Campylobacter* outbreak study for children (Teunis et al. 2005).

# Appendix D Echovirus 12 clinical trial data analysis

Echovirus is a member of the enterovirus family. Haas et al. (1999) reported fitting a one-parameter simple exponential model to clinical trial data for an echovirus 12 study (Akin 1981),[30] with an estimated infection $ID_{50}$ = 54 virions, corresponding to their calibrated *r* value of 0.0128.[31] Haas (1983) had earlier fitted a slightly different value to the Akin data, with $r = 0.012$ (giving infection $ID_{50}$ = 58) and also a two-parameter beta-Poisson curve (with $\alpha = 1.3$ and $\beta = 75$), so that the infection $ID_{50}$ [= $\beta(2^{1/\alpha} - 1)$] = 53. Clearly, these approaches give consistent results with an infection $ID_{50}$ about 50.

The beta-Poisson result was used in the QMRA performed for the Mangere wastewater treatment upgrade (DRG 2002, Simpson et al. 2003), this choice being particularly influenced by the observation that enterovirus illness can give rise to more serious consequences (i.e., sequelae) relative to other virus groups.

Akin's data were in fact preliminary results from an ongoing clinical trial, full results of which were reported three years later in Schiff et al*.* (1984a&b). Their 1984a paper is the proceedings of a conference held two years earlier in Herzliya Israel. It contains the Akin data. But the 1984b document (a peer-reviewed journal paper) multiplied all the doses, including those reported by Akin, by a factor of 33, to account for the re-analysis of the stock dose suspension using a more sensitive cell line[32]. These published data were analysed by Teunis et al*.* (1996) giving rise to a two-parameter "beta-Poisson" model ($\alpha = 0.401$, $\beta = 227.2$, as reported by Teunis et al. 1996) and a higher infection $ID_{50}$ = 1052 virions.[33]

We propose to use the beta-Poisson model ($\alpha = 1.3$ and $\beta = 75$, with infection $ID_{50}$ = 53 virions). Note that this conflicts with the approach taken in the increasingly-influential CAMRA website[34] ($\alpha = 1.06$ and $\beta = 171.3$), giving rise to an infection $ID_{50}$ = 922. This has implications for the enterovirus concentrations to be presented to this dose-response function in the QMRA calculations.[35]

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[30] This widely-quoted paper (Akin 1981) seems to have been read by only a few, given its appearance only in the "grey literature", decades past. The author of this report has a copy, courtesy of Professor Haas (Drexel University), which is available on request.

[31] For the simple exponential model, algebraic manipulation shows that $ID_{50} = -\ell n(½)/r \approx 0.693/r$.

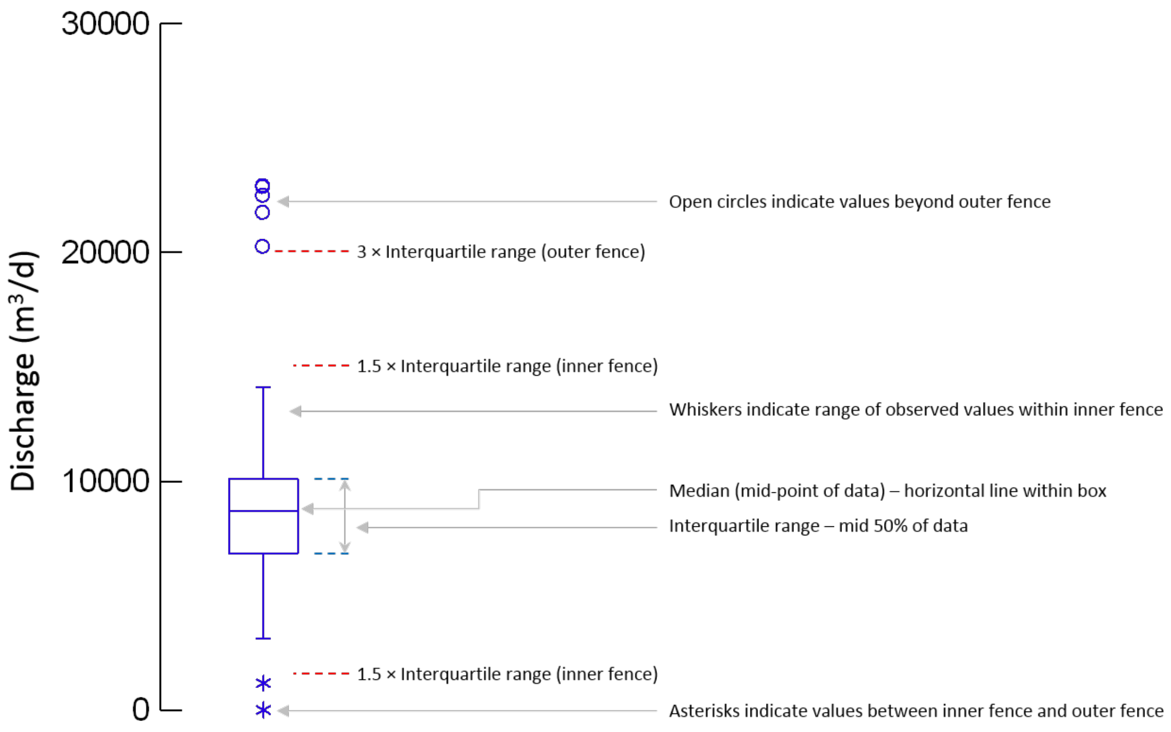
[32] At page 864 of Schiff et al. (1984b): "The original plaque assay used for determination of the titer of the echovirus-12 pool and of the various challenge doses administered to volunteers was based on the use of LLC-$MK_2$ cells and an agar overlay procedure; in the present study this assay was shown to be significantly less sensitive than the plaque neutralization assay involving RD cells and a soft agar overlay procedure. The latter system increased the plaquing efficiency of the challenge virus by 33-fold."

[33] For the approximate beta-Poisson model, algebraic manipulation shows that $ID_{50} = \beta(2^{1/\alpha} - 1)$.

[34] Center for Advancing Microbial Risk Assessment http://qmrawiki.canr.msu.edu/index.php?title=Table_of_Recommended_Best-Fit_Parameters#tab=Viruses

[35] The adopted dose-response function refers to echovirus 12 data gathered using the "LLC-$MK_2$" cell line (Schiff et al. 1984a). The CAMRA dose-response function refers to data re-analysed using "RD" cell line. Comparison of dose-response functions for other members of the enterovirus group (e.g., polio virus, hepatitis A, coxsackie) indicates that $ID_{50}$ of the order of 50 is more tenable than of the order of 1000.

# Appendix E Explanation of a box and whisker plot



# Appendix F Infection and Illness risks – Recreational exposure

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Infection | Child | R1 | Kite Surf Inner | 1 | 0.8045 | 1.2879 | 0.3958 |
| Infection | Child | R1 | Kite Surf Inner | 2 | 0.2931 | 0.6912 | 0.0966 |
| Infection | Child | R1 | Kite Surf Inner | 3 | 0.0623 | 0.2267 | 0.0127 |
| Infection | Child | R1 | Kite Surf Inner | 4 | 0.0065 | 0.0301 | 0.0015 |
| Infection | Child | R2 | Inshore Kite surf | 1 | 0.2138 | 0.6003 | 0.0561 |
| Infection | Child | R2 | Inshore Kite surf | 2 | 0.0362 | 0.1390 | 0.0056 |
| Infection | Child | R2 | Inshore Kite surf | 3 | 0.0035 | 0.0162 | 0.0007 |
| Infection | Child | R2 | Inshore Kite surf | 4 | 0.0004 | 0.0016 | 0.0000 |
| Infection | Child | R3 | Entrance kite surf | 1 | 0.0794 | 0.2749 | 0.0183 |
| Infection | Child | R3 | Entrance kite surf | 2 | 0.0125 | 0.0460 | 0.0020 |
| Infection | Child | R3 | Entrance kite surf | 3 | 0.0016 | 0.0058 | 0.0004 |
| Infection | Child | R3 | Entrance kite surf | 4 | 0.0001 | 0.0006 | 0.0002 |
| Infection | Child | R4 | Northern swimming | 1 | 0.0261 | 0.0948 | 0.0048 |
| Infection | Child | R4 | Northern swimming | 2 | 0.0023 | 0.0113 | 0.0010 |
| Infection | Child | R4 | Northern swimming | 3 | 0.0002 | 0.0008 | 0.0001 |
| Infection | Child | R4 | Northern swimming | 4 | 0.0000 | 0.0000 | 0.0000 |
| Infection | Child | R5 | Northern surfing | 1 | 0.0221 | 0.0887 | 0.0045 |
| Infection | Child | R5 | Northern surfing | 2 | 0.0016 | 0.0092 | 0.0005 |
| Infection | Child | R5 | Northern surfing | 3 | 0.0002 | 0.0010 | 0.0000 |
| Infection | Child | R5 | Northern surfing | 4 | 0.0000 | 0.0001 | 0.0000 |
| Infection | Child | R6 | Bar surf | 1 | 0.0824 | 0.3058 | 0.0178 |
| Infection | Child | R6 | Bar surf | 2 | 0.0088 | 0.0373 | 0.0016 |
| Infection | Child | R6 | Bar surf | 3 | 0.0018 | 0.0042 | 0.0003 |
| Infection | Child | R6 | Bar surf | 4 | 0.0004 | 0.0007 | 0.0000 |
| Infection | Child | R7 | Offshore kite surf/Maui | 1 | 0.0830 | 0.3192 | 0.0157 |
| Infection | Child | R7 | Offshore kite surf/Maui | 2 | 0.0098 | 0.0426 | 0.0026 |
| Infection | Child | R7 | Offshore kite surf/Maui | 3 | 0.0004 | 0.0037 | 0.0003 |
| Infection | Child | R7 | Offshore kite surf/Maui | 4 | 0.0000 | 0.0004 | 0.0000 |
| Infection | Child | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.1190 | 0.4257 | 0.0244 |
| Infection | Child | R8 | Western Swimming & Shellfish In Harbour | 2 | 0.0129 | 0.0563 | 0.0027 |
| Infection | Child | R8 | Western Swimming & Shellfish In Harbour | 3 | 0.0013 | 0.0055 | 0.0003 |
| Infection | Child | R8 | Western Swimming & Shellfish In Harbour | 4 | 0.0001 | 0.0003 | 0.0000 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Infection | Child | R9 | Domain Recreation/Shellfish | 1 | 0.0445 | 0.1956 | 0.0067 |
| Infection | Child | R9 | Domain Recreation/Shellfish | 2 | 0.0041 | 0.0199 | 0.0001 |
| Infection | Child | R9 | Domain Recreation/Shellfish | 3 | 0.0001 | 0.0015 | 0.0000 |
| Infection | Child | R9 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0002 | 0.0000 |
| Infection | Child | R10 | Maui North | 1 | 0.0560 | 0.2268 | 0.0098 |
| Infection | Child | R10 | Maui North | 2 | 0.0053 | 0.0242 | 0.0010 |
| Infection | Child | R10 | Maui North | 3 | 0.0005 | 0.0026 | 0.0000 |
| Infection | Child | R10 | Maui North | 4 | 0.0000 | 0.0003 | 0.0000 |
| Infection | Child | R11 | Kite Surf B | 1 | 0.7322 | 1.1353 | 0.3386 |
| Infection | Child | R11 | Kite Surf B | 2 | 0.2494 | 0.5881 | 0.0938 |
| Infection | Child | R11 | Kite Surf B | 3 | 0.0612 | 0.2035 | 0.0148 |
| Infection | Child | R11 | Kite Surf B | 4 | 0.0097 | 0.0352 | 0.0014 |
| Infection | Child | R12 | Kite Surf C | 1 | 0.3455 | 0.8269 | 0.1335 |
| Infection | Child | R12 | Kite Surf C | 2 | 0.0887 | 0.2819 | 0.0233 |
| Infection | Child | R12 | Kite Surf C | 3 | 0.0132 | 0.0511 | 0.0023 |
| Infection | Child | R12 | Kite Surf C | 4 | 0.0009 | 0.0050 | 0.0004 |
| Infection | Child | R13 | Kite Surf D | 1 | 0.0901 | 0.3253 | 0.0203 |
| Infection | Child | R13 | Kite Surf D | 2 | 0.0116 | 0.0473 | 0.0023 |
| Infection | Child | R13 | Kite Surf D | 3 | 0.0005 | 0.0044 | 0.0003 |
| Infection | Child | R13 | Kite Surf D | 4 | 0.0001 | 0.0005 | 0.0000 |
| Infection | Child | R14 | Kite Surf E | 1 | 0.0836 | 0.3090 | 0.0178 |
| Infection | Child | R14 | Kite Surf E | 2 | 0.0088 | 0.0370 | 0.0016 |
| Infection | Child | R14 | Kite Surf E | 3 | 0.0017 | 0.0040 | 0.0003 |
| Infection | Child | R14 | Kite Surf E | 4 | 0.0003 | 0.0007 | 0.0000 |
| Infection | Child | R15 | Kite Surf F | 1 | 0.0830 | 0.3192 | 0.0157 |
| Infection | Child | R15 | Kite Surf F | 2 | 0.0098 | 0.0426 | 0.0026 |
| Infection | Child | R15 | Kite Surf F | 3 | 0.0004 | 0.0037 | 0.0003 |
| Infection | Child | R15 | Kite Surf F | 4 | 0.0000 | 0.0004 | 0.0000 |
| Infection | Child | Out | Outfall | 1 | 1.5511 | 2.2693 | 0.7213 |
| Infection | Child | Out | Outfall | 2 | 0.5246 | 1.1431 | 0.2263 |
| Infection | Child | Out | Outfall | 3 | 0.1532 | 0.4788 | 0.0327 |
| Infection | Child | Out | Outfall | 4 | 0.0186 | 0.0832 | 0.0025 |
| Infection | Adult | R1 | Kite Surf Inner | 1 | 0.5735 | 1.0393 | 0.2854 |
| Infection | Adult | R1 | Kite Surf Inner | 2 | 0.2045 | 0.5456 | 0.0553 |
| Infection | Adult | R1 | Kite Surf Inner | 3 | 0.0331 | 0.1353 | 0.0073 |
| Infection | Adult | R1 | Kite Surf Inner | 4 | 0.0031 | 0.0138 | 0.0007 |
| Infection | Adult | R2 | Inshore Kite surf | 1 | 0.1330 | 0.4086 | 0.0301 |
| Infection | Adult | R2 | Inshore Kite surf | 2 | 0.0181 | 0.0787 | 0.0019 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Infection | Adult | R2 | Inshore Kite surf | 3 | 0.0017 | 0.0083 | 0.0001 |
| Infection | Adult | R2 | Inshore Kite surf | 4 | 0.0003 | 0.0008 | 0.0000 |
| Infection | Adult | R3 | Entrance kite surf | 1 | 0.0492 | 0.1645 | 0.0095 |
| Infection | Adult | R3 | Entrance kite surf | 2 | 0.0066 | 0.0244 | 0.0008 |
| Infection | Adult | R3 | Entrance kite surf | 3 | 0.0009 | 0.0028 | 0.0003 |
| Infection | Adult | R3 | Entrance kite surf | 4 | 0.0001 | 0.0002 | 0.0000 |
| Infection | Adult | R4 | Northern swimming | 1 | 0.0134 | 0.0529 | 0.0022 |
| Infection | Adult | R4 | Northern swimming | 2 | 0.0010 | 0.0050 | 0.0004 |
| Infection | Adult | R4 | Northern swimming | 3 | 0.0000 | 0.0004 | 0.0001 |
| Infection | Adult | R4 | Northern swimming | 4 | 0.0000 | 0.0000 | 0.0000 |
| Infection | Adult | R5 | Northern surfing | 1 | 0.0097 | 0.0475 | 0.0022 |
| Infection | Adult | R5 | Northern surfing | 2 | 0.0005 | 0.0046 | 0.0003 |
| Infection | Adult | R5 | Northern surfing | 3 | 0.0001 | 0.0007 | 0.0000 |
| Infection | Adult | R5 | Northern surfing | 4 | 0.0000 | 0.0000 | 0.0000 |
| Infection | Adult | R6 | Bar surf | 1 | 0.0423 | 0.1747 | 0.0079 |
| Infection | Adult | R6 | Bar surf | 2 | 0.0042 | 0.0182 | 0.0008 |
| Infection | Adult | R6 | Bar surf | 3 | 0.0013 | 0.0024 | 0.0002 |
| Infection | Adult | R6 | Bar surf | 4 | 0.0002 | 0.0004 | 0.0000 |
| Infection | Adult | R7 | Offshore kite surf/Maui | 1 | 0.0460 | 0.1870 | 0.0095 |
| Infection | Adult | R7 | Offshore kite surf/Maui | 2 | 0.0048 | 0.0214 | 0.0016 |
| Infection | Adult | R7 | Offshore kite surf/Maui | 3 | 0.0001 | 0.0015 | 0.0001 |
| Infection | Adult | R7 | Offshore kite surf/Maui | 4 | 0.0000 | 0.0002 | 0.0000 |
| Infection | Adult | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.0627 | 0.2485 | 0.0124 |
| Infection | Adult | R8 | Western Swimming & Shellfish In Harbour | 2 | 0.0069 | 0.0281 | 0.0015 |
| Infection | Adult | R8 | Western Swimming & Shellfish In Harbour | 3 | 0.0004 | 0.0028 | 0.0002 |
| Infection | Adult | R8 | Western Swimming & Shellfish In Harbour | 4 | 0.0001 | 0.0002 | 0.0000 |
| Infection | Adult | R9 | Domain Recreation/Shellfish | 1 | 0.0227 | 0.0991 | 0.0030 |
| Infection | Adult | R9 | Domain Recreation/Shellfish | 2 | 0.0015 | 0.0106 | 0.0000 |
| Infection | Adult | R9 | Domain Recreation/Shellfish | 3 | 0.0000 | 0.0004 | 0.0000 |
| Infection | Adult | R9 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0001 | 0.0000 |
| Infection | Adult | R10 | Maui North | 1 | 0.0288 | 0.1209 | 0.0039 |
| Infection | Adult | R10 | Maui North | 2 | 0.0024 | 0.0116 | 0.0003 |
| Infection | Adult | R10 | Maui North | 3 | 0.0004 | 0.0012 | 0.0000 |
| Infection | Adult | R10 | Maui North | 4 | 0.0000 | 0.0001 | 0.0000 |
| Infection | Adult | R11 | Kite Surf B | 1 | 0.5123 | 0.9009 | 0.2446 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Infection | Adult | R11 | Kite Surf B | 2 | 0.1731 | 0.4679 | 0.0579 |
| Infection | Adult | R11 | Kite Surf B | 3 | 0.0393 | 0.1264 | 0.0071 |
| Infection | Adult | R11 | Kite Surf B | 4 | 0.0049 | 0.0199 | 0.0007 |
| Infection | Adult | R12 | Kite Surf C | 1 | 0.2418 | 0.6370 | 0.0810 |
| Infection | Adult | R12 | Kite Surf C | 2 | 0.0541 | 0.1798 | 0.0124 |
| Infection | Adult | R12 | Kite Surf C | 3 | 0.0071 | 0.0264 | 0.0020 |
| Infection | Adult | R12 | Kite Surf C | 4 | 0.0005 | 0.0020 | 0.0002 |
| Infection | Adult | R13 | Kite Surf D | 1 | 0.0514 | 0.1940 | 0.0111 |
| Infection | Adult | R13 | Kite Surf D | 2 | 0.0056 | 0.0228 | 0.0008 |
| Infection | Adult | R13 | Kite Surf D | 3 | 0.0003 | 0.0018 | 0.0002 |
| Infection | Adult | R13 | Kite Surf D | 4 | 0.0001 | 0.0002 | 0.0000 |
| Infection | Adult | R14 | Kite Surf E | 1 | 0.0435 | 0.1725 | 0.0084 |
| Infection | Adult | R14 | Kite Surf E | 2 | 0.0044 | 0.0176 | 0.0007 |
| Infection | Adult | R14 | Kite Surf E | 3 | 0.0013 | 0.0025 | 0.0002 |
| Infection | Adult | R14 | Kite Surf E | 4 | 0.0002 | 0.0004 | 0.0000 |
| Infection | Adult | R15 | Kite Surf F | 1 | 0.0460 | 0.1870 | 0.0095 |
| Infection | Adult | R15 | Kite Surf F | 2 | 0.0048 | 0.0214 | 0.0016 |
| Infection | Adult | R15 | Kite Surf F | 3 | 0.0001 | 0.0015 | 0.0001 |
| Infection | Adult | R15 | Kite Surf F | 4 | 0.0000 | 0.0002 | 0.0000 |
| Infection | Adult | Out | Outfall | 1 | 1.0626 | 1.7788 | 0.5450 |
| Infection | Adult | Out | Outfall | 2 | 0.3921 | 0.9346 | 0.1350 |
| Infection | Adult | Out | Outfall | 3 | 0.0879 | 0.3099 | 0.0158 |
| Infection | Adult | Out | Outfall | 4 | 0.0091 | 0.0404 | 0.0010 |
| Illness | Child | R1 | Kite Surf Inner | 1 | 0.3002 | 0.6418 | 0.3958 |
| Illness | Child | R1 | Kite Surf Inner | 2 | 0.1083 | 0.3440 | 0.0966 |
| Illness | Child | R1 | Kite Surf Inner | 3 | 0.0235 | 0.1109 | 0.0127 |
| Illness | Child | R1 | Kite Surf Inner | 4 | 0.0022 | 0.0149 | 0.0015 |
| Illness | Child | R2 | Inshore Kite surf | 1 | 0.0767 | 0.2965 | 0.0561 |
| Illness | Child | R2 | Inshore Kite surf | 2 | 0.0127 | 0.0703 | 0.0056 |
| Illness | Child | R2 | Inshore Kite surf | 3 | 0.0010 | 0.0076 | 0.0007 |
| Illness | Child | R2 | Inshore Kite surf | 4 | 0.0000 | 0.0006 | 0.0000 |
| Illness | Child | R3 | Entrance kite surf | 1 | 0.0310 | 0.1414 | 0.0183 |
| Illness | Child | R3 | Entrance kite surf | 2 | 0.0049 | 0.0245 | 0.0020 |
| Illness | Child | R3 | Entrance kite surf | 3 | 0.0005 | 0.0034 | 0.0004 |
| Illness | Child | R3 | Entrance kite surf | 4 | 0.0001 | 0.0003 | 0.0002 |
| Illness | Child | R4 | Northern swimming | 1 | 0.0090 | 0.0453 | 0.0048 |
| Illness | Child | R4 | Northern swimming | 2 | 0.0008 | 0.0050 | 0.0010 |
| Illness | Child | R4 | Northern swimming | 3 | 0.0001 | 0.0005 | 0.0001 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Illness | Child | R4 | Northern swimming | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Child | R5 | Northern surfing | 1 | 0.0079 | 0.0444 | 0.0045 |
| Illness | Child | R5 | Northern surfing | 2 | 0.0007 | 0.0053 | 0.0005 |
| Illness | Child | R5 | Northern surfing | 3 | 0.0001 | 0.0003 | 0.0000 |
| Illness | Child | R5 | Northern surfing | 4 | 0.0000 | 0.0001 | 0.0000 |
| Illness | Child | R6 | Bar surf | 1 | 0.0329 | 0.1558 | 0.0178 |
| Illness | Child | R6 | Bar surf | 2 | 0.0036 | 0.0203 | 0.0016 |
| Illness | Child | R6 | Bar surf | 3 | 0.0008 | 0.0023 | 0.0003 |
| Illness | Child | R6 | Bar surf | 4 | 0.0001 | 0.0004 | 0.0000 |
| Illness | Child | R7 | Offshore kite surf/Maui | 1 | 0.0307 | 0.1640 | 0.0157 |
| Illness | Child | R7 | Offshore kite surf/Maui | 2 | 0.0036 | 0.0222 | 0.0026 |
| Illness | Child | R7 | Offshore kite surf/Maui | 3 | 0.0002 | 0.0014 | 0.0003 |
| Illness | Child | R7 | Offshore kite surf/Maui | 4 | 0.0000 | 0.0002 | 0.0000 |
| Illness | Child | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.0451 | 0.2142 | 0.0244 |
| Illness | Child | R8 | Western Swimming & Shellfish In Harbour | 2 | 0.0051 | 0.0290 | 0.0027 |
| Illness | Child | R8 | Western Swimming & Shellfish In Harbour | 3 | 0.0002 | 0.0022 | 0.0003 |
| Illness | Child | R8 | Western Swimming & Shellfish In Harbour | 4 | 0.0000 | 0.0001 | 0.0000 |
| Illness | Child | R9 | Domain Recreation/Shellfish | 1 | 0.0184 | 0.1042 | 0.0067 |
| Illness | Child | R9 | Domain Recreation/Shellfish | 2 | 0.0017 | 0.0123 | 0.0001 |
| Illness | Child | R9 | Domain Recreation/Shellfish | 3 | 0.0001 | 0.0009 | 0.0000 |
| Illness | Child | R9 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0002 | 0.0000 |
| Illness | Child | R10 | Maui North | 1 | 0.0202 | 0.1125 | 0.0098 |
| Illness | Child | R10 | Maui North | 2 | 0.0016 | 0.0113 | 0.0010 |
| Illness | Child | R10 | Maui North | 3 | 0.0000 | 0.0011 | 0.0000 |
| Illness | Child | R10 | Maui North | 4 | 0.0000 | 0.0001 | 0.0000 |
| Illness | Child | R11 | Kite Surf B | 1 | 0.2736 | 0.5757 | 0.3386 |
| Illness | Child | R11 | Kite Surf B | 2 | 0.0924 | 0.2974 | 0.0938 |
| Illness | Child | R11 | Kite Surf B | 3 | 0.0234 | 0.1048 | 0.0148 |
| Illness | Child | R11 | Kite Surf B | 4 | 0.0036 | 0.0182 | 0.0014 |
| Illness | Child | R12 | Kite Surf C | 1 | 0.1226 | 0.4123 | 0.1335 |
| Illness | Child | R12 | Kite Surf C | 2 | 0.0320 | 0.1421 | 0.0233 |
| Illness | Child | R12 | Kite Surf C | 3 | 0.0041 | 0.0235 | 0.0023 |
| Illness | Child | R12 | Kite Surf C | 4 | 0.0002 | 0.0021 | 0.0004 |
| Illness | Child | R13 | Kite Surf D | 1 | 0.0341 | 0.1605 | 0.0203 |
| Illness | Child | R13 | Kite Surf D | 2 | 0.0044 | 0.0229 | 0.0023 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Illness | Child | R13 | Kite Surf D | 3 | 0.0001 | 0.0024 | 0.0003 |
| Illness | Child | R13 | Kite Surf D | 4 | 0.0001 | 0.0003 | 0.0000 |
| Illness | Child | R14 | Kite Surf E | 1 | 0.0332 | 0.1566 | 0.0178 |
| Illness | Child | R14 | Kite Surf E | 2 | 0.0036 | 0.0194 | 0.0016 |
| Illness | Child | R14 | Kite Surf E | 3 | 0.0008 | 0.0020 | 0.0003 |
| Illness | Child | R14 | Kite Surf E | 4 | 0.0001 | 0.0004 | 0.0000 |
| Illness | Child | R15 | Kite Surf F | 1 | 0.0307 | 0.1640 | 0.0157 |
| Illness | Child | R15 | Kite Surf F | 2 | 0.0036 | 0.0222 | 0.0026 |
| Illness | Child | R15 | Kite Surf F | 3 | 0.0002 | 0.0014 | 0.0003 |
| Illness | Child | R15 | Kite Surf F | 4 | 0.0000 | 0.0002 | 0.0000 |
| Illness | Child | Out | Outfall | 1 | 0.5738 | 1.1367 | 0.7213 |
| Illness | Child | Out | Outfall | 2 | 0.1963 | 0.5747 | 0.2263 |
| Illness | Child | Out | Outfall | 3 | 0.0616 | 0.2444 | 0.0327 |
| Illness | Child | Out | Outfall | 4 | 0.0078 | 0.0461 | 0.0025 |
| Illness | Adult | R1 | Kite Surf Inner | 1 | 0.4224 | 0.5178 | 0.2854 |
| Illness | Adult | R1 | Kite Surf Inner | 2 | 0.1514 | 0.2704 | 0.0553 |
| Illness | Adult | R1 | Kite Surf Inner | 3 | 0.0249 | 0.0647 | 0.0073 |
| Illness | Adult | R1 | Kite Surf Inner | 4 | 0.0019 | 0.0072 | 0.0007 |
| Illness | Adult | R2 | Inshore Kite surf | 1 | 0.0951 | 0.2026 | 0.0301 |
| Illness | Adult | R2 | Inshore Kite surf | 2 | 0.0126 | 0.0400 | 0.0019 |
| Illness | Adult | R2 | Inshore Kite surf | 3 | 0.0011 | 0.0041 | 0.0001 |
| Illness | Adult | R2 | Inshore Kite surf | 4 | 0.0002 | 0.0001 | 0.0000 |
| Illness | Adult | R3 | Entrance kite surf | 1 | 0.0365 | 0.0861 | 0.0095 |
| Illness | Adult | R3 | Entrance kite surf | 2 | 0.0047 | 0.0125 | 0.0008 |
| Illness | Adult | R3 | Entrance kite surf | 3 | 0.0007 | 0.0017 | 0.0003 |
| Illness | Adult | R3 | Entrance kite surf | 4 | 0.0001 | 0.0002 | 0.0000 |
| Illness | Adult | R4 | Northern swimming | 1 | 0.0095 | 0.0249 | 0.0022 |
| Illness | Adult | R4 | Northern swimming | 2 | 0.0006 | 0.0021 | 0.0004 |
| Illness | Adult | R4 | Northern swimming | 3 | 0.0000 | 0.0003 | 0.0001 |
| Illness | Adult | R4 | Northern swimming | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | R5 | Northern surfing | 1 | 0.0072 | 0.0242 | 0.0022 |
| Illness | Adult | R5 | Northern surfing | 2 | 0.0004 | 0.0025 | 0.0003 |
| Illness | Adult | R5 | Northern surfing | 3 | 0.0001 | 0.0003 | 0.0000 |
| Illness | Adult | R5 | Northern surfing | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | R6 | Bar surf | 1 | 0.0333 | 0.0901 | 0.0079 |
| Illness | Adult | R6 | Bar surf | 2 | 0.0032 | 0.0097 | 0.0008 |
| Illness | Adult | R6 | Bar surf | 3 | 0.0011 | 0.0012 | 0.0002 |
| Illness | Adult | R6 | Bar surf | 4 | 0.0002 | 0.0002 | 0.0000 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Illness | Adult | R7 | Offshore kite surf/Maui | 1 | 0.0349 | 0.0963 | 0.0095 |
| Illness | Adult | R7 | Offshore kite surf/Maui | 2 | 0.0033 | 0.0108 | 0.0016 |
| Illness | Adult | R7 | Offshore kite surf/Maui | 3 | 0.0001 | 0.0005 | 0.0001 |
| Illness | Adult | R7 | Offshore kite surf/Maui | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | R8 | Western Swimming & Shellfish In Harbour | 1 | 0.0247 | 0.1251 | 0.0124 |
| Illness | Adult | R8 | Western Swimming & Shellfish In Harbour | 2 | 0.0026 | 0.0141 | 0.0015 |
| Illness | Adult | R8 | Western Swimming & Shellfish In Harbour | 3 | 0.0000 | 0.0012 | 0.0002 |
| Illness | Adult | R8 | Western Swimming & Shellfish In Harbour | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | R9 | Domain Recreation/Shellfish | 1 | 0.0100 | 0.0545 | 0.0030 |
| Illness | Adult | R9 | Domain Recreation/Shellfish | 2 | 0.0007 | 0.0065 | 0.0000 |
| Illness | Adult | R9 | Domain Recreation/Shellfish | 3 | 0.0000 | 0.0004 | 0.0000 |
| Illness | Adult | R9 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0001 | 0.0000 |
| Illness | Adult | R10 | Maui North | 1 | 0.0101 | 0.0606 | 0.0039 |
| Illness | Adult | R10 | Maui North | 2 | 0.0009 | 0.0058 | 0.0003 |
| Illness | Adult | R10 | Maui North | 3 | 0.0000 | 0.0003 | 0.0000 |
| Illness | Adult | R10 | Maui North | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | R11 | Kite Surf B | 1 | 0.1913 | 0.4567 | 0.2446 |
| Illness | Adult | R11 | Kite Surf B | 2 | 0.0644 | 0.2386 | 0.0579 |
| Illness | Adult | R11 | Kite Surf B | 3 | 0.0150 | 0.0659 | 0.0071 |
| Illness | Adult | R11 | Kite Surf B | 4 | 0.0017 | 0.0099 | 0.0007 |
| Illness | Adult | R12 | Kite Surf C | 1 | 0.0872 | 0.3143 | 0.0810 |
| Illness | Adult | R12 | Kite Surf C | 2 | 0.0196 | 0.0879 | 0.0124 |
| Illness | Adult | R12 | Kite Surf C | 3 | 0.0022 | 0.0120 | 0.0020 |
| Illness | Adult | R12 | Kite Surf C | 4 | 0.0002 | 0.0010 | 0.0002 |
| Illness | Adult | R13 | Kite Surf D | 1 | 0.0186 | 0.0997 | 0.0111 |
| Illness | Adult | R13 | Kite Surf D | 2 | 0.0022 | 0.0126 | 0.0008 |
| Illness | Adult | R13 | Kite Surf D | 3 | 0.0001 | 0.0008 | 0.0002 |
| Illness | Adult | R13 | Kite Surf D | 4 | 0.0001 | 0.0002 | 0.0000 |
| Illness | Adult | R14 | Kite Surf E | 1 | 0.0173 | 0.0878 | 0.0084 |
| Illness | Adult | R14 | Kite Surf E | 2 | 0.0022 | 0.0084 | 0.0007 |
| Illness | Adult | R14 | Kite Surf E | 3 | 0.0005 | 0.0012 | 0.0002 |
| Illness | Adult | R14 | Kite Surf E | 4 | 0.0000 | 0.0002 | 0.0000 |
| Illness | Adult | R15 | Kite Surf F | 1 | 0.0180 | 0.0963 | 0.0095 |
| Illness | Adult | R15 | Kite Surf F | 2 | 0.0017 | 0.0108 | 0.0016 |
| Illness | Adult | R15 | Kite Surf F | 3 | 0.0000 | 0.0005 | 0.0001 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number) | Infection or illness risk | | |
|---|---|---|---|---|---|---|---|
| | | | | | NoV | AdV | EnV |
| Illness | Adult | R15 | Kite Surf F | 4 | 0.0000 | 0.0000 | 0.0000 |
| Illness | Adult | Out | Outfall | 1 | 0.7854 | 0.8963 | 0.5450 |
| Illness | Adult | Out | Outfall | 2 | 0.2909 | 0.4718 | 0.1350 |
| Illness | Adult | Out | Outfall | 3 | 0.0665 | 0.1609 | 0.0158 |
| Illness | Adult | Out | Outfall | 4 | 0.0071 | 0.0229 | 0.0010 |

# Appendix G Illness and Infection risks – Shellfish exposure

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number | Infection or illness risk | |
|---|---|---|---|---|---|---|
| | | | | | NoV | EnV |
| Illness | Adult | S1 | Eastern end of tuatua | 1 | 0.0864 | 0.0297 |
| Illness | Adult | S1 | Eastern end of tuatua | 2 | 0.0126 | 0.0038 |
| Illness | Adult | S1 | Eastern end of tuatua | 3 | 0.0015 | 0.0005 |
| Illness | Adult | S1 | Eastern end of tuatua | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S2 | Mid-point of tuatua | 1 | 0.0665 | 0.0197 |
| Illness | Adult | S2 | Mid-point of tuatua | 2 | 0.0069 | 0.0017 |
| Illness | Adult | S2 | Mid-point of tuatua | 3 | 0.0007 | 0.0001 |
| Illness | Adult | S2 | Mid-point of tuatua | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S3 | Mussel Rocks | 1 | 0.0148 | 0.0036 |
| Illness | Adult | S3 | Mussel Rocks | 2 | 0.0017 | 0.0004 |
| Illness | Adult | S3 | Mussel Rocks | 3 | 0.0001 | 0.0001 |
| Illness | Adult | S3 | Mussel Rocks | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S4 | Western Cockle/Pipi In Harbour | 1 | 0.1687 | 0.0573 |
| Illness | Adult | S4 | Western Cockle/Pipi In Harbour | 2 | 0.0223 | 0.0061 |
| Illness | Adult | S4 | Western Cockle/Pipi In Harbour | 3 | 0.0020 | 0.0003 |
| Illness | Adult | S4 | Western Cockle/Pipi In Harbour | 4 | 0.0001 | 0.0000 |
| Illness | Adult | S5 | Western Swimming & Shellfish In Harbour | 1 | 0.0495 | 0.0139 |
| Illness | Adult | S5 | Western Swimming & Shellfish In Harbour | 2 | 0.0063 | 0.0022 |
| Illness | Adult | S5 | Western Swimming & Shellfish In Harbour | 3 | 0.0006 | 0.0002 |
| Illness | Adult | S5 | Western Swimming & Shellfish In Harbour | 4 | 0.0001 | 0.0001 |
| Illness | Adult | S6 | Western Shellfish In Harbour A | 1 | 0.0231 | 0.0059 |
| Illness | Adult | S6 | Western Shellfish In Harbour A | 2 | 0.0023 | 0.0007 |
| Illness | Adult | S6 | Western Shellfish In Harbour A | 3 | 0.0003 | 0.0001 |
| Illness | Adult | S6 | Western Shellfish In Harbour A | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S7 | Western Shellfish In Harbour B | 1 | 0.0111 | 0.0025 |
| Illness | Adult | S7 | Western Shellfish In Harbour B | 2 | 0.0012 | 0.0001 |
| Illness | Adult | S7 | Western Shellfish In Harbour B | 3 | 0.0001 | 0.0000 |
| Illness | Adult | S7 | Western Shellfish In Harbour B | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S8 | Mid Harbour Shellfish | 1 | 0.0078 | 0.0021 |
| Illness | Adult | S8 | Mid Harbour Shellfish | 2 | 0.0012 | 0.0005 |
| Illness | Adult | S8 | Mid Harbour Shellfish | 3 | 0.0002 | 0.0001 |
| Illness | Adult | S8 | Mid Harbour Shellfish | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S9 | Inner Harbour Shellfish A | 1 | 0.0082 | 0.0018 |
| Illness | Adult | S9 | Inner Harbour Shellfish A | 2 | 0.0007 | 0.0003 |
| Illness | Adult | S9 | Inner Harbour Shellfish A | 3 | 0.0001 | 0.0000 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number | Infection or illness risk | |
|---|---|---|---|---|---|---|
| | | | | | NoV | EnV |
| Illness | Adult | S9 | Inner Harbour Shellfish A | 4 | 0.0001 | 0.0000 |
| Illness | Adult | S10 | Inner Harbour Shellfish B | 1 | 0.0097 | 0.0013 |
| Illness | Adult | S10 | Inner Harbour Shellfish B | 2 | 0.0005 | 0.0002 |
| Illness | Adult | S10 | Inner Harbour Shellfish B | 3 | 0.0000 | 0.0000 |
| Illness | Adult | S10 | Inner Harbour Shellfish B | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S11 | Inner Harbour Shellfish C | 1 | 0.0078 | 0.0020 |
| Illness | Adult | S11 | Inner Harbour Shellfish C | 2 | 0.0011 | 0.0001 |
| Illness | Adult | S11 | Inner Harbour Shellfish C | 3 | 0.0000 | 0.0000 |
| Illness | Adult | S11 | Inner Harbour Shellfish C | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S12 | Inner Harbour Shellfish D | 1 | 0.0129 | 0.0024 |
| Illness | Adult | S12 | Inner Harbour Shellfish D | 2 | 0.0012 | 0.0000 |
| Illness | Adult | S12 | Inner Harbour Shellfish D | 3 | 0.0000 | 0.0000 |
| Illness | Adult | S12 | Inner Harbour Shellfish D | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S13 | Domain Recreation/Shellfish | 1 | 0.0090 | 0.0022 |
| Illness | Adult | S13 | Domain Recreation/Shellfish | 2 | 0.0007 | 0.0003 |
| Illness | Adult | S13 | Domain Recreation/Shellfish | 3 | 0.0001 | 0.0001 |
| Illness | Adult | S13 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0000 |
| Illness | Adult | S14 | Inner Harbour Shellfish | 1 | 0.0134 | 0.0029 |
| Illness | Adult | S14 | Inner Harbour Shellfish | 2 | 0.0014 | 0.0004 |
| Illness | Adult | S14 | Inner Harbour Shellfish | 3 | 0.0004 | 0.0000 |
| Illness | Adult | S14 | Inner Harbour Shellfish | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S1 | Eastern end of tuatua | 1 | 0.1166 | 0.0297 |
| Infection | Adult | S1 | Eastern end of tuatua | 2 | 0.0169 | 0.0038 |
| Infection | Adult | S1 | Eastern end of tuatua | 3 | 0.0016 | 0.0005 |
| Infection | Adult | S1 | Eastern end of tuatua | 4 | 0.0001 | 0.0000 |
| Infection | Adult | S2 | Mid-point of tuatua | 1 | 0.0901 | 0.0197 |
| Infection | Adult | S2 | Mid-point of tuatua | 2 | 0.0098 | 0.0017 |
| Infection | Adult | S2 | Mid-point of tuatua | 3 | 0.0007 | 0.0001 |
| Infection | Adult | S2 | Mid-point of tuatua | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S3 | Mussel Rocks | 1 | 0.0191 | 0.0036 |
| Infection | Adult | S3 | Mussel Rocks | 2 | 0.0023 | 0.0004 |
| Infection | Adult | S3 | Mussel Rocks | 3 | 0.0001 | 0.0001 |
| Infection | Adult | S3 | Mussel Rocks | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S4 | Western Cockle/Pipi In Harbour | 1 | 0.2385 | 0.0573 |
| Infection | Adult | S4 | Western Cockle/Pipi In Harbour | 2 | 0.0331 | 0.0061 |
| Infection | Adult | S4 | Western Cockle/Pipi In Harbour | 3 | 0.0029 | 0.0003 |
| Infection | Adult | S4 | Western Cockle/Pipi In Harbour | 4 | 0.0002 | 0.0000 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number | Infection or illness risk | |
|---|---|---|---|---|---|---|
| | | | | | NoV | EnV |
| Infection | Adult | S5 | Western Swimming & Shellfish In Harbour | 1 | 0.0693 | 0.0139 |
| Infection | Adult | S5 | Western Swimming & Shellfish In Harbour | 2 | 0.0081 | 0.0022 |
| Infection | Adult | S5 | Western Swimming & Shellfish In Harbour | 3 | 0.0008 | 0.0002 |
| Infection | Adult | S5 | Western Swimming & Shellfish In Harbour | 4 | 0.0001 | 0.0001 |
| Infection | Adult | S6 | Western Shellfish In Harbour A | 1 | 0.0294 | 0.0059 |
| Infection | Adult | S6 | Western Shellfish In Harbour A | 2 | 0.0029 | 0.0007 |
| Infection | Adult | S6 | Western Shellfish In Harbour A | 3 | 0.0004 | 0.0001 |
| Infection | Adult | S6 | Western Shellfish In Harbour A | 4 | 0.0001 | 0.0000 |
| Infection | Adult | S7 | Western Shellfish In Harbour B | 1 | 0.0145 | 0.0025 |
| Infection | Adult | S7 | Western Shellfish In Harbour B | 2 | 0.0014 | 0.0001 |
| Infection | Adult | S7 | Western Shellfish In Harbour B | 3 | 0.0001 | 0.0000 |
| Infection | Adult | S7 | Western Shellfish In Harbour B | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S8 | Mid Harbour Shellfish | 1 | 0.0107 | 0.0021 |
| Infection | Adult | S8 | Mid Harbour Shellfish | 2 | 0.0014 | 0.0005 |
| Infection | Adult | S8 | Mid Harbour Shellfish | 3 | 0.0002 | 0.0001 |
| Infection | Adult | S8 | Mid Harbour Shellfish | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S9 | Inner Harbour Shellfish A | 1 | 0.0082 | 0.0018 |
| Infection | Adult | S9 | Inner Harbour Shellfish A | 2 | 0.0007 | 0.0003 |
| Infection | Adult | S9 | Inner Harbour Shellfish A | 3 | 0.0001 | 0.0000 |
| Infection | Adult | S9 | Inner Harbour Shellfish A | 4 | 0.0001 | 0.0000 |
| Infection | Adult | S10 | Inner Harbour Shellfish B | 1 | 0.0097 | 0.0013 |
| Infection | Adult | S10 | Inner Harbour Shellfish B | 2 | 0.0005 | 0.0002 |
| Infection | Adult | S10 | Inner Harbour Shellfish B | 3 | 0.0000 | 0.0000 |
| Infection | Adult | S10 | Inner Harbour Shellfish B | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S11 | Inner Harbour Shellfish C | 1 | 0.0078 | 0.0020 |
| Infection | Adult | S11 | Inner Harbour Shellfish C | 2 | 0.0011 | 0.0001 |
| Infection | Adult | S11 | Inner Harbour Shellfish C | 3 | 0.0000 | 0.0000 |
| Infection | Adult | S11 | Inner Harbour Shellfish C | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S12 | Inner Harbour Shellfish D | 1 | 0.0129 | 0.0024 |
| Infection | Adult | S12 | Inner Harbour Shellfish D | 2 | 0.0012 | 0.0000 |
| Infection | Adult | S12 | Inner Harbour Shellfish D | 3 | 0.0000 | 0.0000 |
| Infection | Adult | S12 | Inner Harbour Shellfish D | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S13 | Domain Recreation/Shellfish | 1 | 0.0090 | 0.0022 |
| Infection | Adult | S13 | Domain Recreation/Shellfish | 2 | 0.0007 | 0.0003 |
| Infection | Adult | S13 | Domain Recreation/Shellfish | 3 | 0.0001 | 0.0001 |
| Infection | Adult | S13 | Domain Recreation/Shellfish | 4 | 0.0000 | 0.0000 |
| Infection | Adult | S14 | Inner Harbour Shellfish | 1 | 0.0134 | 0.0029 |

| Nature of risk | Receptor | Site code | Site name | Level of treatment (log number | Infection or illness risk | |
|---|---|---|---|---|---|---|
| | | | | | NoV | EnV |
| Infection | Adult | S14 | Inner Harbour Shellfish | 2 | 0.0014 | 0.0004 |
| Infection | Adult | S14 | Inner Harbour Shellfish | 3 | 0.0004 | 0.0000 |
| Infection | Adult | S14 | Inner Harbour Shellfish | 4 | 0.0000 | 0.0000 |