NIWA
Taihoro Nukurangi

Blenheim Sewage Treatment Plant Discharge: A Quantitative Microbial Risk Assessment

Prepared for Marlborough District Council

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Executive summary

The Blenheim Sewage Treatment Plant (BSTP) is located approximately 6 km southeast of Blenheim. Marlborough District Council (MDC) operates the plant and has resource consent to allow treated effluent to be discharged to the Wairau Estuary and the land surrounding the plant. MDC is seeking to renew the consent and has commissioned NIWA to assess the potential human health risks associated with primary contact recreation (e.g., swimming) and consumption of raw shellfish harvested following exposure to diluted treated effluent discharged to the estuary. This evaluation uses a quantitative microbial risk assessment (QMRA) and environmental monitoring results.

BSTP discharges treated effluent twice daily into the Wairau Estuary on the outgoing (ebb) tide, which flows into Cloudy Bay. DHI (DHI Water & Environment Ltd) ran a hydrodynamic model to simulate the dispersion and dilution of treated effluent in the environment. The hydrodynamic modelling results were used as key inputs to this QMRA, allowing estimates of Individual Illness Risks (IIR) to be calculated for eleven sites: five within the Wairau Estuary, one at the bar/entrance to the estuary, and five in Cloudy Bay. The estimated risks were incremental risks related to the discharge of effluent from BSTP into the environment. Evidence from the State of the Environment reporting of the Wairau and Ōpaoa Rivers suggests the background levels of microbial contamination from the rivers were low. There is little evidence of any other major source of microbial contamination in the estuary apart from the effluent discharge.

The estimated risks from the QMRA varied by the type of risk, spatially and with the level of treatment of the effluent. Risks from shellfish consumption were higher than from primary contact. Risks were higher in areas that were within the effluent plume. Therefore, risk in some parts of the estuary are much higher than in other parts. Overall, the highest risks were associated with contact with effluent at the outfall during discharge, where dilution was minimal. Higher levels of treatment, as measured in terms of Log Reduction Value (LRV), led to lower risk.

Shellfish risks tended to be highest in the estuary at the mid mixing zone (MM), and at the bar entrance (BE), and of course, at the point of outfall. Swimming risks were highest at the bar entrance (BE) followed by a site in Cloudy Bay called Gorse Surf Break (GB). The difference in relative risks reflects the dynamics of the effluent plume, which travels along the bed of the estuary before upwelling and coming to the surface at the bar. Shellfish risks relate to water quality at the bed, whereas swimming risks relate to water quality at the surface. The dynamics of the effluent plume also explain why a site only 230 m from the outfall (Boat Launching Wharf) had the lowest swimming and shellfish risks of any site. Due to the timing of the discharge on the ebb tide, effluent from the BSTP flows out into Cloudy Bay rather than being dispersed throughout the Wairau Estuary.

The results of the current QMRA were compared with a QMRA run in 2007. There are several differences between the studies, not least the use of norovirus as the reference pathogen in the study and rotavirus in the previous study. The difference means it is not possible to compare the results across studies directly. However, both studies show similarities in terms of the spatial distribution of risks.

The calculated risks are strongly related to LRV. Higher levels of treatment, as measured by LRV, the lower the public health risks. The LRV for a sewage treatment plant depends on the treatment systems in place and how well they operate. BSTP has domestic and industrial waste streams. The domestic stream passes through a fine screen and facultative and maturation ponds before mixing with the industrial stream and then passing into a wetland. The LRV for viruses in the BSTP ponds has not been measured. In a 2007 QMRA, LRV was assumed to be in the range of 2.4 to 3.7, but since then knowledge has increased so a brief literature search was carried out to gain an understanding the expected LRV of the treatment system. In the literature, pond systems (the closest comparator for the BSTP) are reported to have LRV ranging from 0.4 to 5 for viruses. The average levels for a pond system with a similar hydrological retention time as BSTP are expected to be in the range of 1.6 to 2.4. On top of virus removal by ponds, further reductions are expected to occur in the wetland. Studies of virus removal in a wetland are very limited with the reported LRV values in the range of 0.5 to 3. For the BSTP, it was noted that the LRV for faecal indicator bacteria (FIB) in the wetland was approximately 1.4 and given that viruses tend to be more resistant to treatment that bacteria, the virus LRV is likely to be less than 1.4 and possibly more than 0.5.

There is no bottom line for health risks in the marine environment for recreational contact with water or shellfish consumption. Therefore, in this work, I have used the A/B boundary of the Microbiological Assessment Category from the marine guidelines (MfE and MoH 2003) which equates to a 1% illness risk as a comparator. If a 1% IIR target was chosen, LRV = 2 would be sufficient to meet that target for swimming, excluding the outfall site. To keep risks below 1% for shellfish for all sites, excluding the outfall, LRV = 4 would be required. This would fall to an LRV = 3 if only sites in the bay were considered.

Risks are greatest for consumers of raw shellfish. Further work needs to be done to establish whether the current level of treatment and resulting LRV is sufficient to reduce the public health risks to an acceptable level. If the limited and incomplete microbial water quality and shellfish monitoring data, is representative, it suggests shellfish collected in areas impacted by the effluent plume from the BSTP, may not comply with the 'Guidelines for Shellfish-Gathering Waters' and their flesh may occasionally contain levels of FIB organisms indicating they are unsafe for human consumption.

1 Introduction

The Blenheim Sewage Treatment Plant (BSTP) is located approximately 6 km southeast of Blenheim. Marlborough District Council (MDC) operates the plant and has resource consent to allow treated effluent to be discharged to the Wairau Estuary and the land surrounding the plant. MDC is seeking to renew the consent and has commissioned NIWA to assess the potential human health risks associated with primary contact recreation (e.g., swimming) and consumption of raw shellfish harvested following exposure to diluted treated effluent discharged to the estuary. This evaluation uses a quantitative microbial risk assessment (QMRA) and environmental monitoring results.

BSTP takes domestic and industrial untreated wastewater from Blenheim, Marlborough Ridge, Renwick, Riverlands and Cloudy Bay Business Park. According to Beca Ltd (2023) the domestic untreated wastewater passes through a fine screen and facultative and maturation ponds, while the industrial wastewater is treated using fine screening and mechanically aerated and facultative ponds. The two wastewater streams are combined before flowing through a wetland before discharge. A schematic diagram can be found in Figure A-1 in the appendix.

Treated effluent may be discharged through the larger of two outfall pipes (Figure 1-1) into the estuary or irrigated to land when conditions allow (especially in summer) but some flows are conveyed to the outfall to prevent blockages. Beca is currently assessing options to mitigate the blockage issue which will allow MDC to direct more effluent to land.

Figure 1-1: Map showing the location of the outfall, risk assessment sites and features in and around Wairau Estuary.

This report assessed the human health risks associated with recreational exposure to effluent from the BSTP. Raw sewage may contain a wide range of pathogens that harm human health. Treatment reduces the levels of pathogens but does not necessarily remove them all, so treated effluent may be a hazard to health. Estimates of recreational exposure risks are often based on the use of indicator organisms, such as *Escherichia coli* (*E. coli)*, faecal coliforms and enterococci.

In the case of effluent discharges, the treatment processes have the potential to alter the relationship between pathogens and indicators, so compliance with guideline values does not guarantee safety (MfE and MoH 2003). In this case, Quantitative Microbial Risk Assessment (QMRA) is used to estimate the likely risk to health alongside a description of the microbial quality of the effluent and receiving environment. The QMRA estimates the incremental health risks associated with BSTP, which are over and above risks from any other source.

Layout of the report

The report is laid out into sections:

- Section 2 describes the microbiological context, which includes an assessment of the microbial quality of the receiving environment and the inputs to the receiving environment from treated effluent and other sources.
- Section 3 describes the methodology for the QMRA, the parameters used for modelling health risks, and the resulting modelled health risks.
- Section 4 summarises the potential public health impact of the BSTP discharge for recreational water users and consumers of raw shellfish.

2 Microbiological context

This health risks assessment includes an investigation of the microbiological quality of the receiving environment and the inputs to the receiving environment. This assessment primarily focused on faecal indicator bacteria (FIB) that could arise from various contaminant sources (e.g., diffuse, urban runoff, agriculture, wildlife) in addition to the Blenheim Sewage Treatment Plant (BSTP). Data from monitoring wastewater discharge volumes and concentrations of faecal coliforms, *E. coli*, and enterococci in the discharge and receiving environment, as well as the quality of river water from the Wairau and Ōpaoa Rivers, were used to establish the "microbiological context" for the Wairau Estuary, the local receiving environment.

The data were used to indicate the potential risks from human contact with diluted treated effluent as well as other sources of pathogens in the Wairau Estuary. This assessment is based on criteria and guideline values from the New Zealand Recreational Water Quality Guidelines (MfE and MoH 2003) and the National Policy Statement for Freshwater Management (Ministry for the Environment 2024).

It is important to note that the guidelines should not be used alone to assess risks from effluent discharges, as the treatment process may alter the relationship between FIB and pathogens. This lack of a reliable relationship is one of the main reasons the QMRA (Quantitative Microbial Risk Assessment) approach is used to estimate risk.

However, analysing FIB data and comparing them with the guidelines and other frameworks does offer another way to estimate the existing health risk to water users from contamination sources other than effluent, especially during recreational or bathing seasons. Data were also used to identify spatial patterns of risk.

2.1 Data sources

Information about BSTP, the receiving environment (Wairau Estuary) and other potential inputs to the receiving environment came from several sources:

- Beca provided information from routine monitoring of water quality and quantity of the BSTP. In addition to the routine monitoring, information from a short norovirus monitoring campaign was provided.
- Information about the receiving environment came from a report on the Blenheim Sewage Treatment Plant receiving environment monitoring by Mitterwallner and Sneddon (2024).
- Information on the Wairau and Ōpaoa Rivers, two inputs to the Wairau Estuary, were obtained from LAWA. However, information on the microbial quality of water from the Wairau Lagoon or Cloudy Bay were not available.
- A site visit was carried out on 15th August 2024.

As is common in microbiological data, some of the observations are censored, and their actual values are unknown. When this happens, the observations are recorded as "less than" or "greater than" values. In the case of "less than", the actual values are below the detection limit for the test and "greater than" values are above the value that can be enumerated for the test. To simplify the analysis, the censored values were replaced by the respective detection and enumeration limit values.

2.2 BSTP data

This part of the context report focuses on data from the BSTP, including volume of effluent and microbial data.

2.2.1 Volume of effluent discharge

Discharge data for the outfall was provided for a four-year period covering 1 July 2018 to 30 June 2022, see Figure 2-1. The maximum daily discharge was 46,625 m^3/day, and the minimum 12.6 m^3/day with a median value of 15,719 m^3/day. Over the period of the record, there was no discernible overall trend in discharge. However, it was noted that discharge was higher in the winter months (July and August) than in the summer months (December, January and February).

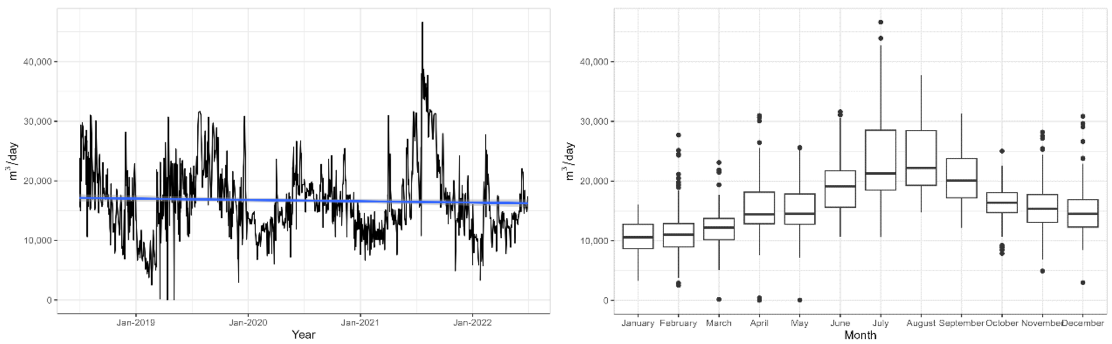

Figure 2-1: Daily and monthly discharge into the estuary from BSTP. The figure on the left indicates the daily discharge between 2018 and 2022. However, there is considerable variation over time, and there was no indication to suggest that the volume of effluent was increasing or decreasing. The figure on the right is a boxplot grouping daily discharge by month. Discharge tended to be higher during the months of June to September.

Domestic effluent accounts for over 90% of the BSTP's total volume, while industrial effluent accounts for less than 10%, but this varies by time of year.

2.2.2 Microbial quality of effluent discharge

Two faecal indicator bacteria (FIB), enterococci and faecal coliforms, are monitored at the outlet of pond 14, from where it is discharged to the estuary or land. The results from 2018 to 2024 are presented in Figure 2-2. Between 2018 and 2020 for enterococci and 2021 for faecal coliforms, there was a trend of increasing FIB concentrations followed by a decrease until 2024, which is the latest data. The decrease was more marked for enterococci than faecal coliforms. Censoring of data can influence trend estimation, and it should also be noted that a higher proportion of the enterococci results (30 out of 138 observations or 22%) were censored as opposed to faecal coliforms measurements (13 out of 139 observations or 9%).

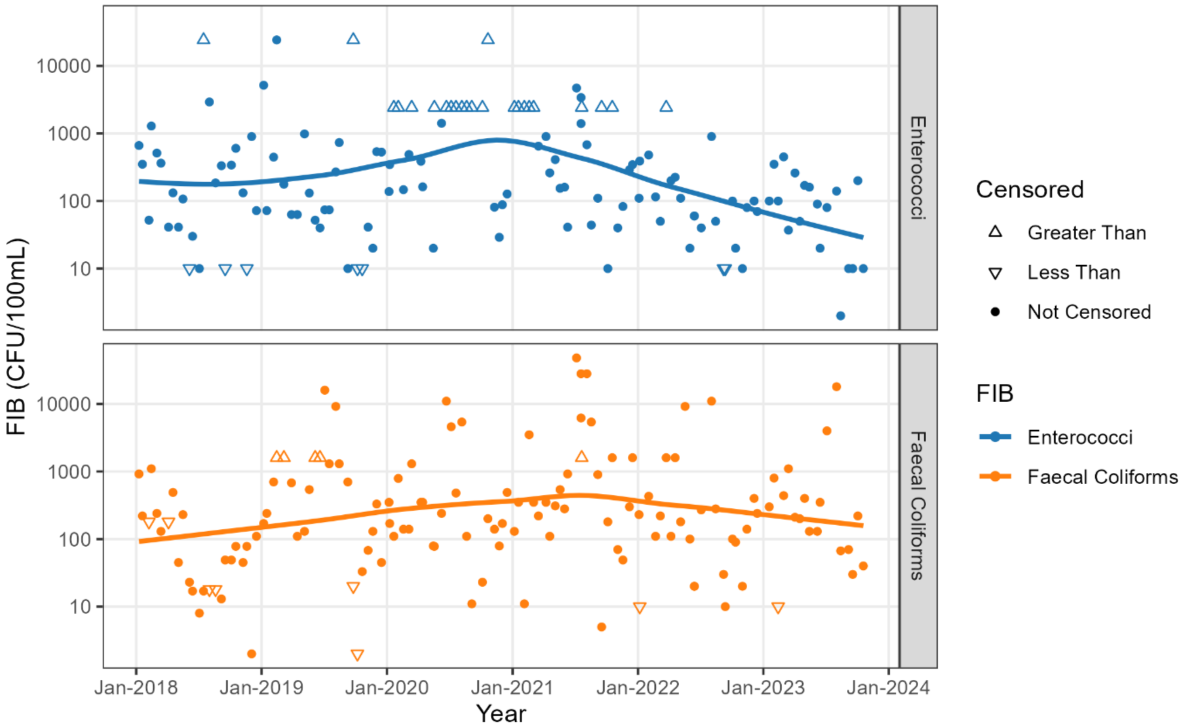

Figure 2-2: Time series of observed FIB levels in the discharge. A trend line has been superimposed on the data, illustrating a general increase in concentration up to 2020-2021, followed by a fall. Note the y-axis is plotted on a logarithmic scale. The trend line has been estimated using loess regression.

Consent condition 62 for BSTP sets the median and 90th percentile values for faecal coliforms to 700 and 2,150 CFU/100 mL, respectively. The annual estimated values, in terms of calendar year, are presented in Table 2-1. The result may differ from those published elsewhere as multiple ways exist to estimate percentiles.

Table 2-1: Levels of faecal coliforms in the effluent discharge (pond 14) by calendar year. The median and 90th percentile values have been estimated using the Hazen method. Note that the 17–18 and 23–24 data only cover part of the financial year. Financial year runs from 1 July – 30 June. Censorship rate is the proportion of observations recorded as "less than" or "greater than" values.

Financial year	Median (CFU/100 mL)	90th percentile (CFU/100 mL)	Observation count	Censorship Rate (%)
17-18	200	974	12	0
18-19	110	1 600	23	17.4
19-20	240	9 200	25	0
20-21	280	3 720	23	0
21-22	300	24 240	27	3.7
22-23	225	890	22	0
23-24	70	15 200	7	0

There are guideline values for marine and freshwater recreational areas (MfE and MoH 2003). Although it is not expected that effluent from BSTP would meet those values, comparing the results against them is instructive. There is a shellfish water quality standard (Section F of the guideline) set at a median value of 14 MPN/100 mL and not more than 10% of samples exceeding 43 MPN/100 mL (equivalent to the 90th percentile) for faecal coliforms. A set of Microbiological Assessment Categories (MAC) for swimming /recreational contact with water (Section H of the Guidelines), presented in Table 2-2, uses enterococci as the FIB.

Therefore, for every year reported in Table 2-1, the undiluted effluent at the point of discharge into the estuary would be unsuitable for shellfish harvesting. Comparing the results in Table 2-3 against Table 2-2 would place the effluent in the D category for years 17–18 to 22–23, implying a gastrointestinal illness (GI) risk of greater than 10% per event, such as a swim. There was insufficient information to estimate risk for the 23–24 year.

Table 2-2: Risks associated with 95th percentile concentration of enterococci in marine waters. From MfE/MoH (2003) guidelines, Table H1, and associated text. The colour coding is used in other parts of the report.

Microbiological Assessment Category (MAC)	Threshold value (95th percentile value of enterococci/100 mL)	Gastrointestinal Illness Risk (%)	Acute Febrile Respiratory Illness Risk (%)
A	≤ 40	<1	<0.3%
B	40-200	1-5	0.3-≤ 1.9
C	201-500	5-10	1.9-3.9
D	>500	>10	>3.9

Table 2-3: Levels of enterococci in the effluent discharge (pond 14) by calendar year. The 95th percentile values have been estimated using the Hazen method. Note that the data for 17–18 and 23–24 only cover part of the financial year, and there were insufficient observations to estimate the 95th percentile in 23–24. Censorship rate is the proportion of observations recorded as "less than" or "greater than" values. Financial year runs from 1 July – 30 June.

Financial year	95th percentile (CFU/100 mL)	Observation count	Censorship Rate (%)	Microbiological Assessment Category (MAC)
17-18	1 225	12	0	D
18-19	24 200	23	4.3	D
19-20	7 865	25	24	D
20-21	10 043	23	52.2	D
21-22	3 660	26	15.4	D
22-23	630	22	0	D
23-24	Insufficient data	7	0	Insufficient data

Focusing on the temporal variability in FIB measurements, Figure 2-3 drills down into the data to investigate if there are any apparent seasonal pattern to the FIB monitoring results. The 2018 data are quite different to the subsequent years and plotted separately from the 2019–2023 monthly data. The 2019–2023 faecal coliforms have higher levels in the months of July and August. Slightly lower levels were found in October and November, and similar values were found in January, but not June. Given the amount of censoring of the enterococci data, it is difficult to make out distinct seasonal patterns, if they exist. However, it does appear that September and October have lower values.

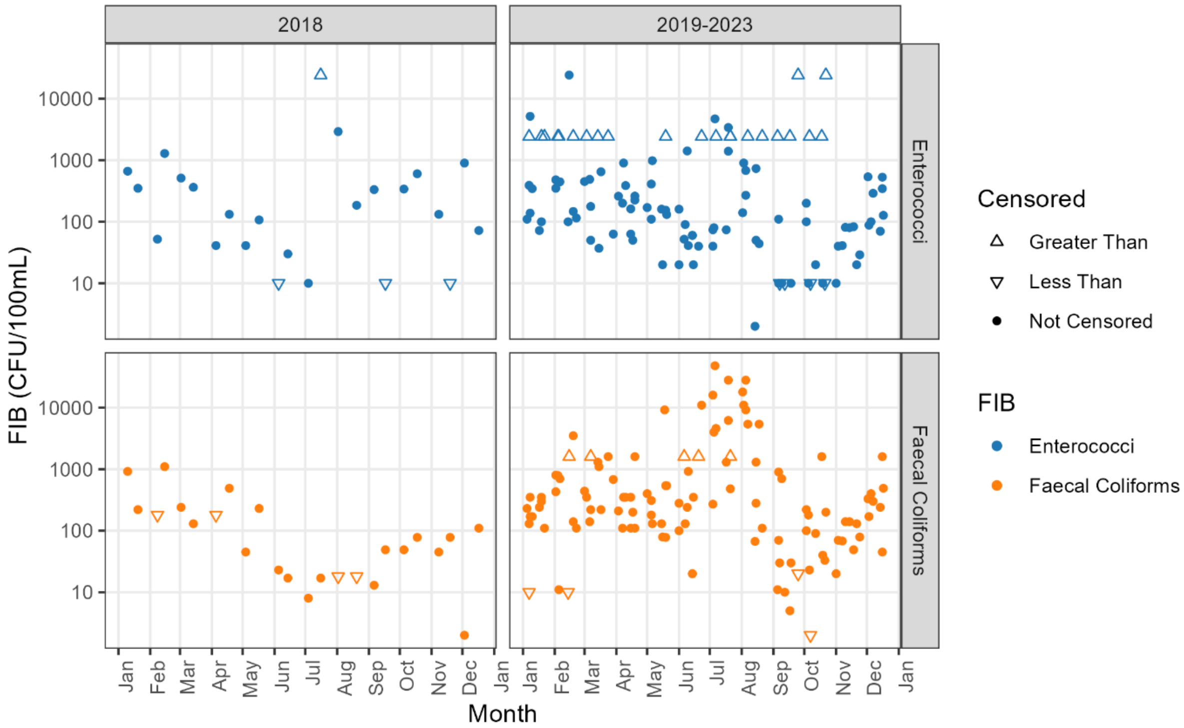

Figure 2-3: Seasonal effluent discharge characteristics, 2018–2023 calendar years. There appears to be a more distinct seasonal pattern for faecal coliforms than for enterococci.

2.2.3 Removal of FIBs by the wetland

The wetlands provide the last stage of the treatment before discharge. Information about the FIB concentration of effluent flowing into and out of the wetlands was provided. This illustrates that the ponds removed FIBs, as illustrated by the fact that the average concentration in the outflow is less than the inflow, as shown in Figure 2-4. This provides insight as to the efficacy of treatment in terms of FIB removal, but the results cannot be extended to pathogens such as viruses, as the relationship between pathogens and FIB has not been established.

The median concentration of faecal coliforms in the inflow (from pond 6) was 5400 CFU/100 mL based on 60 observations and 220 CFU/100 mL based on 139 observations for the outflow (pond 14). This suggests an average LRV of 1.4 for faecal coliforms for the wetland for the period between 2018 and 2024. Enterococci influent concentrations were not measured after 2021. Given the limited number of data points, an enterococci LRV was not estimated.

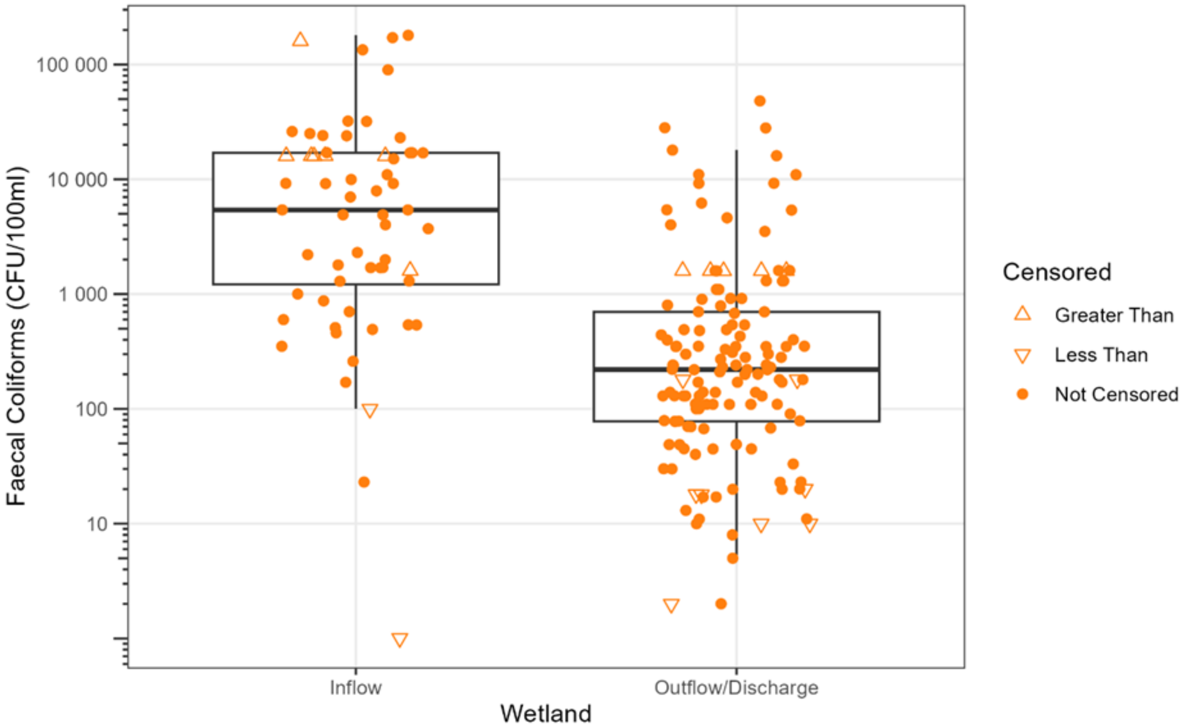

Figure 2-4: Boxplot illustrating the reduction of the faecal coliform concentration by the wetlands. Higher concentrations were measured in the inflow (Pond 6) than in the outflow (Pond 14), which is discharged into the environment.

2.2.4 FIB Microbial Load/Flux

Combining the FIB with the corresponding daily discharge provides an estimate of the flux or daily load discharged into the estuary. The flux probably provides a better estimate of the impact on receiving waters than concentration alone because it takes into account the volume of discharge from the plant.

The median daily flux varied from year to year, ranging from 4.6×10^{9} to 6.9×10^{10} CFU/day for faecal coliforms, to 1.4×10^{10} to 2.0×10^{11} CFU/day for enterococci (see Figure 2-5). There was greater variation between months, with the highest median fluxes in July, at 6.9×10^{11} and 4.9×10^{11} CFU/day for faecal coliforms and enterococci respectively. The lowest median daily faecal coliform concentration was in September, with 5.5×10^{9} CFU/day, and 6.3×10^{9} CFU/day for enterococci in November (see Figure 2-6). It should also be noted that the actual levels vary widely around the median value, with faecal coliform measurements varying from less than 1×10^{9} up to 1×10^{13} CFU/day.

Higher winter flows may contribute to the greater fluxes of FIB organisms in the winter months; however, flow on its own would not be able to account for the increase in flux. Other factors may also be involved, such as lower removal of FIB by the treatment system during the winter months. FIBs are also known to occasionally multiply in treatment systems (Moriarty et al. 2008).

The observations about fluxes are limited to FIB the relationship between pathogens and FIB has not been established.

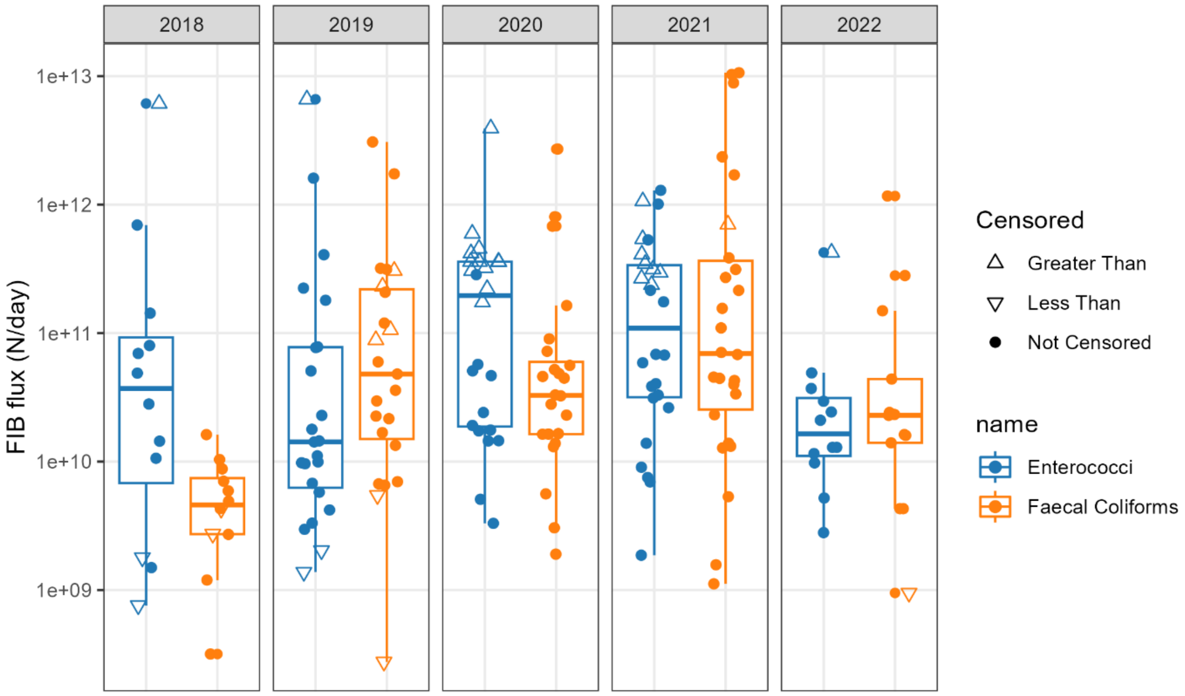

Figure 2-5: Box plot of the daily effluent FIB flux values for the 2018–2022 calendar years. Note only partial years for 2018 and 2022, with estimates starting in July 2018 and ending in June 2022.

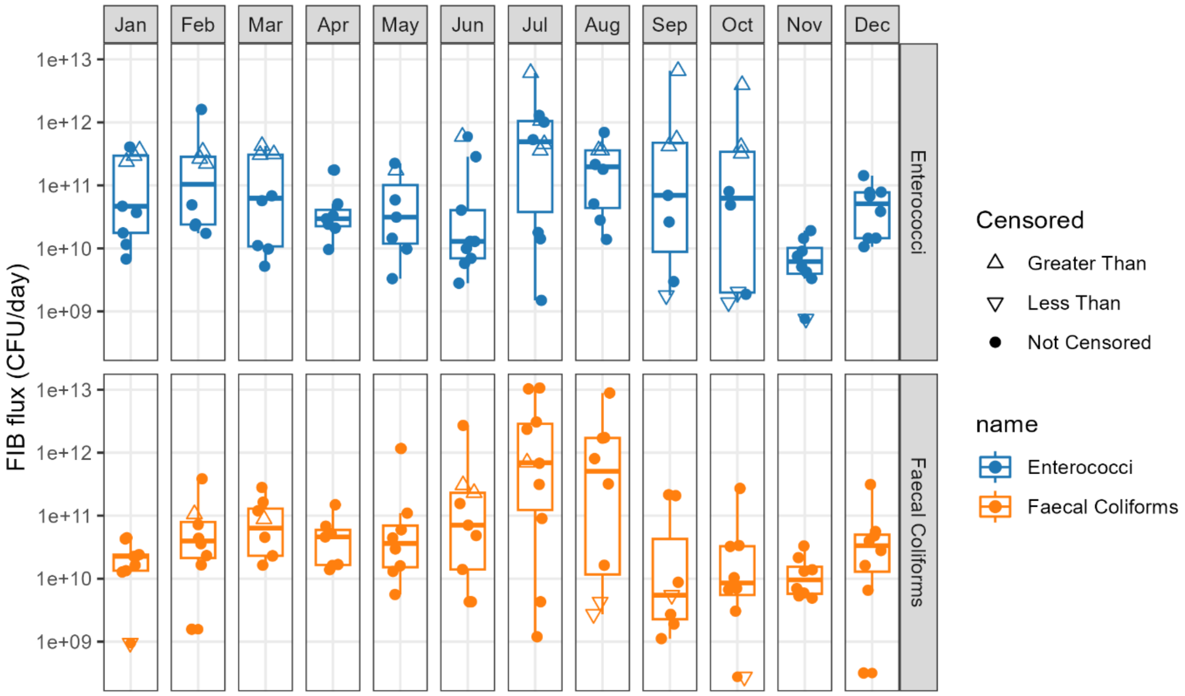

Figure 2-6: Monthly flux estimates. Using data of FIB and outflows from July 2018 to June 2022. Note that the enterococci data is censored, so the estimates, particularly in July, may be underestimated.

2.2.5 Norovirus

There was a short campaign to estimate the level of norovirus in the domestic influent and at the end of the pond system before entering the wetland which is presented in Section 3.2.2 below.

2.3 Other potential inputs to the Wairau Estuary, the receiving environment

Potential sources of pathogens can add to the recreation risk from exposure to water in the Wairau Estuary, the receiving environment for the BSTP discharge, apart from BSTP discharge itself. Additional potential sources of faecal contamination in the Wairau Estuary include water flowing into the estuary from the Wairau and Ōpaoa Rivers and tidally driven inputs from the Wairau Lagoon and Cloudy Bay. Contamination could also be deposited directly into the estuary or from shore.

FIB measurements for the Wairau and Ōpaoa Rivers are available from the LAWA website. The descriptive statistics are presented in Table 2-4, focusing on the most downstream monitoring sites. The water quality data falls into Attribute Band A from the National Policy Statement for Freshwater Management (NPS-FM) (Ministry for the Environment 2024). This implies that the average risk of infection is less than 1% for swimmers per swim. The actual risk will vary from day to day.

Table 2-4: Monitoring results from the most downstream sites on the Wairau and Ōpaoa Rivers compared with Table 9 of the NPS-FM. Two sites are in band A and imply that the infection risk from swimming in the water is less than 1%.

River	Site Name	% exceedances over 540 #/100 mL	% exceedances over 260 #/100 mL	Median concentration #/100 mL	95th percentile of *E. coli* #/100 mL	Attribute Band	No samples
Wairau	Wairau River at Blenheim Rowing Club	3.2	4.8	31	245.8	A	126
Ōpaoa	Ōpaoa River at Swamp Road	3.2	4.3	36	222.5	A	185

No other quantitative FIB data was available. During a visit to the Wairau Estuary, the surrounding land was noted to be flat. The area was rural, with grass, gravel, and marsh areas in some locations. The immediate land use was recreation, primarily on the north bank of the estuary. Agriculture occurs on the rivers upstream of the estuary, and the town of Blenheim is situated on the Ōpaoa River. BSTP is a significant land-use in an area adjoining the southern end of the estuary.

Regarding recreation, there is fishing and boating on the estuary. Along the shore, there is freedom camping, walking, birdwatching, etc. There is a toilet block to serve people involved in recreational activities, and a few houses.

It was also noted that there are significant populations of birds in the Wairau Estuary and Lagoon (DOC ND), and bird populations can impact water quality (Grant et al. 2001).

There may be some direct inputs of treated/untreated effluent into the estuary apart from BSTP. We note the river water quality is high overall, and there is no urban or intensive agriculture on the estuary's shores. Thus, the BSTP is probably the most significant source of effluent entering the estuary.

2.4 The estuary/receiving environment

Cawthron Institute (Mitterwallner and Sneddon 2024) conducted a microbiological sampling of the BSTP receiving environment on a single day in February 2024. Microbiological information on water quality and shellfish was collected. Similar surveys have been conducted in the past. Sneddon (2024) also performed a quantitative dye study to estimate effluent dilution upon discharge into the environment and inform dilution modelling.

The BSTP discharge occurs on the ebb tide. The observations made by Sneddon (2024) suggest a highly elongated and slightly meandering plume of effluent in the estuary after effluent discharge. The plume tends to stay on the estuary's right bank (south-east side). Observed dilutions ranged from less than 10:1 within 100 m of the outfall to greater than 250:1. Dilutions between 10:1 and 15:1 were observed 450 m from the outfall. The plume makes its way out of the estuary and into Cloudy Bay.

The expected track of the effluent plume informed the microbiological sampling (Mitterwallner and Sneddon 2024). Water quality data consisted of faecal coliforms and enterococci measurements taken at three sites, one site 400 m upstream of the outfall and two sites 300 m and 550 m downstream. The 300 m sample is in the mixing zone of the effluent discharge. Water quality sampling took place as the BSTP discharged during the ebb tide. This is when the concentration of effluent in the estuary would be the highest.

Shellfish data, consisting of faecal coliform measurements in shellfish flesh, were collected at 12 target sites downstream of the discharge, both within and outside the discharge plume. Results are only available for sites where shellfish were present.

Cawthron Institute compared the water quality results against the ANZECC (2000) Guidelines for primary contact and the MoH (1995) reference criteria for shellfish. For two reasons, caution must be used when interpreting these sampling results. Firstly, these guidelines values are intended to be applied to observations taken over a whole bathing season rather than a single point in time, as is the case here. Secondly, these guideline values should not be applied to locations impacted by effluent discharges, as the treatment process alters the relationship between indicators and pathogens (MfE and MoH 2003). Nevertheless, the results of this comparison may be informative.

2.4.1 Primary Contact

The observed water quality data indicated that the level of indicator organisms was in the range of 5–10 MPN/100 mL for enterococci and 4-33 MPN/100 mL for faecal coliforms. It is impossible to assign a grade on the suitability of the estuary for primary contact, as the single day of observations may not be representative of the longer-term water quality. None of the enterococci observations breached the alert level (MfE and MoH 2003), which would require investigation as to the cause to assess the risks to bathers.

The lowest concentration of indicator organisms was upstream of the outfall, demonstrating that the outfall impacts the microbial water quality during discharge. In addition, higher concentrations of FIB were observed at the bed than at the surface for the equivalent site, suggesting that the effluent plume is not fully mixed through the water column, which is consistent with the dye study results.

2.4.2 Shellfish

Two guideline values are relevant to shellfish. Section F of the MfE and MoH (2003) Guidelines set a median faecal coliform level in water for a shellfish gathering season at 14 MPN/100 mL and not more than 10% of the samples exceeding 43 MPN/100 mL. The MoH (1995) have criteria for the level of faecal coliforms in shellfish flesh set at a marginally acceptable level of 230 MPN/100 g and an unacceptable level of 330 MPN/100 g.

Shellfish are benthic organisms living on the estuary bed. Both water quality samples, taken from near the bed downstream of the discharge, had levels greater than 14 MPN/100 g. So, if these samples were representative of water quality over the bathing period, then it would suggest that parts of the estuary were unsuitable for shellfish gathering.

Shellfish flesh was sampled from 12 sites, and the observed level (350 MPN/100 mL) at one site would be considered unacceptable. Shellfish were also sampled at the same sites in 2006, 2016, 2018, and 2004. A total of 34 measurements were made in the combined studies, and 22 samples (65%) would be regarded as safe, having levels below 230 MPN/100g. Five samples (15%) fell into the marginal category, and 7 samples (21%) fell into the unacceptable category.

There was little evidence that the rate of samples being regarded as unacceptable inside the plume was greater than outside, though there was some evidence that the rates were higher in the region from 25 to 200 m from the outfall. No samples at the outfall or at the 300 m site were recorded as having unacceptable faecal coliform levels.

Berthelsen et al. (2015) identified through interviews that shellfish were gathered recreationally within the estuary, and the monitoring data raises questions about its safety, with approximately 35% of shellfish measurements falling into the unacceptable or marginal category.

3 QMRA

This Quantitative Microbial Risk Assessment (QMRA) aims to assist MDC and the local community in understanding the potential health risks associated with the discharge of treated effluent from the BSTP into the Wairau Estuary and, ultimately, Cloudy Bay.

The QMRA assessment only considers risks associated with effluent discharge, and it does not account for background risks or risks associated with other potential sources of microbial contaminants from agriculture (Phiri et al. 2020), wildfowl (Moriarty et al. 2011), stormwater or illicit discharges from boats into the estuary or bay (Landrigan et al. 2020). Therefore, the estimated risk will be the incremental risks from effluent rather than the total risks.

The health risk assessment process comprises multiple steps, including:

1. Select the hazard(s), i.e., the pathogen(s) of concern—exposure to which can give rise to illness.
2. Assess exposures to the pathogens at key sites.
3. Characterise the pathogens' dose response.
4. Risk characterisation.

The "Quantitative" aspect of QMRA relates particularly to item 4—risk characterisation—in which Monte Carlo computer simulation is used. These simulations use repetitive sampling, where possible, to take into account variability and uncertainty in model inputs, so it does not restrict the analysis to using single-point estimates, which may misrepresent the risk. This approach is particularly important given that higher risks may be caused by combinations of inputs toward the extremes of their ranges, the combined effects of which may not be detected when using single values.

3.1 Select the hazard

Human-derived wastewater potentially contains a wide range of pathogenic organisms, which can harm human health if they enter the environment. Assessing the risk from every potential pathogen found in treated effluent is impracticable. Instead, in this analysis, norovirus is chosen as a reference pathogen. Reference pathogen(s) represent the risk of a broader group of pathogens that may be found in the expected exposure pathways (World Health Organization 2016). The exposure pathway is the route to which people outside the boundary of BSTP come into contact with a pathogen from the effluent.

3.1.1 Why norovirus?

Several factors lead to the choice of norovirus as the pathogen. They include evidence from epidemiological studies, results from previous QMRA studies, evidence associated with the relative risks from various exposure routes, availability of data to estimate norovirus risks, and evidence of the resistance of viruses to removal by wastewater treatment.

Viruses, such as norovirus, tend to be more resistant to removal from wastewater via treatment than bacterial pathogens, such as *Campylobacter* (McBride 2017a).

Epidemiological evidence for people exposed to treated effluent from human sources (Sinclair et al. 2009; Graciaa et al. 2018; Landrigan et al. 2020) points towards norovirus causing a significant proportion of gastrointestinal illness.

Noroviruses are a diverse group of single-stranded RNA viruses that currently consist of 10 genogroups (Chhabra et al. 2019). Most of the human norovirus infections are caused by GII and GI genogroups (Robilotti et al. 2015).

Results from QMRAs (Soller et al. 2010; Boehm et al. 2018; Stott and Wood 2022) are consistent with the epidemiological evidence and suggest that the norovirus represents a higher risk than any other individual pathogens considered within these QMRAs. In this sense, risk is in terms of the rate of infection or illness rather than the severity of illness.

The principal mode in which norovirus causes enteric illness is via the faecal-oral route through ingesting water or food, such as shellfish, exposed to treated effluent. Though the focus is on risks associated with ingestion, other modes of infection do occur for viruses. For example, inhalation or contact with water may result in respiratory or eye infections in the case of adenovirus (Sinclair et al. 2009). The inhalation route was not considered here as previous New Zealand QMRAs indicate that for the same activity, individual ingestion risks from exposure to norovirus is greater than inhalation risks from adenovirus (Stott and Wood 2022; Wood and Hudson 2023).

It should be noted that norovirus was not used as the reference pathogen in the 2007 QMRA. The principal reason was the lack of a dose-response model at the time. A dose-response model is a central component of QMRA. So, rotavirus was chosen as a pathogen model, especially for norovirus.

3.2 Assess exposures to the pathogens at key sites.

Assessing exposure requires identifying and quantifying the routes whereby people could be exposed to pathogens from effluent. This includes assessing the source of the pathogen(s), barriers to preventing people from being exposed to pathogens and mechanisms of exposure (World Health Organization 2016). The aim of assessing exposure is to estimate the number of pathogens a person might be exposed to, which is called the dose. In the first part of this section, we provide a qualitative description of the exposure routes before quantifying them.

The health risks of norovirus exposure are assessed based on infection/illness due to exposure to dilute treated effluent. Norovirus is highly infectious and is easily transmitted from a person infected via effluent to another person. However, person-to-person transmission is excluded from this analysis. This aligns with the approach adopted by the National Policy Statement for Freshwater Management (Ministry for the Environment 2024) and Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas (MfE and MoH 2003).

3.2.1 Qualitative description of exposure and site assessment

The BSTP is a potential source of pathogens. Pathogens may be introduced to the environment when the plant discharges treated effluent into the estuary on the ebb tide. People may become exposed to these pathogens in diluted effluent via ingestion of water during activities such as swimming or indirectly through consuming raw or lightly cooked shellfish collected from waters exposed to diluted effluent.

Two key factors influence the dose of pathogens a person may be exposed to: the concentration of pathogens in the water and the amount of water ingested, either directly or indirectly (through food exposed to the water). The more pathogens a person ingests, the higher the risk of becoming infected or ill.

Factors influencing the concentration of pathogens in the environment include the concentration of norovirus in raw wastewater, the removal of norovirus by the treatment process, dilution of treated effluent as it moves through the environment and the removal of pathogens from the environment by various mechanisms, including inactivation. Removal of pathogens from the environment was not considered, as norovirus persists in the environment for some time (Rexin et al. 2024).

The Wairau Estuary and Cloudy Bay are highly dynamic environments. The intention is for the effluent discharge to be quickly washed out of the estuary on the ebb tide. However, the effluent plume may become trapped in the estuary or re-entrained during the flood tide. So, to assess risks, we need to consider multiple sites that effluent may impact under multiple conditions.

A total of 11 sites were identified, drawn from the eight sites in the 2007 Report, plus three more sites chosen for this study. The sites reflected locations within the estuary and the bay that the effluent plume may directly and indirectly impact. Some sites were chosen because people were likely to be exposed to water at these locations, such as the Boat Landing (BL). Other sites, such as Lagoon Entrance (LE), were chosen to provide an understanding of the risk. Site coordinates and locations can be found Table 3-1 and Figure 1-1. The concentration of effluent from the discharge is expected to vary between sites.

Table 3-1: Coordinates of sites assessed for health risks.

Site	Site ID	NZTM_E	NZTM_N	Longitude	Latitude	Distance from outfall (m)
Outfall	OO	1688334	5404612	174.0584	-41.5052	0
Mid Mixing Zone	MM	1688431	5404709	174.0596	-41.5044	140
Lagoon Entrance	LE	1688370	5404427	174.0589	-41.5069	190
Boat Landing	BL	1688180	5404785	174.0566	-41.5037	230
Edge Mixing Zone	EM	1688517	5404842	174.0606	-41.5032	290
Bar Entrance	BE	1688650	5405250	174.0621	-41.4995	710
Gorse Surf Break	GB	1688446	5405633	174.0596	-41.496	1030
Commercial Shellfish Site	CS	1688650	5406100	174.062	-41.4918	1520
Mussel Reef	MR	1689997	5403626	174.0785	-41.5139	1930
Commercial Shellfish 2K from Bar	C2	1687390	5406802	174.0468	-41.4856	2390
Hayshed Shellfish Sanitation Sampling Point	HS	1686569	5407397	174.0369	-41.4804	3300

Knowing the concentration of pathogens in water is only one part of the risk calculation. We also need to consider the volume of water ingested. People can accidentally or coincidentally ingest water during activities such as swimming (primary contact) or boating (secondary contact). We chose to focus on primary contact as there is a tendency to ingest greater volumes and result in greater risk than secondary contact (Dorevitch et al. 2011). People can also ingest water indirectly by consuming shellfish (bivalve molluscan), and shellfish can bioaccumulate pathogens. They are frequently eaten raw, exposing consumers to risk (Pouillot et al. 2022).

3.2.2 Quantifying exposure routes

The goal of quantifying exposure routes is to estimate the norovirus dose an individual may receive during an exposure event. The quantification involves estimating the concentration of norovirus in raw (influent) wastewater, removal of norovirus through treatment systems, dilution of effluent in the environment, and ingesting food and water. The modelling parameters are discussed below, and, with the exception of the dilution parameters, are summarised in Table 3-4 at the end of the section.

The concentration of norovirus in raw wastewater

A short sampling campaign was carried out to enumerate the level of norovirus for both the domestic influent and effluent. The results are presented in Table 3-2. The primary aim of the sampling was to ground truth the assumptions used in the QMRA modelling around the level of pathogens entering and leaving the plant.

Table 3-2: Observed norovirus concentrations estimated by qPCR. Estimates were made using RT-qPCR by the ESR environmental and food virology laboratory. GI and GII refer to genogroup I and II of norovirus.

Date	Influent		Effluent	
	GI (gc/L)	GII (gc/L)	GI (gc/L)	GII (gc/L)
10 June 2024	1.0×10^3	1.3×10^6	1.1×10^3	2.8×10^3
12 June 2024	7.6×10^3	5.9×10^5	2.8×10^3	9.2×10^3
13 June 2024	5.1×10^3	4.3×10^5	1.7×10^3	<50 (Below LoD)
17 June 2024	2.2×10^2	4.8×10^3	5.8×10^3	<50 (Below LoD)
20 June 2024	8.7×10^2	5.8×10^4	1.6×10^3	<100 (below LoQ)

LoD = Limit of Detection. Below LoD implies that the norovirus level was below the level the laboratory could reliably distinguish norovirus from noise.

LoQ = Limit of Quantification. Below LoQ implies norovirus was detected but the level could not be determined with an acceptable level of precision and accuracy.

gc/L = genome copies per litre.

The observed level of norovirus for the influent ranged from 2.2×10^2–7.6×10^3 gc/L for GI and 4.8×10^3–1.3×10^6 gc/L for GII. In the effluent 1.1×10^3–5.8×10^3 gc/L for GI and <50–9.2×10^3 gc/L for GII. It takes over a month for the wastewater to move through the plant due to long retention times in the pond system. Thus, it is not possible to relate the influent and effluent samples, which in this case were taken over an 11-day period. This explains why the minimum concentration of the GI strain in the effluent (leaving the plant) was higher than the minimum influent (raw wastewater) concentration. Therefore, it is not possible to come up with a robust measure of norovirus removal from this data.

Nothing suggests that the measured levels of norovirus in BSTP were atypical for a New Zealand wastewater treatment plant. Examples of measurements from other plants are presented in Table 3-3. In New Zealand, QMRA modelling studies it is commonly assumed to be described by the hockey stick distribution with a breakpoint at the 95th percentile (McBride 2005). The level of norovirus in effluent is in the range of 1.0×10^3 to 1.0×10^7 gc/L with a median value of 1.0×10^5 gc/L as used by Stott et al. (2023), Wood and Hudson (2023), Wood and Stott (2023) and Wood and Stott (2024). In their work, Cressey (2021) and Armstrong (2022) use a slightly lower level, but it makes no practical difference to the resulting risk estimates. The risk estimates are sensitive to the upper level of pathogen concentrations, not the lowest levels.

Table 3-3: Observed norovirus concentrations estimated by qPCR from studies of New Zealand Sewage Treatment Plants.

Date	Influent		Effluent		Reference
	GI (gc/L)	GII (gc/L)	GI (gc/L)	GII (gc/L)	
New Plymouth	$1.7x10^4 – 2.8x10^5$	$4.7x10^5 – 2.8x10^7$	<100 – 460	$1.3x10^4 – 3\ x10^4$	(McBride 2012)
Napier	$2.7x10^2 – 3.2x10^5$	$1.8x10^5 – 3.2x10^5$	NA	NA	(McBride 2012)
Hawera	$2.7x10^2 – 2.9x10^3$	$7.5x10^4 – 8.2x10^5$	<100 – $4.9\ x10^3$	$2.5x10^3 – 4.2x10^4$	(McBride 2012)
Assorted NZ WWTP	$1.3x10^2 – 4.4x10^4$	$1.5\ x\ 10^2 – 2.9\ x\ 10^5$	$1.5x10^2 – 1.2x10^5$	$7.6\ x\ 10^2$-$2.9x10^5$	(Hewitt et al. 2011)

Hamadieh et al. (2021) reported maximum concentrations of ~1 × $10^{8.5}$ genome copies/L which are greater than those used in New Zealand QMRAs. Eftim et al. (2017) noted in their systematic literature review that the concentration of norovirus was lower in New Zealand than in Europe or Africa. Given the observation that New Zealand studies suggest lower norovirus concentrations than elsewhere in the world, it is reasonable to use the standard factors used in previous New Zealand QMRA's.

Note the measurements quoted above are made with molecular techniques and do not distinguish between viruses that can cause infections and those which do not.

Removal of norovirus by the treatment process

One of the principal roles of a wastewater treatment plant is to remove pathogenic microorganisms before the effluent discharges to the environment. Removal of microorganisms are simulated in the QMRA by assuming 10-fold, 100-fold, 1,000-fold and 10,000-fold, 100,000-fold and 1,000,000-fold reductions of microorganisms by treatment. These levels of treatments are referred to as 1, 2, 3, 4, 5 and 6 log reduction values (LRV).

The norovirus measurement provides insufficient data about the efficacy of virus removal in BSTP. The 2007 QMRA assumed BSTP had an LRV of 3.7 and 2.4 for summer and winter respectively in the 2007 QMRA.

Removal depends on the design of the system and the virus of interest, and there can be considerable differences between systems, even using the same design (World Health Organization 2016).

The BSTP has two separate treatment systems, a domestic and an industrial wastewater system. Domestic wastewater is treated using a fine screen, facultative and maturation ponds, while industrial wastewater undergoes fine screening, aeration, and facultative treatment. These two streams are combined in Pond 6. The resulting treated effluent then passes through the wetland before being discharged from Pond 14 to the Wairau Estuary via two outfall pipes (Beca Ltd 2023). The hydraulic retention times (HRT) for the domestic system, pond 6, and the wetland are approximately 22, 13, and 6 days, respectively (Graeme Jenner, personal communication July 3, 2024).

Table A5.5 in World Health Organization (2016) provides indicative levels of indicator and pathogen removals for various systems, but this is based on data collected before 2006, and as Verbyla and Mihelcic (2015) noted, our conclusions about virus removal have evolved over time.

We carried out a brief literature search to identify review papers for virus removal on pond and wetland systems. There are several challenges in studying virus removal, including the highly variable nature of virus-influent concentrations and the use of molecular techniques that cannot distinguish which viruses are viable and capable of infecting a person and those that are not (Rexin et al. 2024). This is particularly problematic when the treatment process involves the inactivation of viruses rather than their physical removal (Verbyla et al. 2023). So, some workers used phages as a surrogate for enteric virus – enteric viruses are capable of infecting humans. Phages, or bacteriophages, are viruses that infect and replicate within bacteria, and are easier to detect. Verbyla and Mihelcic (2015) reviewed the results from 71 pond systems, including aeration and facultative treatment, but not constructed wetlands. Focusing on their graphical results for ponds with HRT greater than 10 days, the reported LRV values for viruses and phage's lie in the range of 0.4 to 5 with a median value of about 1.5[1]. They estimated that, on average, ponds had a one log reduction after 14.5–20.9 days.

Amarasiri et al. (2017) reported an average LRV for ponds of 2.26 with a 95% confidence interval of 1.32–3.21 from three studies. Based on five studies, they also reported an average LRV for constructed wetlands of 0.88 with a confidence interval of 0.63–1.13. Amarasiri et al. (2017) work was based on a meta-analysis of phage data.

Rachmadi et al. (2016) reported 1-3 LRV of enteric virus in a study of two wetlands. A review of wetlands by de Campos and Soto (2024) noted LRV in the range of 0.5–2.3 for bacterial indicators. Bacteria are generally less resistant to treatment than viruses. The estimated bacterial LRV for total coliforms in BSTP is approximately 1.4, as discussed in section 2.2.3 above. So it is unlikely the virus LRV of the wetland is greater than 1.4.

If this average from Verbyla and Mihelcic (2015) were applicable to the BSTP ponds, it suggests the LRV could be in the range of 1.6 and 2.4 for the domestic system in combination with pond 6. This is in line with the confidence interval from Amarasiri et al. (2017).

There appear to be fewer studies on wetlands than ponds, and the range of LRVs is large, especially for viruses. The precautionary approach would be to assume the LRV would be less than bacterial indicators, so it might be in the range of 0.5 to 1.4 based on the work of de Campos and Soto (2024) using the BSTP FIB monitoring data reported above for the upper limit.

The current LRV for the BSTP is unknown. Assuming, as per the literature, that the LRV range in the domestic ponds and pond 6 is 1.6-2.4 and the LRV range in the wetlands is 0.5-1.4, we have a "worst-case" LRV for the ponds of 1.6 + 0.5 = 2.1. However, it may be that the LRV is higher, with a possible upper limit of 2.4 + 1.4 = 3.8. These values are similar to the assumptions used in the 2007 QMRA.

[1] Estimated by visual inspection of Fig 4 of Verbyla and Mihelcic (2015).

Dilution

Treated effluent is discharged into the estuary twice a day on the ebb tide from the outfall pipe. The discharge starts at least an hour after high tide. According to Sneddon (2024) the approach is intended to minimise the impact of the effluent to the estuary by rapidly conveying the discharge from the “diffuser-less” outfall into Cloudy Bay where it becomes diluted.

Results of a dye study show an elongated plume of effluent originating at the outfall heading toward the bar and keeping to the right bank of the estuary. Dilutions were in the order of 20:1 along the centreline of the plume, 300 m from the outfalls (Sneddon 2024).

Dye studies are very useful in understanding dilution, but the results only represent a point in time and are limited in geographical extent. So, they do not provide insight into effluent dilution under a full range of weather, river flow and tidal conditions.

DHI conducted a hydrodynamic modelling exercise to quantify the mixing of treated effluent within the estuary and along the coast. The modelling exercise simulated the dilution of the discharge over a one-year period. This ensured a range of river flow, weather and tidal conditions were considered. It was assumed that the bar at the entrance to the estuary was stable and did not change.

Dilutions at the outfall were not calculated, but it was assumed that during discharge, there was no mixing at the point of discharge, so the dilution would have a value of one if it was plotted on Figure 3-1.

Cumulative distribution curves of DHI dilution estimates are shown in Figure 3-1. Higher figures mean greater dilution. The highest risks to health are often associated with the lowest dilutions, so the graphs have been ordered in terms of the 5^{th} percentile of dilution. As might be expected based on the intended approach to move effluent rapidly into the bay, the lowest dilutions are in the effluent plume, such as Mid Mixing zone (MM), Edge of the Mixing zone (EM), Bar Entrance (BE) and Gorse Bay (GB). Dilutions can be quite high at other sites in the estuary, such as at the Boat Landing (BL).

The dilution profiles for many of the sites show a smooth s-shaped curves except for sites MM, EM, BE and GB which appear to be more directly impacted by the plume.

Risks to swimmers relate primarily to surface water, whilst shellfish risks relate the concentration at the seabed. Dilutions were estimated at the bed and the surface. For sites within the estuary (MM, EM, LE, BL) the dilutions were higher on the surface than on the bed. For sites in the bay (GB, HS, MR, CS and C2), dilutions were higher on the bed than at the surface. At the bar entrance (BE), overall dilutions were similar at the surface and on the bed.

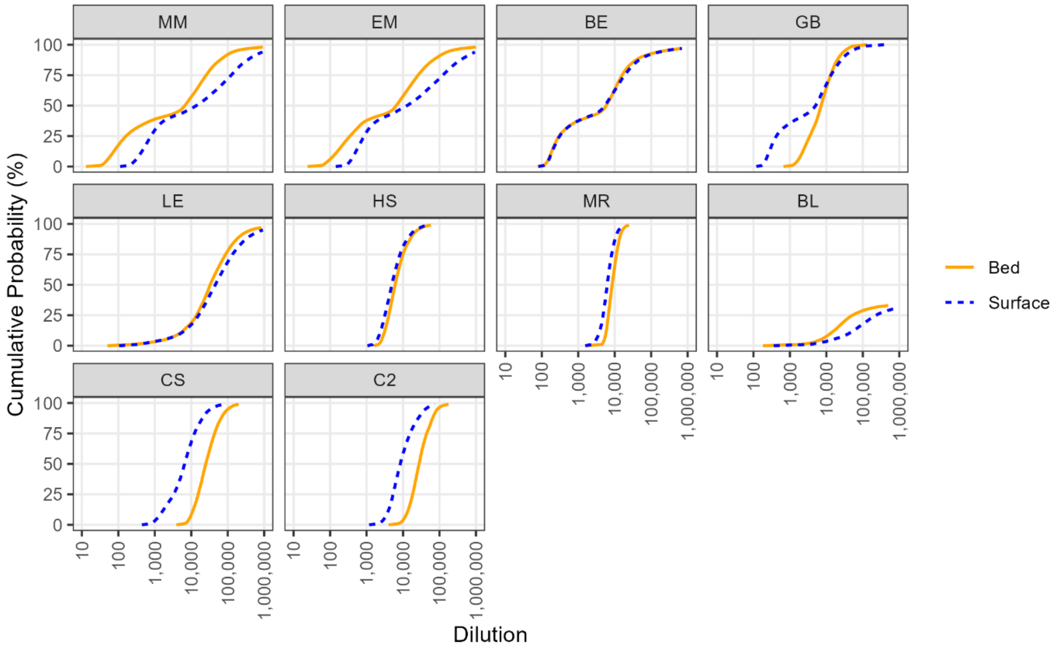

Figure 3-1: Cumulative distribution curves for dilution at 10 sites. The sites are in order from lowest dilution (top left) to highest dilution (bottom right). Note the logarithmic scale for the dilution axis and values over 1,000,000 have not been plotted.

Delving into the timing of discharge and dilution, it is noted that dilution is strongly related to the timing of the discharge and the tides. Focusing on the Bar Entrance (BE) it can be seen that between two and seven hours after high tide, dilutions were in the range of 1:100 and 1:1000. Outside these times, the dilution was higher, approximately 1:10,000. Given the impact of timing, it was decided to estimate the average risk at any time of the day and in addition, estimate swimming risks between one and six hours after high tide.

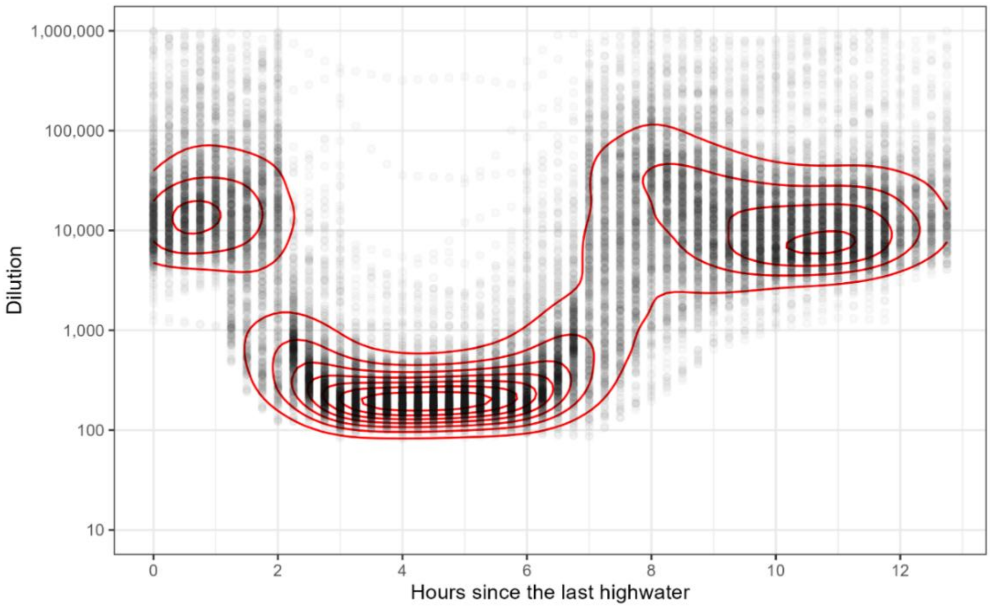

Figure 3-2: Estimated dilution in surface water at the Bar Entrance (BE) with respect to time since high tide. Each point is the dilution every 15 minutes over a year. The red lines are contours illustrating the density of data points.

Ingestion of food and water

Viruses in water can be ingested directly through water consumption or indirectly through the ingestion of animals or plants that have been exposed to viruses in water. In the case of direct ingestion, the question is how much water people ingest, and for foods, the question is how much food is consumed and what the virus content of the food is.

Direct ingestion of water

Water-related activities can result in the unintentional ingestion of water. Swimming, referred to as primary contact, tends to result in greater volumes of water being ingested than secondary contact activities such as boating or fishing, etc. (Dorevitch et al. 2011). Evidence suggests that children ingest water at a higher rate and spend more time in the water swimming than adults (Dufour et al. 2017). So, children swimming in water were chosen as a susceptible part of the population.

New Zealand specific data is not available. However, the World Health Organisation's (2016) guidance on QMRAs quotes a range of figures for the volume of water accidentally ingested during swimming, ranging from 20–100 mL per event. The World Health Organization (2021) Guidelines on recreational water quality quote higher "per event" figures of 140–250 mL for children.

The volume of water accidentally ingested will likely vary from person to person. Schets et al. (2011) published information on the duration of swimming, with average durations ranging from 8-240 minutes and 12–270 minutes for children in seawater and freshwater, respectively. Dufour et al. (2017) estimated the ingestion rate to vary from 0–280 mL/h with an arithmetic mean of 32 mL/h.

This work assumed a log-normal distribution with minimum, mean, standard deviation and maximum ingestion rates of 5, 53, 75 and 250 mL/h. The duration of events was modelled with a PERT distribution with a minimum value of 12 minutes, mode of 1 hour and maximum of 4 hours. These figures have been used in previous QMRAs (Stott and Wood 2022; Wood and Stott 2023) and result in a mean ingestion volume of approximately 64 mL per event with 5th and 95th percentile ranging from 6.6 to 216 mL per event. The mean values are in the range given by the World Health Organization (2016) guidance on QMRAs, and though the parameters are different from those used by Cressey (2021), the overall results are expected to be similar.

Shellfish

Shellfish can bioaccumulate pathogens in their flesh, so consuming 1 g of shellfish is equivalent to ingesting more than 1 mL of water. Burkhardt and Calci (2000) estimated Bioaccumulation Factors (BAF) for shellfish and noted that BAF varied by season. Following the precautionary approach, we used the maximum BAF value (Burkhardt and Calci 2000). McBride (2007) estimates of shellfish consumption using survey data from Russell et al. (1999) along with BAF and the concentration of pathogens in the water, it is possible to estimate the pathogen dose associated with the consumption of raw or lightly cooked shellfish. McBride (2007) estimates that the mean meal size of 100 g is similar to the average shellfish meal size estimated by Guy et al. (2021), which is 106 g.

Dose-response

The risks from norovirus depend on the dose individuals receive, i.e., the number of viruses ingested. Teunis et al. (2008) developed a dose-response model for norovirus, which suggests that higher doses lead to a higher chance of infection. Information from Teunis et al. (2008) was used to estimate what proportion of the population was susceptible to norovirus and what proportion of those infected became ill.

Noroviruses are a diverse group of single-stranded RNA viruses comprising 10 genogroups (Chhabra et al. 2019). Teunis et al. (2008) only report dose-response models for norovirus genogroup 1 (GI). In contrast, norovirus genogroup 2 (GII) concentrations in New Zealand studies are typically greater in raw sewage than those of GI, as illustrated in Table 3-2 and Table 3-4. Due to the lack of a specific dose-response model for genogroup 2 (GII)[2] we assume that GI and GII have the same dose-response relationship.

Since Teunis et al. (2008) developed the dose-response model, analytical techniques have also improved. We therefore include a dose-response method harmonisation factor (MHF) to account for these differences (McBride et al. 2013).

Norovirus may exist in aggregated (clumped) and disaggregated forms, and Deere and Ryan (2022) recommend that norovirus QMRAs be modelled in both aggregated and disaggregated forms. However, previous QMRA modelling (e.g., McBride 2017b), indicated that disaggregated norovirus creates a consistently greater illness risk than the aggregated form. In response, we have limited our consideration and discussion to illness risks arising from the disaggregated norovirus form (i.e., we have taken the more conservative approach) – this is consistent with previous QMRA practice e.g., McBride (2017a).

[2] A model has recently been proposed for NoV GII by Teunis et al. (2020) but the application of this dose-response model is less certain.

Table 3-4: Summary of QMRA modelling input parameters.

Component	Statistic/parameter		Distributions/comments
Influent virus concentration			Bounded "hockey stick" distribution (McBride 2005), strongly right-skewed.
Influent norovirus concentration, genome copies per litre (gc/L)	Minimum	$1x10^3$	Typical range found for New Zealand cities (e.g., Napier, New Plymouth— (McBride 2011; McBride 2012; McBride 2017)).
	Median	$1x10^5$	
	Maximum	$1x10^7$	
Hockey stick, norovirus, Xp	Unitless	0.95	
Treatment efficacy			
Wastewater treatment efficacy, Log10 virus reduction (LRV)	Unitless	1 - 6	LRVs represent a range of treatment efficacies
Exposure parameters - swimming			
Duration of swim (hours)	Minimum	0.2	Distribution for a child after Schets et al. (2011) based on Program Evaluation and Review Technique (PERT)distribution.
	Mode	1	
	Maximum	4	
Swimmers water ingestion rate (mL per hour)	Minimum	5	Truncated lognormal distribution (ESR 2016), (Table 19); (Dufour et al. 2017) for children (<16 yr). The minimum value was set at 5 mL/h, an ingestion rate equivalent to one tablespoon of seawater per hour. This estimation of the minimum value took into account information from ESR (2021), which evaluated the raw data from Dufour et al. (2017) and the observation that ingestion rates appear to be greater than inhalation rates, so the minimum value was set to be greater than the minimum inhalation rate of Dorevitch et al. (2011).
	Mean	53	
	Std. Dev	75	
	Maximum	250	
Exposure parameters - shellfish			
Shellfish meal size (g)	α	2.2046	A log logistic distribution was used, truncated below at 5 g and above at 800 g, from bivalve mollusc consumption data from Parnell et al. (2001) and McBride (2012).
	β	75.072	
	ϒ	–0.903	
Bioaccumulation factor, ratio	Mean	49.9	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.
	Std. Dev.	20.93	

Component	Statistic/parameter		Distributions/comments
Dose Response			
Probability infection norovirus GI5 per exposure event (disaggregated)	α β	0.04 0.055	Beta-binomial (for individual doses, i) is described by two parameters α and β (Teunis et al. 2008), Table III, 8fII1+8fIIb, no aggregation. ID_{50} infection = 26
Fraction of secretor-positive individuals (susceptible to norovirus infection)	Unitless	0.74	Pr (ill|Inf) NoV: estimated from Soller et al. (2008)
The conditional probability of illness given infection NoV (norovirus)	Unitless	0.68	Pr (Ill|Inf) NoV: estimated from Soller et al. (2008)
Method Harmonization Factor (MHF) for norovirus	Unitless	18.5	The dose-response equation and current monitoring methods use RT-qPCR methodology but on different genetic target sequences with differences in critical threshold standard curves (McBride et al. 2013). Current PCR methods more effectively detect virions, norovirus concentration data divided by harmonisation factor.

3.3 Risk characterisation

Risk characterisation brings together information on exposure and passes it through the dose-response model to estimate the risk of infection and illness per exposure event: a swim or a feed of raw or lightly cooked shellfish.

Monte Carlo statistical modelling is used as it allows for a range of likely conditions to be included in health risk estimates, including relatively infrequent but highly influential elevated virus concentrations (McBride 2005; Haas et al. 2014). Therefore, we can consider a range of dilutions from the modelling, not simply the average dilution. A "Monte Carlo" approach consists of repeated sampling of various parameters, such as the concentration of pathogens in shellfish and meal size. The parameters of variables used within the QMRA modelling are shown in Table 3-4. The Monte Carlo simulations were conducted in Excel using the @Risk add-in (LUMIVERO 2024).

Health risks are estimated following exposure of a hypothetical population (a group of 100 "individuals") to an individual "dose" on any particular day. The total number of individuals becoming ill from 100 people exposed is determined as the risk outcome for that iteration. This procedure is repeated for a total of 10,000 iterations drawn at random from the distributions of key input variables. For instance, the consumption of one million shellfish meals is simulated to capture the variability and uncertainty in the model's inputs.

The risks were calculated for two different exposure routes: one for people swimming and another for those consuming shellfish. These risks were estimated by assuming a person was exposed to water or food at a random point in time (any time of day). A second set of estimates was made for swimmers who were exposed to water between one and six hours after high tide to understand the increased risks during and immediately after effluent discharge. While swimming risks reflect instantaneous water quality, shellfish risks do not, as they accumulate pathogens over time. Therefore, the post-discharge risks for shellfish were not estimated separately. The exception was at the outfall, where swimming and shellfish risks were only considered during discharge, the worst possible case.

3.4 Results

The results of the QMRA are presented in tabular and graphical forms. They are compared against Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas (MfE and MoH 2003), specifically against Table H1 which is for marine waters. The Guidelines provide four Microbiological Assessment Categories (MAC) and are presented in Table 2-2.

3.4.1 Swimming risks

The mean Individual Illness Risk (IIR)% is a function of the level of virus removal (LRV), risk assessment location, and the risk assessment period. The results are presented in Figure 3-3 and Table 3-5.

- Higher levels of virus removal result in lower risk for all sites. With the exception of the outfall (end pipe site), the IIR's were less than 1% when LRV was 2 or higher.
- Risks vary by site. The highest risk is to a person who ingested undiluted effluent as it exits the outfall. The next highest risks were at the Bar Entrance (BE), and Gorse Surf Break (GB), both higher than the next highest estimated risks, which were in the estuary, the Mid Mixing (MM) and Edge Mixing (EM) zone. The lowest risk was at another estuary site, the Boat Landing (BL).
- Risks were higher than the average risk during discharge for several sites. For example, the IIR at discharge was about twice that of the average risk as the Bar Entrance (BE), though the relative difference varied by site. In some cases, including Hayshed Shellfish Sanitation Sampling Point (HS), Commercial Shellfish Site (CS), and Boat Landing (BL), the risks during discharge were lower than average.

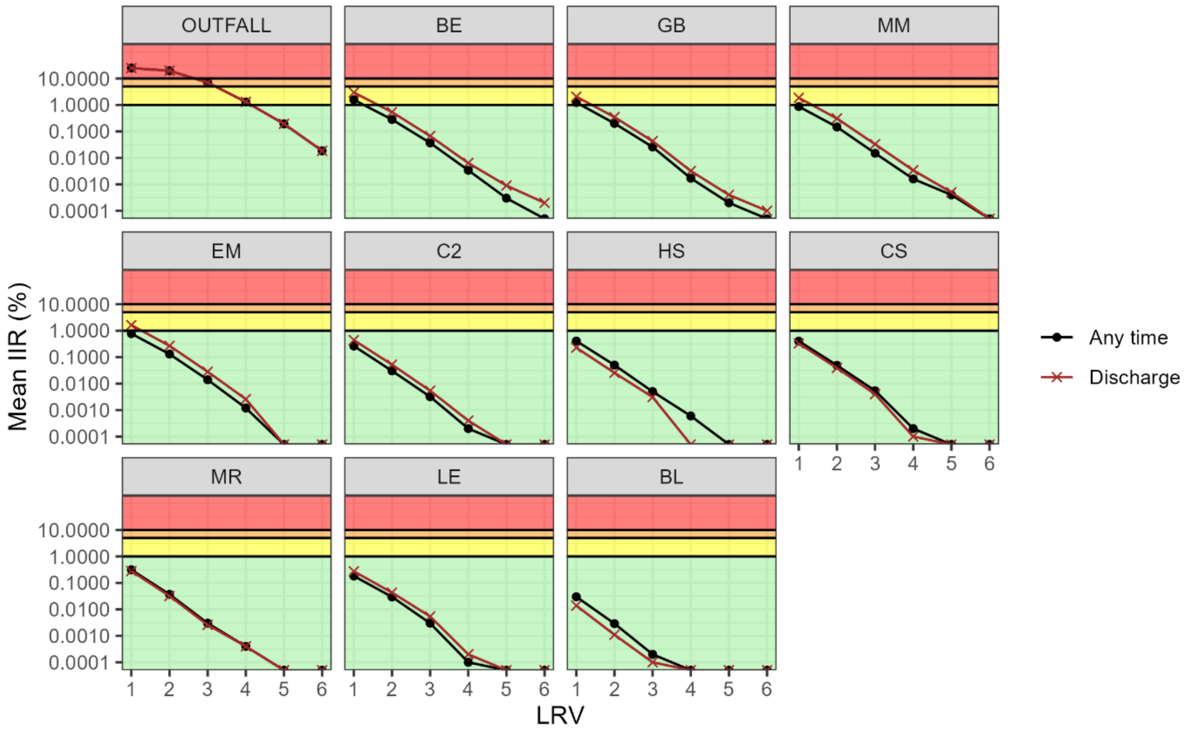

Figure 3-3: Mean Individual Illness Risks (IIR) for primary contact (swimming) in waters exposed to treated effluent. Given six levels of water treatment at any time of day or during and immediately after discharge. Note y-axis has log_{10} scale. The green, yellow, orange, red, represent the Microbiological Assessment Categories in the "Microbiological Water Quality Guidelines" (MfE/MoH 2003).

Table 3-5: Estimated IIR values at each site for swimming risks for a range of LRV. The green, yellow, orange, red, represent the Microbiological Assessment Categories in the "Microbiological Water Quality Guidelines" (MfE/MoH 2003).

		Log Reduction Value (LRV)					
Site ID	Scenario	1	2	3	4	5	6
OO	Discharge	24.8662	19.6447	7.0264	1.3042	0.1929	0.0185
BE	Any time	1.5653	0.2823	0.0368	0.0034	0.0003	<0.0001
BE	Discharge	2.9585	0.5368	0.0668	0.0064	0.0009	0.0002
GB	Any time	1.2420	0.2009	0.0258	0.0017	0.0002	<0.0001
GB	Discharge	2.0670	0.3478	0.0428	0.0032	0.0004	0.0001
MM	Any time	0.8718	0.1471	0.0150	0.0016	0.0004	<0.0001
MM	Discharge	1.8606	0.3085	0.0328	0.0034	0.0005	<0.0001
EM	Any time	0.7763	0.1306	0.0140	0.0012	<0.0001	<0.0001
EM	Discharge	1.6273	0.2683	0.0282	0.0026	<0.0001	<0.0001
C2	Any time	0.2644	0.0304	0.0032	0.0002	<0.0001	<0.0001
C2	Discharge	0.4415	0.0523	0.0054	0.0004	<0.0001	<0.0001
HS	Any time	0.4089	0.0502	0.0050	0.0006	<0.0001	<0.0001
HS	Discharge	0.2291	0.0256	0.0031	<0.0001	<0.0001	<0.0001
CS	Any time	0.4036	0.0492	0.0054	0.0002	<0.0001	<0.0001
CS	Discharge	0.3307	0.0392	0.0040	0.0001	<0.0001	<0.0001
MR	Any time	0.3131	0.0368	0.0030	0.0004	<0.0001	<0.0001
MR	Discharge	0.2802	0.0321	0.0026	0.0004	<0.0001	<0.0001
LE	Any time	0.1827	0.0296	0.0030	0.0001	<0.0001	<0.0001
LE	Discharge	0.2740	0.0431	0.0054	0.0002	<0.0001	<0.0001
BL	Any time	0.0300	0.0029	0.0002	<0.0001	<0.0001	<0.0001
BL	Discharge	0.0139	0.0011	0.0001	<0.0001	<0.0001	<0.0001

3.4.2 Shellfish risks

The calculated mean Individual Illness Risk (IIR)% for shellfish consumption is a function of the level of virus removal (LRV) and risk assessment location. The results are presented in Figure 3-4 and Table 3-6.

- Higher levels of virus removal result in lower risk for all sites. With the exception of the Outfall (end pipe site), the IIR's were less than 1% when LRV was 4 or higher. However, sites beyond the bar LRV of 3 or higher are sufficient to keep IIR below 1%.
- Risks vary by site. The highest risk is to a person who ate raw or lightly cooked shellfish collected from the outfall where it is exposed to undiluted effluent. The next highest risks were sites impacted by the effluent plume, the Mid Mixing Zone (MM), followed by the Bar Entrance (BE) and the Edge Mixing (EM) zone. The lowest risk was for shellfish collected at the Boat Launching Wharf (BL).

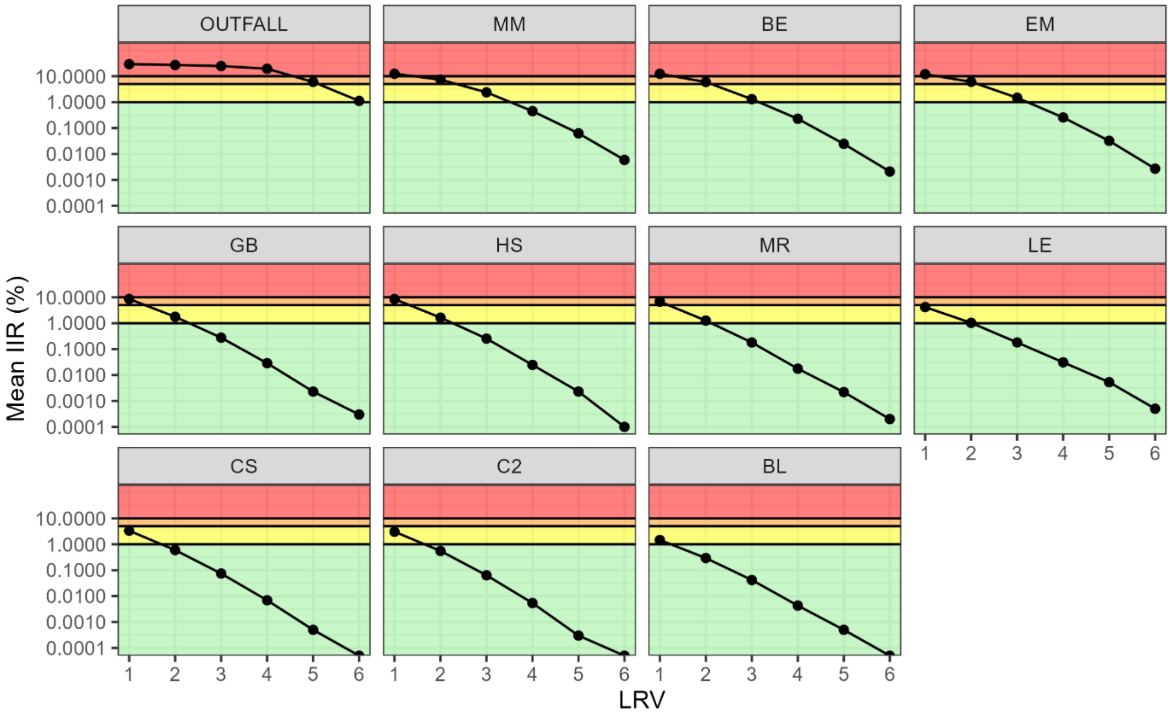

Figure 3-4: Mean Individual Illness Risks (IIR) for consumers of shellfish gathered in waters exposed to treated effluent. Given six levels of water treatment at any time of day or during and immediately after discharge Note y-axis has log_{10} scale. The green, yellow, orange, red, represent the Microbiological Assessment Categories in the "Microbiological Water Quality Guidelines" (MfE/MoH 2003).

Table 3-6: Estimated IIR values at each site for shellfish risks for a range of LRV. The green, yellow, orange, red, represent the Microbiological Assessment Categories in the "Microbiological Water Quality Guidelines" (MfE/MoH 2003).

	Log Reduction Value (LRV)					
Site ID	1	2	3	4	5	6
OO	29.1611	27.102	24.7279	19.4208	6.0088	1.1139
MM	12.4172	7.3671	2.3899	0.4483	0.0626	0.0059
BE	12.3722	5.9705	1.3039	0.2273	0.0245	0.0021
EM	11.9702	6.0695	1.4753	0.2562	0.0321	0.0027
GB	8.6017	1.7721	0.2760	0.0287	0.0023	0.0003
HS	8.5311	1.6465	0.2551	0.0246	0.0023	0.0001
MR	6.8151	1.2623	0.1802	0.0176	0.0022	0.0002
LE	4.1817	1.0319	0.1807	0.0310	0.0053	0.0005
CS	3.3429	0.5974	0.0744	0.0069	0.0005	<0.0001
C2	3.0560	0.5532	0.0637	0.0054	0.0003	<0.0001
BL	1.4661	0.2938	0.0419	0.0043	0.0005	<0.0001

4 Discussion and conclusions

In this section, we discuss the QMRA results and put them in the context of the observed FIB levels and the 2007 QMRA, including necessary assumptions in our calculations.

4.1 Assumptions

Public health requires us to take a precautionary approach when there is uncertainty (Goldstein 2001). As a result of this, several assumptions have been made when estimating risk. These assumptions and their impact on estimated risks include:

- The norovirus is in a disaggregated state (if the norovirus were in an aggregated state estimated risks would be lower).
- All viruses were viable and have the potential to infect (risks would be less if a proportion of the virus load were non-viable).
- Virus inactivation has not been modelled (declining virus infectivity with time would reduce risks, particularly for sites not directly impinged by the effluent plume).
- We do not know the health status of the Blenheim community (if we could reliably know when there was an outbreak of gastroenteritis in the community, risk could be managed by dissuading the public from swimming and shellfish gathering after an outbreak).
- Conservative assumptions, which are consistent with current standard QMRA practice in NZ have been made about the volume of water ingested, quantity of shellfish eaten and the bioaccumulation of pathogens in shellfish (if these assumptions were too conservative, the actual risks may be less).
- The worst case of bioaccumulation in shellfish has been assumed (bioaccumulation is thought to vary seasonally, and if this had been taken into account shellfish risks may be less).

There is no bottom line for health risks in the marine environment for recreational contact with water or shellfish consumption. So, in this work, we have used the A/B boundary of the Microbial assessment Category from the marine guidelines (MfE and MoH 2003) which equates to a 1% illness risk as a comparator. Table 22 of the NPS-FM (Ministry for the Environment 2024) has a bottom line set in terms of percentile standards, with illness risk set at less than 5% for 95% of the time. Unfortunately, this does not align well with QMRA results. However, mapping Table 22 with Table 9 of the NPS-FM suggests the bottom line equates to 1% Individual Infections Risk (IInfR) or 0.7% Individual Illness Risk (IIR), taking into account what proportion of people become ill after norovirus infection from Table 3-4.

4.1.1 QMRA results

The results are presented in terms of the Individual Infection Risk (IIR) from norovirus and compared with the Microbiological Assessment Categories from the Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas (MfE and MoH 2003). The IIR from BSTP is in addition to risks from other sources.

The QMRA results show that risks vary spatially, with the degree of treatment (LRV) and with respect to the type of health risk, either primary contact recreation (e.g., swimming) or consumption of raw shellfish harvested following exposure to treated diluted effluent discharged to the estuary.

Keeping all other things the same and ignoring risks at the outfall, the risks from shellfish consumption are greater than primary contact recreation (swimming). This reflects that shellfish can bioaccumulate pathogens from the water they are in, and the mass of shellfish eaten during a feed can result in a greater norovirus dose than if a child (the highest risk group) accidentally ingested the same water during a swim.

Again, keeping everything the same and ignoring risks at the outfall, risks vary spatially. The spatial distribution of risk reflects the effluent's dispersion during discharge on the ebb tide. As a result, sites that come into direct contact with the effluent plume within the mixing zones (MM and EN) in the estuary, the bar entrance, and sites in the bay close to the bar entrance have the highest risks. The dynamics of the effluent plume, staying close to the bed in the estuary, means that swimming risks are greatest at the bar entrance and sites in the bay when the plume comes to the surface. Shellfish risks are highest in the estuary at the Mid Mixing zone (MM). The spatial variability is illustrated by the fact that the Boat Landing (BL), which is only 230 m from the outfall, has the lowest overall risk, whilst the Bar Entrance (BE), 710 m from the outfall, has one of the highest risks.

The level of treatment modifies the risks for all sites. Higher levels of treatment will lead to lower risks. We do not have information about the efficacy of treatment, and competent engineers with knowledge of BSTP and the efficacy of virus removal will be in the best position to estimate it. We have provided some references from the literature on the efficacy of the ponds and wetlands worldwide.

If a 1% IIR target was chosen, LRV = 2 would be sufficient to meet that target for swimming, excluding the outfall site. To keep risks below 1% for shellfish for all sites, excluding the outfall, LRV = 4 would be required. This would fall to LRV = 3 if only sites in the bay were considered.

4.1.2 Context report and QMRA results

Faecal indicator bacteria (FIB) monitoring of water quality and shellfish provides limited information on the risk to public health in the Wairau Estuary. The QMRA results provide additional information on the likely risk under various treatment levels.

The FIB and QMRA results indicate that the highest risk would occur if a person or shellfish is exposed to undiluted treated effluent at the end of the outfall pipe during discharge.

The FIB monitoring results indicate that approximately 35% of the shellfish samples taken from the estuary during monitoring events fall into the marginally acceptable and unacceptable levels of quality concerning human health risks. The limited water quality data suggest estuary water within the effluent plume may not meet human health guidelines for shellfish gathering, however it is worth noting that the water quality data at the monitored sites did not breach the guidelines for marine recreational water quality. In the presence of effluent discharges, complying with guidelines values is not a guarantee of safety. However, it would appear that there are risks from consuming shellfish gathered in the estuary, though the risks associated with swimming are uncertain.

The QMRA modelling illustrates that risks associated with shellfish consumption are higher than risks associated with primary recreation, which appear to align with the FIB monitoring results.

Given the low estimated risks from the two rivers, the Wairau and Ōpaoa, the majority of the human health risks in the effluent plume within the estuary during discharge appear to come from the effluent.

4.1.3 Comparing a contrasting result with the 2007 QMRA study

A 2007 study (McBride 2007) preceded this QMRA. Both studies had similar objectives in assessing recreational and shellfish risk from BSTP. However, improvements in technical capabilities and scientific knowledge have led to differences between the two studies' approaches, though there are many similarities. The outcome of these differences is that the results of these two studies are not directly comparable. A limited comparison is given below to help stakeholders understand why there are differences and similarities between the two studies.

The two studies have similarities in that they estimate the same types of risks: recreational contact with water and shellfish consumption risks. In addition, the risks were assessed at the same eight locations (though the current study has three additional risk assessment locations). The differences in approach between the two studies include:

- Reference pathogen—The 2007 study used rotavirus. At the time, there was no dose-response model for norovirus, which is used in the current study.
- Health outcomes—The 2007 study used infection as the health outcome, whereas the current work uses illness. The guideline values for marine recreation are based on illness. Illness risks are lower than infection risks. In the case of norovirus, illness risks are approximately 68% of infection risks.
- Pathogen concentrations – The 2007 study separately modelled normal and extreme pathogen concentrations associated with an outbreak, as it was assumed we would know if there was an outbreak of gastroenteritis in Blenheim. However, we generally do not know the population's health status, and norovirus outbreaks do go undetected or underreported (CDC 2024). In the current study, we modelled normal and occasional extreme concentrations together.
- Scenarios—The 2007 study considered a set of scenarios lasting two days, each involving fixed weather, river flow, bar configurations, variety of volumes and timing of discharge plus normal vs. extreme virus concentrations. The current study modelled a one-year period that included a range of weather, river flow, and virus concentrations under a fixed bar configuration. The previous study enabled the identification of high-risk events, but the current study provides a better estimate of the average risks.
- Hydrodynamic modelling – The 2007 study made use of 2D hydrodynamic modelling (Andersen and Tuckey 2007). The current work used 3D modelling, which can better estimate the variability of effluent dispersion with depth.
- LRV—The 2007 study assumed that the BSTP could remove 99.98% and 99.6% of viruses in the summer and winter, respectively, equating to an LRV of 3.7 and 2.4. The current study modelled a range of LRVs from 1 to 6 in increments of one, but made no assumptions on LRV for the BSTP. The reason is uncertainty about the LRV the plant can achieve and questions about the level of treatment required.

Both studies indicated average risks were higher for shellfish consumption than for recreational contact.

Both studies show the highest estimated recreational risks in parts of the estuary, the bar entrance, and sites in Cloudy Bay near the bar entrance. For the 2007 QMRA[3], the highest recreational risks occurred at the Edge Mixing zone (EM), Bar Entrance (BE), Commercial Shellfish site (CS) and Gorse Surf Break (GB). For the current QMRA, for the sites which were included in the 2007 study, the four sites with the highest IIR were Bar Entrance (BE), Gorse Surf Break (GB), Edge Mixing zone (EM) and Commercial Shellfish 2 Km from Bar (C2). Both studies showed that the risk within the estuary were highly variable, with risks at the Boat Launching Wharf (BL) being lower than at Edge Mixing zone (EM) or Bar Entrance (BE).

The infection risks calculated in the 2007 study for recreational contact ranged from <0.01 to 1.04% for rotavirus at the Bar Entrance (BE). This assumed that 99.98% of viruses were removed by treatment (LRV = 3.7). The range reflected the scenarios modelled. The highest risks were associated with illness outbreaks in the community; under normal conditions, the risks were <0.01% when there was no outbreak. The current study estimates the risk of illness from norovirus in the range of 0.0034 to 0.0668% depending on LRV (3 to 4) and whether the plant was discharging. As a result, the 2007 study had a much greater range than the current study, as they modelled multiple separate events, but the results overlapped even with different pathogens and health outcomes.

[3] Only considering the S4 scenario presented in the 2007 QMRA

5 Acknowledgements

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6 Glossary of abbreviations and terms

Attribute	Something that can measure and tells us about the state of a river or lake.
E. coli	*Escherichia coli.* The preferred faecal indicator bacteria for freshwater microbiological water quality assessment in New Zealand.
enteric virus	A pathogenetic virus which results in gastroenteritis in humans.
Enterococci	Enterococci are the preferred faecal coliforms for marine microbiological water quality assessment in New Zealand.
exposure pathway	Describes the source of the pathogen, transport route, barriers to exposure and the mechanism of exposure.
faecal coliforms	Faecal coliforms are a faecal indicator bacteria used in some standards to assess the safety of water for shellfish harvesting.
FIB	Faecal indicator bacteria. Excreted bacteria whose presence indicates faecal contamination and the potential presence of other excreted microorganisms such as pathogens.
hazard	Something, such as a pathogenic microorganism, that has the potential to cause harm,
hazardous event	An event which introduces a hazard (pathogen) into the water or fails to remove the hazard from the water.
hockey stick distribution	An empirical distribution commonly used to describe pathogen concentration in raw wastewater.
ID_{50}	Infectious median dose. The dose which 50% of the population is infected.
Illness	Infection with enteric viruses causes symptoms such as gastroenteric symptoms, sickness, and diarrhoea.
Infection	When a pathogenic organism enters a person and the pathogenic organism multiplies.
LRV	Log Reduction Value is a measure of how effective a treatment system is at removing pathogens.
MAC	Microbiological Assessment Category – estimate of water quality data.
Norovirus	An enteric virus commonly associated with gastroenteritis.
NPS-FM	National Policy Statement - Fresh Water.
PERT distribution	The Program Evaluation and Review Technique or PERT distribution is a continuous statistical distribution defined by minimum, mode and maximum values. It is used to model values obtained from expert opinion.
Phage	A virus that infects bacteria which is often used as a surrogate for enteric viruses because phages are more straightforward to monitor than enteric viruses.
QMRA	Quantitative microbial risk assessment is a risk assessment technique where risks are quantified.
Risk	the likelihood a hazardous event causes harm, such as infection or illness.

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Appendix A Schematic diagram of the BSTP

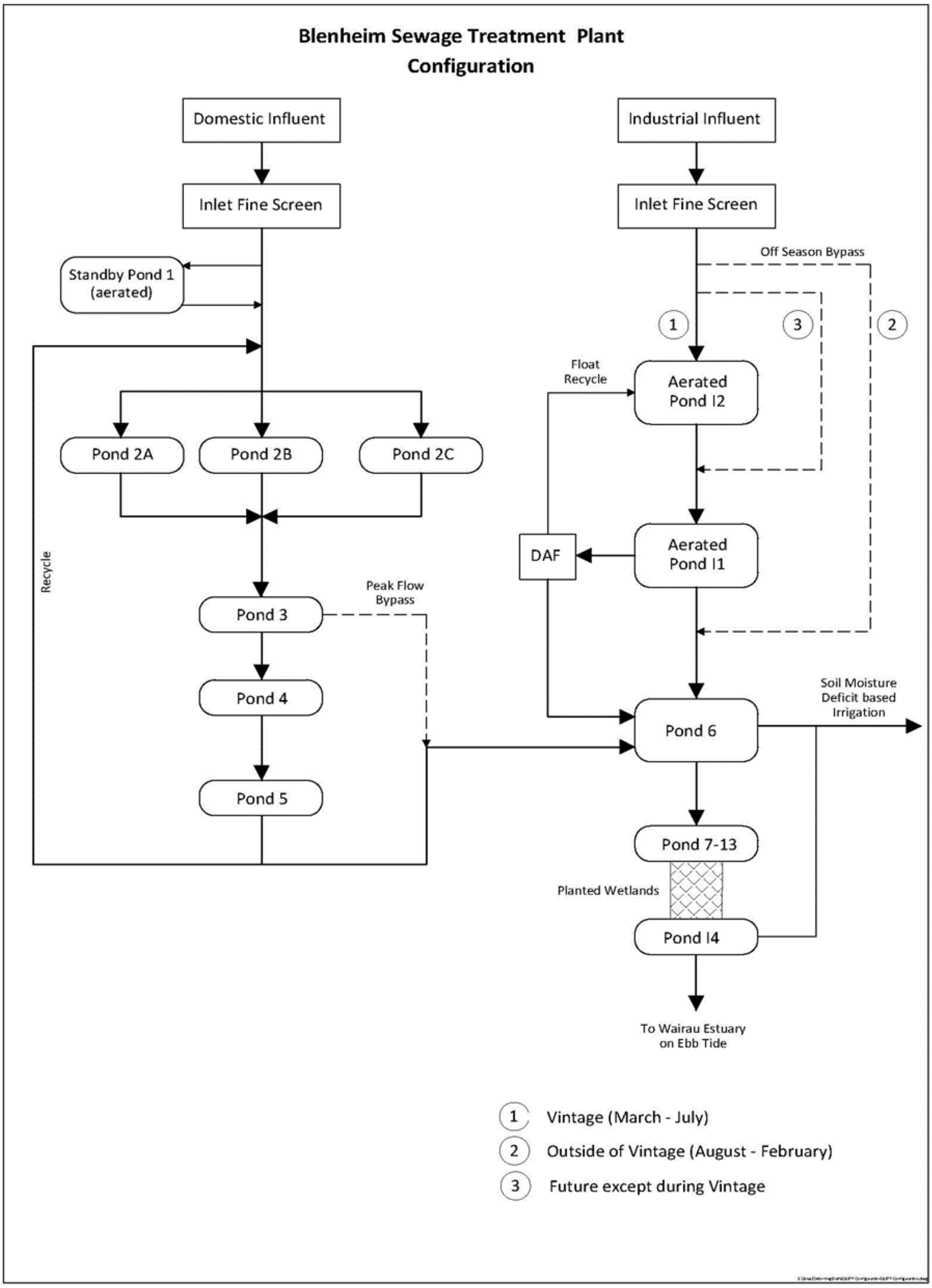

Figure A-1: Schematic layout of BSTP showing the configuration of the domestic and industrial streams. Diagram provided by Beca Ltd.