

Water quality of Estuary of the Heathcote and Avon Rivers / Ihutai

Prepared for Environment Canterbury

June 2020



NIWA - enhancing the benefits of New Zealand's natural resources

www.niwa.co.nz

Prepared by:

Jennifer Gadd Bruce Dudley John Montgomery Amy Whitehead Richard Measures David Plew

For any information regarding this report please contact:

Bruce Dudley

National Institute of Water & Atmospheric Research Ltd Private Bag 99940 Viaduct Harbour Auckland 1010

Phone +64 9 375 2050

| NIWA CLIENT REPORT No: | 2020183AK |
|------------------------|-----------|
| Report date: | June 2020 |
| NIWA Project: | ENC20504 |

| Quality Assurance Statement | | | | | | | |
|-----------------------------|--------------------------|-----------------|--|--|--|--|--|
| Joen Jules | Reviewed by: | John Zeldis | | | | | |
| Jan | Formatting checked by: | Emma Hope-Ede | | | | | |
| J.P. Moorco | Approved for release by: | Jonathan Moores | | | | | |

© All rights reserved. This publication may not be reproduced or copied in any form without the permission of the copyright owner(s). Such permission is only to be given in accordance with the terms of the client's contract with NIWA. This copyright extends to all forms of copying and any storage of material in any kind of information retrieval system.

Whilst NIWA has used all reasonable endeavours to ensure that the information contained in this document is accurate, NIWA does not give any express or implied warranty as to the completeness of the information contained herein, or that it will be suitable for any purpose(s) other than those specifically contemplated during the Project or agreed by NIWA and the Client.

Contents

| Exec | utive s | ummary | 7 |
|------|---------|---|----|
| 1 | Intro | duction | 10 |
| 2 | Wate | er quality monitoring | 11 |
| 3 | Ihuta | ai / Avon-Heathcote Estuary water quality has changed over time | 14 |
| | 3.1 | How do we assess change over time? | 14 |
| | 3.2 | Water quality has improved since 2007 | 15 |
| | 3.3 | Diversion of wastewater discharge improved water quality | 26 |
| | 3.4 | Changes since 2014 have been more variable | 30 |
| | 3.5 | Nutrients and faecal indicator bacteria can vary seasonally | 33 |
| | 3.6 | Patterns in seasonality have not changed substantially | 39 |
| | 3.7 | Summary | 40 |
| 4 | Wate | er quality is currently poor at many sites | 41 |
| | 4.1 | How do we assess the current state of water quality? | 41 |
| | 4.2 | Estuaries are where rivers meet the sea | 41 |
| | 4.3 | Water quality has potential to affect human health | 42 |
| | 4.4 | Water quality has potential to affect ecological health | 46 |
| | 4.5 | Summary | 54 |
| 5 | Nutr | ients may be leading to eutrophication of Ihutai/Avon-Heathcote Estuary | 55 |
| | 5.1 | Nutrient concentrations only tell part of the story | 55 |
| | 5.2 | Ihutai/Avon-Heathcote Estuary is in a eutrophic state | 55 |
| | 5.3 | Algal growth affects sea grass, invertebrates and fish | 57 |
| 6 | Ther | e are multiple drivers of water quality in the estuary | 58 |
| | 6.1 | The wastewater diversion improved water quality and ecology | 58 |
| | 6.2 | Nutrient loads from rivers affect estuary water quality | 58 |
| | 6.3 | The nutrient contribution from bird life is likely minor | 59 |
| | 6.4 | Some sites are more affected by river water than others | 61 |
| | 6.5 | Effects of estuary water residence time and temperatures | 64 |
| | 6.6 | Summary | 64 |
| 7 | The | current monitoring programme is fit for purpose | 65 |

| | 7.1 | low did we approach this?65 |
|------|--------|--|
| | 7.2 | lumber and locations of monitored sites65 |
| | 7.3 | Nonitored variables67 |
| | 7.4 | ampling and measurement methods70 |
| | 7.5 | dditional information and metadata to explain patterns of water quality and cological issues |
| | 7.6 | ummary74 |
| | 7.7 | ecommendations |
| 8 | Sumn | ry of key questions |
| 9 | Ackno | vledgements |
| 10 | Gloss | y of abbreviations and terms78 |
| 11 | Refer | nces |
| Арре | ndix A | Key questions addressed in this report |
| Арре | ndix B | Seasonality analysis for trend assessment |
| Арре | ndix C | Time series plots of variables not shown in section 3.2 |
| Арре | ndix D | Water quality trends from 2014 to 201991 |
| Арре | ndix E | Methods used in this assessment102 |

Tables

| Table 3-1: | Level of confidence categories used to convey the probability that water | |
|------------|---|-------|
| | quality trends were increasing or decreasing. | 15 |
| Table 3-2: | Summary of trend directions (arrows) and magnitude (percentage change | e per |
| | year) for each water quality variable for 2007 to 2019. | 16 |
| Table 3-3: | Nutrient and chlorophyll-a changes before and after diversion of treated | |
| | wastewater discharge for river (blue), estuary (teal) and coastal sites (gree | en). |
| | | 27 |
| Table 3-4: | Changes in suspended solids, turbidity, dissolved organic carbon and bact | eria |
| | before and after diversion of treated wastewater discharge for river (blue | e), |
| | estuary (teal) and coastal sites (green). | 28 |
| Table 3-5: | Changes in salinity, dissolved oxygen and pH before and after diversion of | F |
| | treated wastewater discharge for river (blue), estuary (teal) and coastal si | tes |
| | (green). | 29 |
| Table 3-6: | Summary of trend directions (arrows) and magnitude (percentage change | e per |
| | year) for each water quality variable for the period January 2014 to Decer | nber |
| | 2019. | 31 |

| Table 4-1: | 1: Summary of indicator bacteria counts and suitability for swimming and other | | | | |
|------------|--|----|--|--|--|
| | forms of contact recreation. | 44 | | | |
| Table 4-2: | Summary of faecal coliform concentrations and suitability for shellfish | | | | |
| | gathering. | 45 | | | |
| Table 4-3: | Water quality guidelines used to assess potential effects on ecological health | ۱. | | | |
| | | 46 | | | |
| Table 6-1: | Contributions of rivers and oceanic nutrient sources to each estuary zone. | 62 | | | |
| Table 7-1: | Recommended water quality variables for SoE and recreational water quality monitoring of marine and estuarine water quality from selected recent | y | | | |
| | reports. | 68 | | | |

Figures

| Figure 2-1: | Locations in and around Ihutai / Avon-Heathcote Estuary where water qualities is measured. | ty 11 |
|--------------|---|----------|
| Figure 3-1: | Ammoniacal nitrogen and Dissolved Reactive Phosphorus concentrations fro January 2007 to December 2019 at estuary monitoring sites. | om 17 |
| Figure 3-2: | Total nitrogen (left) and total phosphorus (right) concentrations from Janua 2007 to December 2019 at estuary monitoring sites. | ry 19 |
| Figure 3-3: | NOx-N (left) and chlorophyll- <i>a</i> (right) concentrations from January 2007 to December 2019 at estuary monitoring sites. | 20 |
| Figure 3-4: | Total (left) and volatile (right) suspended solids concentrations from January 2007 to December 2019 at estuary monitoring sites. | ′ 22 |
| Figure 3-5: | Turbidity (left) and dissolved oxygen (right) from January 2007 to December 2019 at estuary monitoring sites. | 23 |
| Figure 3-6: | Dissolved organic carbon and enterococci from January 2007 to December 2019 at estuary monitoring sites. | 24 |
| Figure 3-7: | Total copper (left) and zinc (right) from January 2007 to December 2019 at estuary monitoring sites. | 25 |
| Figure 3-8: | Summary plot representing for each water quality measurement the proportion of sites with trends at each categorical level of confidence. | 30 |
| Figure 3-9: | Seasonal variations in NOx-N at all sites. | 34 |
| Figure 3-10: | Seasonal variations in DRP at all sites. | 35 |
| Figure 3-11: | Comparison of the seasonal variation in monthly means of NOx-N and DRP a all sites. | nt 36 |
| Figure 3-12: | Seasonal variations in enterococci at all sites. | 37 |
| Figure 3-13: | Seasonal variations in <i>E. coli</i> at the eight sites regularly monitored. | 38 |
| Figure 3-14: | Seasonal variation in NOx-N at Ōpāwaho/Heathcote River (left) and Ōtākaro/Avon River (right) from 2013 to 2019. | 39 |
| Figure 3-15: | Seasonal variation in DRP at Ōpāwaho/Heathcote River (left) and Ōtākaro/Avon River (right) from 2013 to 2019. | 40 |
| Figure 4-1: | Salinity of river, estuary and coastal monitoring sites. | 42 |
| Figure 4-2: | Enterococci and <i>E. coli</i> counts at monitoring sites. | 43 |
| Figure 4-3: | Faecal coliform concentrations at estuary monitoring sites. | 45 |
| Figure 4-4: | Temperature and dissolved oxygen at monitoring sites relative to guideline values. | 48 |

| Figure 4-5: | Dissolved inorganic nitrogen atmonitoring sites relative to guideline in | |
|-------------|---|------|
| | Canterbury Land and Water Regional Plan . | 49 |
| Figure 4-6: | Nutrients at monitoring sites relative to guideline values as listed in Table 4 | 4-3. |
| | | 50 |
| Figure 4-7: | Total suspended solids and turbidity at monitoring sites relative to guidelin | e |
| | values. | 52 |
| Figure 4-8: | Total copper, lead and zinc at monitoring sites compared to guidelines to | |
| | protect species from toxicity (ANZECC 2000). | 53 |
| Figure 5-1: | Annual riverine and drain loads of DIN to the estuary. | 56 |
| Figure 6-1: | Annual riverine inputs of DIN and DRP to the estuary | 59 |
| Figure 6-2: | Bird species' contributions to total number, and total daily nutrients and | |
| | bacteria. | 59 |
| Figure 6-3: | Contribution of waterbirds and three rivers to the total daily nutrients and | |
| | bacteria in the estuary. | 60 |
| Figure 6-4: | Percent contribution of coastal seawater to waters of Ihutai/Avon-Heathco | te |
| | Estuary | 61 |
| Figure 6-5: | Modelled concentrations (mg/L) of ammoniacal-N (left), NOx-N (middle) an | nd |
| | DRP (right) in Ihutai/Avon-Heathcote Estuary based on median flow and | |
| | nutrient concentrations from 2014-2019. | 63 |

Executive summary

The Canterbury Regional Council (Environment Canterbury, ECan) have commissioned this report to provide up-to-date information on the water quality of the Estuary of the Avon and Heathcote Rivers / Ihutai (hereafter Ihutai / Avon-Heathcote Estuary). ECan requested analyses to provide up-to-date answers to eight key questions, and a report on the findings for use by ECan, the Christchurch-West Melton Zone committee and partner agencies including local rūnanga, the Christchurch City Council and Avon-Heathcote Estuary Ihutai Trust to make informed decisions regarding environmental management. This report presents the statistical analysis, modelling, and ecological and human health assessments to answer these key questions (see below).

ECan monitor physico-chemical and microbiological water quality of the estuary every month at 11 sites. Samples are collected and on-site measurements are taken during high tide at seven sites within the estuary and two sites on the coast; and during low tide at three sites: the Ōtākaro/Avon River mouth, Ōpāwaho/Heathcote River mouth and at Shag Rock (also sampled at high tide). This report covers data collected from January 2007 to December 2019.

Question 1: Are parameter concentrations/values at each monitored site changing (increasing or decreasing), i.e. are there trends over time? The main focus is to be on the most recent 6-year period (Jan 2014-Dec 2019), however, where possible also complete trend analysis for the complete time period.

Results from the water quality monitoring programme show that water quality of the estuary has improved since 2007. The diversion of Christchurch City's wastewater treatment plant discharge in March 2010 resulted in major improvements almost immediately, particularly for nutrients important for macroalgal growth (dissolved reactive phosphorus (DRP) and ammoniacal-N) and volatile suspended solids. Nitrate-nitrite-N (NOx-N) in the estuary did not decrease substantially as the major sources of this form of nitrogen are the Otākaro/Avon and Opāwaho/Heathcote rivers. Since 2014, there have been significant increases of chlorophyll-a, an indicator of phytoplankton abundance, at all sites, and at South New Brighton Park, Humphreys Drive and Sandy Point chlorophyll-a has increased by more than 20% each year, from concentrations of <1 μ g/L in 2014 to \sim 3 µg/L by 2019. The faecal indicator bacteria enterococci has also increased at all sites, except the Penguin Street site, where it may or may not have increased. The rate of increase could only be calculated at three sites and was ~2-3 MPN/100mL each year. Since 2014 there have been decreases in concentrations of total nitrogen at most sites, particularly those near the estuary mouth and coast. Turbidity, a measure of water clarity has decreased at most sites including the river mouths, and at some sites by >10% per year. DRP is also likely to have decreased at most sites, except at the Ōtākaro/Avon River mouth.

Question 2: Determine the changes in parameter concentrations/values as a result of the removal of wastewater discharge from the estuary in March 2010.

The diversion of Christchurch City's wastewater discharge to the ocean outfall, resulted in a significant decrease in the ammoniacal-N, DRP and TP concentrations at all sites in the estuary, including at river mouths, the estuary mouth and on the coast. Chlorophyll-*a* also decreased at almost all sites. Other forms of nitrogen (NOx-N and total nitrogen) did not decrease at all sites, indicating the on-going contribution of NOx-N from the rivers to the estuary nutrient load. Volatile suspended solids, a measure of organic matter suspended in the water, also decreased at almost all sites, excluding the two river mouths and the South New Brighton Park site, close to the

Ōtākaro/Avon River mouth. There was a less dramatic effect on other variables, including faecal indicator bacteria, with some sites increasing, some decreasing and others showing little change.

Question 3: What is the likely impact of the measured nutrient, oxygen, turbidity, TSS, temperature, metals, and faecal indicator bacteria concentrations on the ecological functioning (ecosystem health) of the estuary and human health? That is, assess measured values against relevant guideline values for ecosystem and human health. Consider macroalgae growth and hence trophic state and seafood safe to eat.

The faecal indicator bacteria concentrations indicate high risks of gastrointestinal illness or campylobacter infection at the sites near the river mouths or inner estuary. Sites near the outer estuary have moderate risk, whereas the sites at the coast have low risk. The water is not suitable for shellfish gathering, except at the Beachville Road site.

The water quality could be adversely affecting the estuarine ecology, based on comparing the water quality measured in the last five years to a number of different guidelines appropriate for freshwater, estuarine and coastal waters. Ammoniacal-N, NOx-N, total nitrogen, DRP and total phosphorus frequently exceed the guidelines at many sites, except those closest to the estuary mouth and coast. Total Suspended Solids (TSS) at the Ōpāwaho/Heathcote River mouth exceeds the water quality guideline developed for rivers in Canterbury, and turbidity exceeds the ANZG (2018) guidelines for freshwater at both river mouth sites, though these measurements are lower and within guidelines at other locations. Dissolved oxygen concentrations are consistently above guidelines, as expected for daytime monitoring.

Nutrient concentrations in estuaries can be low when measured, yet still be affecting the ecological state, because they can be rapidly taken up by algae (phytoplankton and macroalgae) The best way to assess the effect of nutrients is to look at the loads entering the estuary. The Estuary Trophic Index (ETI) Tool 1 was used to determine the susceptibility of the estuary to eutrophication. The model combined nutrient loads from the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers, the drains near the wastewater treatment ponds, and coastal waters to determine nutrient condition in the estuary. This assessment indicated the estuary is in Band D of the four ETI classifications – 'very high susceptibility to macroalgal eutrophication'. Macroalgal eutrophication can lead to reduced habitat for benthic macrofauna, reductions in sea grass and consequent reductions in shellfish and juvenile fish, and noxious odour when algae dies off and rots.

Question 4: Are seasonal patterns in parameter concentrations changing?

Dissolved nutrient concentrations (ammoniacal-N, NOx-N and DRP, vary seasonally at many of the estuary sites and in the two main rivers entering the estuary, predominantly with lower concentrations during summer and higher concentrations during winter. However the seasonal pattern in DRP varies from the river sites, where lowest concentrations are found in spring and highest in autumn, throughout the estuary to the coast, where lowest concentrations are found in summer and highest in winter. The faecal indicator bacteria E. coli also varies seasonally, with lowest concentrations in winter and highest in summer, however the enterococci bacteria do not show such a pattern.

The seasonal patterns in NOx-N and DRP concentrations at the two river mouths were compared over time. Typical seasonal patterns were seen from 2013 to 2016, and since then there has been some variation due to flood events and unusual results, however there has not been a clear shift in the seasonal patterns.

Question 5: The estuary supports a diversity and abundance of birds. What contribution are the birds likely making to the nutrient and micro-organism concentrations and hence water quality within the estuary?

Birds using the estuary can add to the nitrogen, phosphorus and bacteria load. Estimates of their nutrient contribution suggest that this is minor compared to the rivers and drains, but their input of faecal indicator bacteria is potentially very high. There is considerable uncertainty around this estimate of bacteria input from birds including whether bacteria are deposited into the estuary or on the margins, as well as their die off rates and spatial distribution. We do not have recent data on bird counts in the estuary to understand whether the contribution from birds has changed over time.

Question 6: Identify the possible current drivers of water quality issues in the estuary and the likely ecological effects of these.

There are multiple historic and current drivers of water quality in the estuary. The diversion of the wastewater discharge resulted in rapid improvements in nutrient concentrations, macroalgal biomass and condition, and reductions in pollution-tolerant polychaetes. These effects were all observed between 2010 and 2014.

The two main rivers entering the estuary remain major drivers of estuarine water quality. The nutrient mass loads from both rivers change over time, but there is not a clear direction of change; the loads fluctuate as the flows fluctuate due to rainfall variation. These fluctuating loads affect the water quality in the estuary, particularly at sites close to the rivers. Three-dimensional numerical modelling confirms that Ōtākaro/Avon and Ōpāwaho/Heathcote rivers are the primary drivers of water quality at the South New Brighton Park and Humphreys Drive sites respectively, but there are other influences on the water quality at Sandy Point. It is likely that the perimeter drains of the wastewater treatment ponds affect this site, resulting in ammoniacal-N and DRP concentrations that are much higher than at other sites. The changes in bathymetry of the estuary due to the Christchurch earthquakes may also affect the water quality and the growth of phytoplankton. Changing temperatures are also likely to be influencing the growth of algae, which in turn affects water quality.

Question 7: An assessment of the adequacy of the current water quality monitoring programme for measuring state and trend in the estuary, and for identifying issues as they arise.

The current water quality monitoring programme includes a high number of sites for an estuary of this size which enables the spatial variability across the estuary to be examined. The variables monitored are appropriate for assessing changes in ecological state and the frequency of monitoring is satisfactory for assessing trends over time. Regular monitoring of macroalgal populations is recommended as a key bioindicator of the response to nutrient and climatic conditions in the estuary.

Question 8: Recommendations for further data sets required to investigate issues further

Further investigations in the estuary should focus on:

- acquiring accurate and up-to-date information on the water quality of the drains entering the estuary near Sandy Point and in the vicinity of the wastewater treatment ponds,
- acquiring flow information for the Linwood Canal, and
- modelling of residence time in the estuary to assess the effect post-earthquake changes to bathymetry could be having on water quality and algal growth.

1 Introduction

The Canterbury Regional Council (Environment Canterbury - ECan) has monitored physico-chemical and microbiological water quality monthly at 11 sites within and just outside Ihutai / Avon-Heathcote Estuary since 2007, as part of the "Healthy Estuary and Rivers of the City" (HERC) monitoring programme. ECan need the data collected to be analysed to provide up to date information on the water quality of the estuary for use by ECan, the Christchurch-West Melton Zone committee and partner agencies including local rūnanga, the Christchurch City Council and Avon-Heathcote Estuary Ihutai Trust to make informed decisions regarding environmental management.

ECan requested that this report provide answers to the eight questions listed in Appendix 1. This report provides that information, based on robust statistical analysis of the water quality data, use of the national Estuary Trophic Index (ETI) tool (Robertson et al. 2016a, Zeldis et al. 2017c), and 3-dimensional estuary modelling to understand mixing of fresh and salt waters in the estuary. The contents and structure of this report are derived from the eight questions. Hence this report includes:

- an explanation of the water quality monitoring that is undertaken by ECan (Section 2);
- an assessment of the changes in water quality over time, including long-term trends, changes with the diversion of the wastewater discharges and more recent trends (since 2014) and an assessment of the seasonality in water quality variable concentrations (Section 3);
- an assessment of the current state of water quality in the estuary, based on data for the most recent years (2015-2019) (Section 4);
- a more in-depth assessment of nutrient concentrations and loads and susceptibility to eutrophication (Section 5);
- a discussion of the main drivers of water quality in the estuary, including an assessment of the contribution of birds to water quality, and using a water quality model to understand sources at each monitoring location (Section 6);
- a review of the current monitoring programme, including its suitability and effectiveness (Section 7); and
- a summary of the main findings with answers to the key questions asked by ECan (Section 8).

2 Water quality monitoring

Water quality has been monitored monthly since 2007 at 11 sites, within and near the mouth of the Ihutai / Avon-Heathcote estuary (Figure 2-1). Every month, measurements are made on site for temperature and pH and water samples are collected for analysis (see panels below for descriptions of water quality variables measured). Samples are collected at high tide for all sites except the two rivers, where samples are collected around low tide. Samples are also collected at the Shag Rock site at low tide, so there are samples from both high and low tide for this site. The Spit Tip site is no longer monitored (sampled 2007-2014) as measurements were very similar to the nearby Shag Rock site.



Figure 2-1: Locations in and around Ihutai / Avon-Heathcote Estuary where water quality is measured. This sampling is conducted as part of the "Healthy Estuary and Rivers of the City" (HERC) monitoring programme. Sites are coloured by type: river (blue), estuary (light blue) or coastal (green).

Salinity

Salinity measures how salty the water is and indicates mixing of fresh and salt waters. In estuaries it tells us how much of the water at any one place comes from the land (freshwater) and how much from the sea. Because of this, salinity is important for assessing sources of contaminants.



Photo: A Herbstritt (NIWA)

Nutrients

Nitrogen and phosphorus are important nutrients supporting marine plant growth. In spring and summer, when light and temperature levels do not limit plant growth, nutrient supply typically limits growth rate. Hence, especially in spring and summer, increases in nutrient supply (e.g., via sewage or fertiliser runoff from land) may lead to excessive algal growth, eutrophication, and noxious algal blooms. The photo to the right shows build-up of the nuisance algae *Ulva* spp. (sea lettuce) in Ihutai / Avon-Heathcote Estuary in January 2020. As algae rots it can degrade sediment chemical conditions, increasing toxic hydrogen sulphide levels and reducing oxygen availability for animals.



Photo: L Bolton-Ritchie (ECan)

Three forms of nitrogen are measured: dissolved ammoniacal-N (NH₄-N, which is formed from the breakdown of organic matter and is also found in sewage; dissolved nitrate+nitrite-N (NOx-N), which occurs when ammoniacal-N is oxidised or from fertilisers; and total nitrogen which includes the dissolved nitrogen and the nitrogen content of small particles in the water (primarily organic). Dissolved inorganic nitrogen (DIN¹) is calculated from ammoniacal-N and NOx-N and is an important indicator of nitrogen, as it includes all forms that can be readily used by primary producers like algae.

Two forms of phosphorus are measured: dissolved reactive phosphorus (DRP), which comes from soil, rock and fertilisers, and is the form most easily used by algae, and total phosphorus (TP), which includes dissolved and particulate phosphorus. Chlorophyll-*a* is a measure of the green algae (phytoplankton) in the water, which grows in the presence of nitrogen and phosphorus.

Water temperature and pH

The temperature of the water affects what plants and animals can live in the estuary. It also controls the amount of oxygen that can dissolve in the water, which is essential for animal life. Changes in water temperature over time can show us the effect climate change is having on the estuary.

The pH measures how acidic or basic / alkaline the water is. The pH of seawater is very resistant to change, but in an estuary, pH can change due to plant or microbial activity and their effect on CO₂ levels. When macrophytes (large plants) or microalgae (phytoplankton) are abundant, pH can get higher during the day when the plants photosynthesise and



consume CO_2 , and lower (more acidic) at night, when they respire (releasing CO_2). Microbial respiration also releases CO_2 , reducing pH.

¹ We calculated dissolved inorganic nitrogen (DIN) as the sum of ammoniacal-N and nitrate+nitrite-N (NOx-N). When either ammoniacal-N or NOx-N were not measured, we did not calculate DIN. When both measures were below the detection limit, we calculated DIN as below detection, using the highest detection limit. If either were below the detection limit, we calculated DIN by assuming the value below detection was equal to half of the detection limit.

Dissolved oxygen (DO)

Oxygen is needed by animals to survive. Oxygen is produced by marine algae during photosynthesis and reduced by respiration (by animals, plants and bacteria). Eutrophic waters (those with high algal growth) tend to have very high oxygen levels when algae are photosynthesising (e.g., during spring, and during the daytime), but can have dangerously low levels at other times (e.g., during the night when algae respire, and in autumn, as algae die and rot. Sediments in eutrophic environments tend to have very low oxygen levels and are black because of high hydrogen sulphide levels.



Photo: A-M. Schwarz (NIWA)



Sediment and turbidity

Total suspended solids (TSS) is a measure of all the particles floating within the water. This includes mud, stirred up from the seabed or transported into the estuary, as well as organic material like degrading plants and animals. Volatile suspended solids (VSS) is a measure of how much of that TSS is made of organic matter.

Turbidity is a measure of water clarity – how clear or murky the water is. Turbidity measures the light scattering by suspended particles, and so it is usually related to TSS (when TSS is high, turbidity is high).

Faecal indicator bacteria

Enterococci, *E. coli* and faecal coliform bacteria are measured in coastal waters because their abundances indicate recent faecal pollution (e.g., from wastewater overflows, birds or dogs) and the potential presence of human faecal pathogens. At high levels they represent a higher risk of infectious disease from waterborne pathogens. Enterococci is an indicator of the suitability of water for contact recreation in saline waters, *E. coli* is the preferred indicator for freshwaters, and faecal coliform bacteria are an indicator of the suitability for gathering shellfish.



Photo: L Bolton-Ritchie (ECan)

Metals

Metallic elements like cadmium, chromium, copper, lead, nickel and zinc can be found in estuaries in urban areas and originate from sources including stormwater (e.g., from zinc-coated roofs, copper from wear of copper brake pads), industrial discharges, wastewater overflows and landfills. These metals can be toxic to aquatic organisms if present at high enough concentrations. They can also accumulate in sediments and some metals accumulate in shellfish. Metals are only measured every two months in the estuary.



3 Ihutai / Avon-Heathcote Estuary water quality has changed over time

3.1 How do we assess change over time?

We first checked the monitoring data for errors, such as data in the wrong units. We also used timeseries plots and box-plots to identify outliers for each variable, and ECan looked at the original laboratory data sheets to check for errors ². For several water quality variables, some values were too low for laboratories to measure, and these were reported as less than a "detection limit". The data set included many DRP, NH₄-N, metal and indicator bacteria measurements that are below detection limits, and are referred to as "censored values" and need to be treated appropriately in statistical analyses (see below).

To assess how water quality at the Ihutai/Avon-Heathcote Estuary has changed over time we looked at trends in water quality over the entire HERC 13-year record, from January 2007 to December 2019, and for the most recent six-year period from January 2014 to December 2019. Trends in metal concentrations were only assessed over a shorter timeframe, from October 2016 to December 2019 because, prior to October 2016, metals were measured only once per year in August. The change in metals monitoring frequency, coupled with the large proportion of results below the laboratory limit of detection, meant that trends determined across the entire period were not reliable.

The trends were assessed using the most recent methods developed for assessing water quality trends in New Zealand (McBride 2019) and as we have previously used in studies of river and lake water quality (Larned et al. 2018). These methods provide a more robust assessment of trends when there are censored values in the data set, and are an improvement on those used in the national assessment of coastal water quality trends (Dudley & Jones-Todd 2018).

We assessed the direction of the trend using a Kendall test and we estimated our confidence in that direction based on Kendall p-values, expressed as a probability that a trend is increasing or decreasing (Snelder & Fraser 2019). We used nine confidence categories to express this, based on categorical levels of confidence recommended by the Intergovernmental Panel on Climate Change (IPCC; Stocker et al. 2014). The narrative descriptions of confidence associated with different probabilities are shown in Table 3-1.

Increasing trends in concentrations of nutrients, enterococci, *E. coli*, suspended solids, chlorophyll-*a*, turbidity and metals indicate degrading water quality and they can be detrimental to ecosystem health. For these measurements, we use colours to indicate increases (degrading trends, shown in orange-red) or decreases (improving trends, shown in green) in the arrows and trend lines. We have not classed trends in pH, salinity and temperature as 'improving' or 'degrading' as we cannot say with confidence which direction of trend in these variables reflects improvement in ecosystem health. For these variables we use grey shading to indicate the level of confidence: light grey indicates low probability (as likely as not in Table 3-1) and black indicates high probability of either an increase or decrease.

² Any errors detected through this process were corrected in the data we used and in the original database held by Environment Canterbury.

Table 3-1:Level of confidence categories used to convey the probability that water quality trends wereincreasing or decreasing.The confidence categories are those used by the Intergovernmental Panel on ClimateChange (IPCC; Stocker et al. 2014).

| Probability (%) | Level of confidence in direction | Colour used for arrows and trend lines |
|-----------------|--|--|
| 99–100 | Virtually certain increase | Dark red |
| 95–99 | Extremely likely increase | Bright red |
| 90–95 | Very likely increase | Dark orange |
| 67–90 | Likely increase | Orange |
| 33–67 | As likely to have increased or decreased | Yellow |
| 10–33 | Likely decrease | Yellow green |
| 5–10 | Very likely decrease | Mid green |
| 1–5 | Extremely likely decrease | Dark green |
| 0–1 | Virtually certain decrease | Very dark green |

We estimated the trend magnitude using the Sen Slope Estimator (SSE), or the seasonal version where appropriate. Some variables vary seasonally, and this affects trend assessment. We tested all variables and sites for seasonality using the Kruskall-Wallis test, and if they were seasonal, then a seasonal version of the trend assessment was used and the Seasonal Sen Slope Estimator (SSSE) was calculated. The tables in Appendix B indicate whether the trends used seasonal or non-seasonal versions of the SSE.

We assessed the effect of the wastewater diversion on water quality by looking at water quality data for three years before the discharge was diverted (1 March 2007 to 1 March 2010) and three years after. The time period we selected for after the diversion was from 1 October 2011 to 1 October 2014, to avoid the emergency discharges of untreated wastewater that occurred after the Christchurch earthquakes. The three-year period provides a good number of data points (up to 36 for the monthly monitoring) to allow an average to be calculated, that is not affected by the seasonal influence. We used statistical methods suitable for censored data to calculate the median concentrations for the before and after period (robust Regression on Order Statistical (ROS) methods, described by Helsel (2012), see section 4.1 for more details) and to compare the before and after data sets (generalized Wilcoxon test, which tests if the distribution of each data set is the same (Helsel 2012)).

3.2 Water quality has improved since 2007

The trend directions and magnitude for all sites and variables are summarised in Table 3-2. The most apparent changes over time (of greatest magnitude and occurring at the most sites) have occurred in nutrients. In particular, **ammoniacal-N** and **dissolved reactive phosphorus (DRP)** have shown much lower concentrations since the wastewater discharge into the estuary ceased in 2010 (see further analysis in section 3.3). Ammoniacal-N and DRP concentrations decreased at all sites including the two river sites and the coastal locations (Figure 3-1, Table 3-2). The highest concentrations of ammoniacal-N and DRP were measured at Sandy Point and Humphreys Drive and South New Brighton Park – the three sites most affected by the wastewater discharge and at these sites the changes over time were most obvious. From 2011 onwards, seasonal variation in DRP concentrations become apparent at Beachville Road, Shag Rock, Cave Rock and the Southshore Beach sites.

Table 3-2: Summary of trend directions (arrows) and magnitude (percentage change per year) for each water quality variable for 2007 to 2019. The direction of the arrow indicates the direction of the trend (e.g., up arrow means concentrations are increasing, down arrow means concentrations are decreasing). The colour of the arrow refers to the categorical level of confidence in the trend: for decreasing trends (improving water quality), dark green signifies a virtually certain decrease in concentration and light green a likely decrease, while for increasing trends (degrading water quality), dark red signifies a virtually certain increase in concentration and orange a likely increase. Horizontal arrows indicate concentrations are as likely to be increasing as decreasing (no clear trend). Black / grey arrows are used for salinity, water temperature and pH as trend directions for these variables do not indicate improvement in ecosystem health. The level of confidence in the increases or decreases in concentration are shown as a gradient from black (virtually certain) to light grey (likely).

| | Ōtākaro / Avon River | Ōpāwaho / Heathcote River | South New Brighton Park | Humphreys Drive | Sandy Point | Penguin Street | Beachville Road | Shag Rock Iow | Shag Rock high | Cave Rock | Southshore Beach |
|------------------------------------|-------------------------|---------------------------------|-------------------------------|--------------------|-------------------|-------------------|--------------------|------------------|-------------------|---------------|---------------------|
| Salinity | ⇔ 0.0 | -5.1 | 1.6 | ⇔ 0.1 | 1 .1 | ♠ 0.4 | ♠ 0.3 | ♠ 0.6 | -0.1 | ♠ 0.1 | 1 0.1 |
| Water temperature | 1 0.3 | 10.1 | 1 0.2 | ₽ -0.7 | - 0.7 | 10.3 | ↓ -0.5 | -0.1 | ₽ -0.3 | - 0.6 | 1 0.2 |
| Dissolved Oxygen saturation (%) | 1 0.5 | - 0.5 | 1 0.6 | - 2.6 | - 0.1 | 1 0.6 | ↓ -1.2 | ₽ -0.5 | ₽ -0.6 | ₽ -0.3 | ₽ -0.3 |
| pH ¹ | 1 0.3 | 10.0 | 1 0.3 | - 0.1 | 1 0.3 | 1 0.3 | ♠ 0.0 | 1 0.2 | ♠ 0.1 | ♠ 0.1 | 1 0.1 |
| Ammoniacal-nitrogen | -12 | -17 | - 27 | - 24 | - 17 | - 40 | - 19 | - 31 | - 7.0 | - 7.7 | ↓ NC |
| Nitrate-N + nitrite-N (NOx-N) | ₽ -2.5 | <u></u> | - 4.8 | - 5.0 | -6.4 | ₽ -12 | - 9.0 | - 5.5 | -6.9 | - 7.2 | ₽ -5.9 |
| Dissolved Inorganic Nitrogen (DIN) | - 3.5 | ↓ -1.4 | - 11 | -12 | ₽ -13 | ₹-22 | - 14 | - 17 | -8.8 | ₽ -11 | ₽ -11 |
| Total Nitrogen | - 3.8 | ↓ -1.3 | - 7.4 | -6.8 | -8.8 | - 9.1 | ↓ -2.5 | - 9.7 | + -0.6 | ⇔ 0.0 | ↓ -1.3 |
| Dissolved Reactive Phosphorus | - 2.9 | ↓ -12.3 | - 21 | - 18 | ₽ -14 | -25 | -13 | - 19 | - 7.2 | - 10 | - 7.2 |
| Total Phosphorus | - 4.7 | -7.6 | - 15 | -13 | - 10 | - 16 | - 12 | - 19 | - 9.0 | - 7.1 | - 8.4 |
| Chlorophyll-a | + -1.4 | 1.7 | 1 2.8 | ₽ -6.2 | - 4.5 | - 4.5 | ↓ -2.9 | - 7.4 | -6.6 | + -1.6 | + -1.1 |
| Total Suspended Solids (TSS) | - 1.7 | <mark>↓</mark> -0.6 | - 1.5 | - 5.5 | - 7.3 | - 4.4 | - 9.6 | ↓ -3.4 | ↓ -13 | ₽ -11 | -13 |
| Volatile Suspended Solids (VSS) | - 4.0 | ↓ -3.6 | - 2.4 | ₽ -6.2 | -6.7 | ₩ 0.0 | - 0.7 | - 4.1 | - 10 | - 4.9 | - 5.9 |
| Turbidity | 1.7 | 1.7 | 10.6 | + -1.2 | - 7.4 | + -1.1 | ↓ -6.4 | <u></u> 1.3 | - 5.8 | - 3.1 | - \ 3.5 |
| Enterococci | -6.4 | - 4.9 | ⇔ 0.0 | <u>†</u> 1.6 | <mark>↓</mark> NC | ₽ NC | 1 6.5 | ⇔ NC | NA | 1 3.2 | ⇔ NC |
| E. coli | <u>†</u> 1.9 | 1 3.7 | 1 2.5 | 1 2.6 | - 4.0 | ↓ NC | 1 NC | 1 NC | NM | NM | NM |

Note: ¹ pH was measured in the laboratory from 2007 to June 2015, then in the field from March 2013 to present. For the period where both was measured, the average pH was calculated. NC: Trend magnitude could not be calculated due to high proportion of censored data. NM: not measured at that particular site. NA: trend could not be assessed due to high proportions of censored data.



Figure 3-1: Ammoniacal nitrogen and Dissolved Reactive Phosphorus concentrations from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.

Changes in **total nitrogen (TN)** and **total phosphorus (TP)** are perhaps less dramatic (Figure 3-2), however there was a virtually certain reduction in TP at all sites, and in TN at all but three sites (Table 3-2). Of those three sites, located at the mouth or outside the estuary, the reduction was still considered likely at Shag Rock and Southshore Beach, but unclear at Cave Rock. TP includes organic and particulate phosphorus-containing compounds, while TN includes nitrate+nitrite-N (NOx-N), as well as organic and particulate nitrogen-containing compounds. The river waters entering the estuary have high concentrations of TN (Figure 3-2), mostly in the form of NOx-N (Figure 3-3). These inputs have dampened the positive effects of the wastewater diversion on reducing the dissolved forms of nitrogen (dissolved inorganic nitrogen, **DIN**, see Figure C-1), the form most readily used by algae in the estuary.

There were decreases in **NOx-N** concentrations at all sites (Table 3-2) except at Heathcote River at Ferrymead Bridge, where a decrease in concentrations is unlikely and it is likely to be increasing. Also, while the NOx-N decrease at Avon River was highly likely, it was not strong in terms of percent annual decrease relative to other sites. NOx-N shows strongly seasonal patterns at most sites (Figure 3-3), especially sites close to the estuary mouth and the coastal sites (these seasonal patterns are examined further in section 3.5).

Reduction in nutrient loading to the estuary due to diversion of the effluent discharge in 2010 is a likely cause of observed reductions in **chlorophyll**-*a* concentrations in estuary waters, although high variation over time makes trends in chlorophyll-*a* less visually apparent than nutrient reductions (Figure 3-3). Decreases in chlorophyll-*a* were extremely likely at Humphreys Drive, Sandy Point, Penguin Street, Beachville Road and Shag Rock (at both low and high tide), likely at Cave Rock, Southshore Beach and in the Avon River and very unlikely at South New Brighton or the Heathcote River (very likely increasing at these sites).



Figure 3-2: Total nitrogen (left) and total phosphorus (right) concentrations from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.



Figure 3-3: NOx-N (left) and chlorophyll-*a* (right) concentrations from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.

Total suspended solids (TSS) and **volatile suspended solids (VSS)** have also decreased over time at all sites (Figure 3-4), with the possible exception of TSS at the $\bar{O}p\bar{a}waho/Heathcote$ River where a decrease is only considered likely. On the other hand, the related indicator of **turbidity**, which is a measure of water clarity, showed a definite decrease at some, but not all sites (Table 3-2), and increases at both the river sites (Figure 3-5).

Dissolved organic carbon (DOC) was measured from 2007 to 2012 and was typically low in the estuary waters (e.g., 0.5-2 mg/L) but there were occasions where it was much higher and over 10 mg/L. For this reason, it is plotted using a log scale (Figure 3-6). There were no obvious changes in DOC over the period of monitoring or associated with the diversion of the treated wastewater discharge in 2010.

Dissolved oxygen saturation has decreased over time at Humphreys Drive, Beachville Road, Shag Rock (at both low and high tide), Cave Rock, Southshore Beach, in Ōpāwaho/Heathcote River, and possibly at Sandy Point (Table 3-2, Figure 3-5). However DO is extremely likely to have increased in the Avon River, at South New Brighton Park and at Penguin Street. An increase or decrease in DO in surface waters is not necessarily good or bad – but large fluctuations and very low values are eutrophication symptoms. Typically, large fluctuations would be most apparent in areas with both high photosynthetic and respiration rates. Very low values are most likely in deeper sections of estuaries where water column stratification occurs, and this is unlikely in wide, shallow, well-mixed estuaries such as Ihutai / Avon-Heathcote. The DO varied considerably at the Humphreys Drive site prior to the wastewater diversion, although night-time data (that would show DO levels when photosynthesis is low) are not available. Conversely Penguin Street has shown periodically very high DO values in recent years, possibly indicating increases in photosynthetic rates relative to respiration at that site. Seagrass beds have expanded at the Penguin Street area over that period (Gibson & Marsden 2016), potentially explaining this.

Enterococci counts have decreased at the Avon and Heathcote River and Penguin Street sites and are likely to have decreased at Sandy Point. However, counts have increased at Beachville Road and Cave Rock and are likely to have increased at Humphreys Drive. There is no clear direction of trend in enterococci counts at South New Brighton, Shag Rock and Southshore Beach (as likely to have increased as decreased). The trend analyses for the sites in the mid-estuary, mouth and coast are affected to some extent by the large number of measurements that were below the detection limit of 10 MPN/100mL from 2011 onwards.

Metals were only measured once per year from 2011 to August 2016, after which the frequency changed to every two months and many of the measurements have been below laboratory detection limits (Figure 3-7). The trends were not assessed for metals for the time period from 2007 to 2019, due to the combination of the low number of measurements above detection limits and the change in monitoring frequency. We recommend assessing trends in metals after August 2021, when five years of two-monthly monitoring data will be available.

The time series plots for water temperature, salinity, *E. coli* and faecal coliforms are shown in Appendix C. For these variables there was no consistency in the trend direction across sites, and for the most part only minor increases or decreases in the values.







Figure 3-5: Turbidity (left) and dissolved oxygen (right) from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.



Figure 3-6: Dissolved organic carbon and enterococci from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). DOC not monitored after November 2012. Note y-axis on log-scale and differs for each site to allow visibility of results. Flat line for enterococci at many sites is where limit of detection increased to 10 MPN/100mL from 2011 onwards. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.



Figure 3-7: Total copper (left) and zinc (right) from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Points are shown as follows: filled points are data above detection limit, and open circles are censored data (below detection limit). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage. Data from 7 August 2012 with very high detection limits (<0.053 and <0.11 mg/L for copper and zinc respectively) have been removed to improve visibility.

3.3 Diversion of wastewater discharge improved water quality

The diversion of the treated wastewater discharge had a major effect on nutrient concentrations in the estuary, but most clearly on **ammoniacal-N**, **DRP and TP** (Table 3-3). Concentrations were significantly lower after the discharge was diverted, both within the estuary and at nearby coastal sites (Southshore Beach and Cave Rock). There were decreases in **NOx-N**, **TN** and **chlorophyll-***a* at many of the sites within the estuary and at the Ōtākaro/Avon River mouth, but not at the Õpāwaho/Heathcote River mouth or at the coastal sites. Chlorophyll-*a* decreased at the three estuary sites that had highest concentrations prior to the diversion (Humphreys Drive, Sandy Point and Shag Rock), but not at all estuary locations (Table 3-3), indicating the influence of other inputs on chlorophyll-*a* at these sites.

The diversion had a mixed effect on concentrations of **TSS** in the estuary with decreased concentrations near the estuary mouth but not at other sites (Table 3-4). **VSS** concentrations reduced at most sites except the rivers and South New Brighton Park. There was also no overall change in **turbidity**, as it increased at some sites and decreased at others. Faecal indicator bacteria, as measured by **enterococci**, showed no change at estuary or coastal sites, but decreased at both river mouths.

There was no obvious change in **salinity** associated with the diversion of the discharge, at least not at the Sandy Point site closest to the historic wastewater release point in the inner estuary. Salinity increased at the $\bar{O}t\bar{a}karo/Avon$ River mouth and at the South New Brighton Park site; however this is likely due to changes in the bathymetry of the estuary after the earthquakes as there was considerable subsidence (up to ~ 0.4 m) at these two locations during the February 2011 earthquake (Measures et al. 2011). Conversely, salinity decreased at the $\bar{O}p\bar{a}waho/Heathcote$ River mouth, where there was considerable uplift (up to ~0.4 m; ibid.).

Dissolved oxygen increased slightly at the Ōtākaro/Avon River and South New Brighton Park sites, but decreased somewhat at Humphreys Drive and Beachville Road sites. There were large, temporary DO decreases at Ōtākaro/Avon and Ōpāwaho/Heathcote river sites immediately following the February 2011 earthquake, when raw wastewater overflows entered the rivers. Because of the range of factors that may affect dissolved oxygen in surface waters the causes for these changes are difficult to establish with confidence. Changes in primary production rates (for example, due to high production rates of phytoplankton and macroalgae), reductions in volatile suspended sediment supply to the estuary, and changes in mixing of ocean and river water may all contribute to the observed changes.

Table 3-3:Nutrient and chlorophyll-a changes before and after diversion of treated wastewater dischargefor river (blue), estuary (teal) and coastal sites (green). Data used are from 3-year periods before wastewaterdiversion (1 March 2007 to 1 March 2010) and after diversion and emergency discharges of untreatedwastewater (1 October 2011 to 1 October 2014). Boxplots show the spread of data, arrows show direction ofchange (yellow arrows indicate changes were not statistically significant), percentage is difference betweenmedian concentrations and data below plots are median concentrations (mg/L) for each 3-year period.

| Site | NH4-N | NOx-N | DRP | TN | ТР | Chlorophyll-a |
|---------------|--------------|-------------|---------------|-----------------|---------------|---------------|
| Ōtākaro/Avon | -36% | -20% | -31% | -25% | 4 -33% | -61% |
| River | | | | | | |
| | · · · · · · | | | | | |
| | 0.11 0.070 | 0.90 0.72 | 0.045 0.031 | 1.4 1.0 | 0.091 0.061 | 2.7 1.0 |
| Ōpāwaho/ | -63% | <u>17%</u> | -64% | ↔ 7% | 45% | 41% |
| Heathcote | <u> </u> | | | | | |
| River | T | | ' + | \top \top | | |
| | 0.28 0.10 | 0.94 1.1 | 0.095 0.034 | 1.5 1.6 | 0.17 0.094 | 2.2 1.3 |
| South New | -89% | -52% | .81% | -49% | <u>-62</u> % | -27% |
| Brighton Park | <u>-</u> | | T . | | | |
| | | · · — | | | | $\neg \neg =$ |
| | 0.60 0.070 | 0.47 0.22 | 0.13 0.025 | 1.3 0.66 | 0.17 0.065 | 1.4 1.0 |
| Humphreys | -79% | -24% | .75% | -36% | -62% | -72% |
| Drive | | | · · · | | | |
| | | | _ | ' T | | |
| | 0.42 0.090 | 0.33 0.25 | 0.11 0.027 | 1.0 0.64 | 0.17 0.065 | 4.3 1.2 |
| Sandy Point | -86% | -8% | -80% | -52% | -79% | -81% |
| | | | | | | |
| | | | | | | |
| | 1.1 0.15 | 0.25 0.23 | 0.18 0.034 | 1.6 0.80 | 0.40 0.081 | 3.8 0.75 |
| Penguin | -89% | -79% | -78% | -57% | -74% | -61% |
| Sileei | | | ' _ | | | |
| | 0.27 0.029 | 0.13 0.027 | 0.081.0.018 | | | 2.0 0.77 |
| Poachvillo | 0.27 0.029 | 6.15 0.027 | 670/ | 210/ | 560/ | 2.0 0.17 |
| Road | -00% | -00% | -07% | -31% | -50% | -31% |
| Noud | | | | | | |
| | 0.13 0.016 | 0.051 0.018 | 0.034 0.011 | 0.26 0.18 | 0.054 0.024 | 0.95 0.65 |
| Shag Rock low | | | | -75% | | |
| | • | | \rightarrow | | \rightarrow | |
| | _ | | | ' 📥 | | |
| | 0.86 0.061 | 0.22 0.13 | 0.18 0.025 | 1.4 0.34 | 0.22 0.045 | 3.0 0.50 |
| Shag Rock | -82% | -33% | -70% | ↔ 0% | -45% | -71% |
| high | | | | | ÷ _ | |
| | | | | | | |
| | 0.066 0.012 | 0.026 0.018 | 0.024 0.0072 | 0.20 0.20 | 0.047 0.026 | 3.4 1.0 |
| Cave Rock | -87% | -36% | -66% | ; 5% | -36% | 44% |
| | | | | | i i i | |
| | | | · · | | | - - |
| | 0.058 0.0076 | 0.025 0.016 | 0.022 0.0075 | 0.21 0.22 | 0.049 0.031 | 3.0 1.6 |
| Southshore | -91% | ↔ -42% | -63% | -8% | -37% | + -18% |
| Beach | | | | | | |
| | | | | \top \top | | |
| | 0.053 0.0045 | 0.029 0.016 | 0.014 0.0050 | 0.25 0.23 | 0.059 0.037 | 4.6 3.8 |

Table 3-4:Changes in suspended solids, turbidity, dissolved organic carbon and bacteria before and after
diversion of treated wastewater discharge for river (blue), estuary (teal) and coastal sites (green). Data used
are from 3-year periods before wastewater diversion (1 March 2007 to 1 March 2010) and after diversion and
emergency discharges of untreated wastewater (1 October 2011 to 1 October 2014). Boxplots show the spread
of data, arrows show direction of change (yellow arrows indicate changes were not statistically significant),
percentage is difference between median concentrations and data below plots are median concentrations for
each 3-year period.

| Site | TSS (mg/L) | VSS (mg/L) | Turbidity (NTU) | DOC (mg/L) | Enterococci (MPN/100mL) |
|-----------------------------|------------|-------------|------------------|-----------------|----------------------------|
| Ōtākaro/Avon | + 10% | + -13% | + 15% | + 2% | -57% |
| River | | | | | |
| | | | | | |
| | 15 16 | 2.9 2.5 | 7.6 8.7 | 2.0 2.0 | 46 20 |
| Opāwaho/ Heathcote River | + 6% | <u>→</u> 0% | 1 26% | | -41% |
| | | | | | |
| | 42 44 | 48 48 | 21 26 | 18 22 | 61 36 |
| South New | ↔ 20% | +.e +.e | 41% | ↔ 23% | |
| Brighton Park | . 📕 | | | | |
| | | | | | |
| | 15 18 | 2.7 2.7 | 6.4 9.0 | 1.3 1.6 | 14 15 |
| Humphreys | ↔ 38% | -35% | 103% | ↔ 4% | + 29% |
| Drive | | | | 📥 💻 | |
| | | | | | |
| Sandy Point | 20 28 | 4.3 2.8 | 7.0 14. | 1.2 1.2 | 24 31 |
| Sandy Fornt | -3 % | -30 % | -43 % | | |
| | | | | | |
| | 32 32 | 5.2 3.3 | 16. 9.1 | 1.5 1.5 | 17 20 |
| Penguin Street | -40% | -42% | -22% | + 14% | -67% |
| | | | | | |
| | | | | | |
| | 15 9 | 3.1 1.8 | 4.9 3.8 | 1.1 1.2 | 9 3 |
| Beachville Road | -42% | -49% | -44% | 33% | -20% |
| | | | | | |
| | 18 10 | 28 15 | 51 28 | 0.60 0.80 | 2 16 |
| Shag Rock low | ↔ -6% | -36% | | ↔ 10% | |
| SQ30546 | | | | | |
| | | | | | |
| | 24 22 | 3.4 2.2 | 8.7 9.0 | 1.0 1.1 | 4 3.6 |
| Shag Rock high | -66% | -58% | -38% | 80% | -60% |
| | | | | | — — |
| | 37 12 | 4.7 2.0 | 7.2 4.4 | 0.50 0.90 | 8 3.2 |
| Cave Rock | -57% | -53% | -29% | 1 33% | + 24% |
| | | | | | |
| | | | | | |
| Southshara | 80 35 | 5.8 2.7 | 8.0 5.8 | 0.60 0.80 | 3 3.7 |
| Beach | -/1% | -44% | -59% | | -30% |
| | | | | F | |
| | 160 45 | 6.2 3.4 | 16. 6.6 | 0.55 0.85 | 3 2.1 |

Table 3-5:Changes in salinity, dissolved oxygen and pH before and after diversion of treated wastewaterdischarge for river (blue), estuary (teal) and coastal sites (green). Data used are from 3-year periods beforewastewater diversion (1 March 2007 to 1 March 2010) and after diversion and emergency discharges ofuntreated wastewater (1 October 2011 to 1 October 2014). Boxplots show the spread of data, arrows showdirection of change (yellow arrows indicate changes were not statistically significant), percentage is differencein median concentrations and data below plots are median concentrations for each 3-year period.

| Site | Salinity (‰) | DO (mg/L) | DO (%) | рН | |
|-------------------------|-----------------|------------------|--------------------|------------|--|
| Ōtākaro/Avon River | 145% | 1 9% | 1 8% | 1 % | |
| | | | | | |
| | 1.7 4.2 | 8.8 9.6 | 86 93 | 7.7 7.8 | |
| Ōpāwaho/Heathcote River | -49% | ↔ 5% | → -3% | 1% | |
| | | | | | |
| | 88 45 | 81 85 | | 77 78 | |
| South New Brighton Park | 12% | | 1 9% | 1% | |
| South New Brighton Funk | - <u>-</u> | 📥 🚞 | - <u>-</u> | <u> </u> | |
| | | | | | |
| | 20 23 | 7.4 8.0 | 82 89 -14% | 7.8 7.9 | |
| Humphreys Drive | | | | | |
| | T T | | ₽ | | |
| | 25 25 | 9.4 8.3 | 104 89 | 8.0 8.0 | |
| Sandy Point | | | -4% | ↔ 0% | |
| | | | | | |
| | 25 26 | 8.2 8.4 | 95 91 | 7.9 7.9 | |
| Penguin Street | <u></u> 3% | -4% | <mark>↔</mark> -3% | <↔ 0% | |
| | | - - | | | |
| | 30 31 | 92 88 | 102 99 | 80 80 | |
| Beachville Road | 1 3% | -9% | -10% | ↔ 0% | |
| | = | <u> </u> | | | |
| | T | T T | | | |
| | 32 33 | 9.2 8.4 | 109 98 | 8.1 8.1 | |
| Shag Rock low | | -1% | -4% | - 0% | |
| | | | | | |
| | 26 28 | 8.9 8.8 | 103 99 | 8.0 8.0 | |
| Shag Rock high | 1% | ↔ 0% | -3% | ↔ 0% | |
| | | | | — — | |
| | 33 33 | 8.6 8.6 | 102 100 | 8.0 8.0 | |
| Cave Rock | + 1% | ++ 5% | , 1 % | ↔ 1% | |
| | | | | | |
| | | — — | T 1 | | |
| | 33 33 | 8.2 8.6 | 100 101 | 8.0 8.1 | |
| Southshore Beach | | $-\frac{2\%}{1}$ | -1% | | |
| | | | | | |
| | 32 33 | 8.4 8.3 | 101 100 | 8.0 8.1 | |

3.4 Changes since 2014 have been more variable

In this section we focus on the latest six-year period, from January 2014 to December 2019. This period is after the diversion of the treated wastewater discharge, and after the untreated wastewater discharges related to the Christchurch earthquakes. The changes over time during this period are more subtle and for many variables are difficult to discern by eye (e.g., using time-series plots such as Figure 3-1 to Figure 3-8) due to seasonal variation or high proportions of censored data. All data are shown in Appendix D (Figure D-1to Figure D-21) for all the variables plotted over time with trend lines overlaid.

The most consistent trends (that is, occurring at multiple sites, Figure 3-8, Table 3-6) are for **turbidity** and total nitrogen which are decreasing over time and for **chlorophyll**-*a* and **enterococci** which are likely to be increasing over time. The **water temperature** and **pH** are also increasing over time at most sites, however as discussed in section 3.1, this does not necessarily constitute degradation or improvement. **Chlorophyll**-*a* has increased at all sites with the possible exceptions of Beachville Road, Shag Rock and Southshore Beach where an increase is considered "likely". The largest increases have occurred at Humphreys Drive, South New Brighton Park and Sandy Point, each site showing an increase at least 20% per year. Nutrient concentrations at these sites do not show any increase over time, and in fact there are likely decreases in most forms of nitrogen and phosphorus at these sites.

Total nitrogen has decreased during 2014-2019 at all sites except $\bar{O}p\bar{a}$ waho/Heathcote River, Humphreys Drive and South New Brighton Park. The largest decreases were at Sandy Point, Shag Rock (high) and Southshore Beach, where TN decreased by about 5% per year. At other sites the decrease was 2.7-4.7% per year. Such decreases were not observed for TP which showed little change at most sites, except at Shag Rock during low tide (4% per year).



Figure 3-8: Summary plot representing for each water quality measurement the proportion of sites with trends at each categorical level of confidence. Trends assessed over 6-year time period from January 2014 to December 2019. The plot shows the proportion of sites with trends at levels of confidence defined in Table 3-1.

Table 3-6: Summary of trend directions (arrows) and magnitude (percentage change per year) for each water quality variable for the period January 2014 to December 2019. The direction of the arrow indicates the direction of the trend (e.g., up arrow means concentrations are increasing, down arrow means concentrations are decreasing). The colour of the arrow refers to the categorical level of confidence in the trend: for decreasing trends (improving water quality), dark green signifies a virtually certain decrease in concentration and light green a likely decrease, while for increasing trends (degrading water quality), dark red signifies a virtually certain increase in concentration and orange a likely increase. Horizontal arrows indicate concentrations are as likely to be increasing as decreasing (no clear trend). Black / grey arrows are used for salinity, water temperature and pH as trend directions for these variables do not indicate improvement in ecosystem health. The level of confidence in the increases or decreases in concentration are shown as a gradient from black (virtually certain) to light grey (likely).

| | Ōtākaro / Avon River | Ōpāwaho / Heathcote River | South New Brighton Park | Humphreys Drive | Sandy Point | Penguin Street | Beachville Road | Shag Rock Iow | Shag Rock high | Cave Rock | Southshore Beach |
|---------------------------------|-------------------------|---------------------------------|-------------------------------|---------------------|---------------------|---------------------|--------------------|---------------------|---------------------|--------------|---------------------|
| Salinity | ➡ -14 | - 2.2 | ⇔ -0.2 | ₽ -2 | 1 | 10.3 | ⇔ 0.1 | - 0.8 | -0.1 | 10.1 | ⇔ 0.1 |
| Water temperature | 10.6 | 10.2 | ⇔ 0.7 | 10.6 | 1.2 | 1 .9 | 1 0.5 | ⇔ 0 | 1 0.7 | 10.6 | 10.7 |
| Dissolved Oxygen saturation (%) | -1 | -0.8 | ⇔ -0.1 | ₽ -2.2 | 1 .2 | 1 3.7 | 10.4 | 1 0.9 | ♠ 0.5 | 10.3 | 1 0.5 |
| рН | -0.4 | - 0.3 | 1 0.2 | 1 0.2 | 1 0.3 | 1 0.7 | 1 0.2 | 1 0.5 | 1 0.3 | 1 0.3 | 1 0.2 |
| Ammoniacal-N | + -4.8 | - 12 | + -6.1 | - 18 | <mark>↓</mark> -5 | - 0.9 | 1 6.4 | ↓ -3.6 | † 5.6 | NA | 1 NC |
| Nitrate-N + nitrite-N (NOx-N) | - 5.6 | ↓ -3.5 | 1 3.5 | <mark>↓</mark> -4.2 | -14 | - 9.8 | ⇔ NC | <mark>↓</mark> -2.3 | <mark>↓</mark> -5 | + -2.8 | ⇔ NC |
| DIN | ↓ -6.2 | ↓ -3.4 | ⇔ 1.6 | -6.8 | - 9.6 | - 5.7 | <u></u> | <mark>↓</mark> -3 | <u></u> 1.9 | ⇔ 2.8 | † 6.4 |
| Total Nitrogen | ↓ -3.5 | <mark>↓</mark> -1.2 | ⇔ 0 | + -3.9 | - 5.2 | ↓ -3.3 | - 2.8 | - 2.7 | ↓ -5.1 | -4.7 | - 5.2 |
| Dissolved Reactive Phosphorus | 1 3.2 | ⇔ 0 | <mark>↓</mark> -1.5 | <mark>↓</mark> -4.3 | <mark>↓</mark> -4.6 | ₽-8.2 | <mark>↓</mark> -1 | - 4.5 | <mark>↓</mark> -1.2 | -6.8 | + -3.3 |
| Total Phosphorus | <u></u> 1.8 | ⇔ 0 | 1 | ⇔ 0 | ⇔ 0 | <mark>↓</mark> -1.2 | ⇔ 1.6 | - 4 | + -1.3 | ⇔ 0 | + -3.5 |
| Chlorophyll-a | 1 8.9 | 1 9.3 | 1 23 | 1 26 | 1 20 | 12 | <u></u> | <u>†</u> 6 | <u></u> 4.2 | 1 9.7 | <u></u> 2.4 |
| Total Suspended Solids | + -2.8 | -6.8 | ⇔ 0 | -13 | ⇔ 0 | 1 8.7 | <u></u> 3.1 | - 5 | 11 | 14 | † 6.9 |
| Turbidity | - 4.8 | -6.3 | ↓ -3.4 | -15 | - 9 | ⇔ 0.6 | - 4.9 | -10 | -6.5 | -7.9 | + -3.8 |
| Enterococci | 13 | † 6.9 | 1 NC | 14 | 1 NC | ⇔ NC | 1 NC | 1 NC | 1 NC | 1 NC | 1 NC |
| E. coli | - 7.6 | ⇔ -1.3 | <u></u> | 16 | <u></u> 4.1 | NA | NM | <u> </u> 0 | NM | NM | NM |
| Faecal coliforms | NM | NM | NM | NM | ⇔ -0.6 | - 4.6 | ⇔ 0 | ⇔ 1.5 | + -6.1 | NM | NM |
| Total copper * | ↓ NC | -8.0 | ⇔ NC | ₽ NC | ⇔ NC | ➡ NC | NC | NC | NC | ↓ NC | ↓ -2.1 |
| Total chromium * | 1 NC | <u></u> 2.8 | 1 29 | 1 23 | <u></u> 21 | <u></u> | 1 30 | ⇔ NC | 1 28 | 13 | <u></u> 21 |
| Total lead * | ↓ NC | 1 9.6 | ➡ NC | NC | 1 NC | ↓ NC | NC | NC | NC | 1 NC | 1 NC |
| Total zinc * | + -6.7 | ⇔ 0.5 | 1 NC | 1 NC | 1 NC | + NC | NC | NC | NC | 1 NC | ⇔ NC |

NC: Trend magnitude could not be calculated due to high proportion of censored data. NM: not measured at that particular site. NA: trend could not be assessed due to high proportions of censored data. * Trends for metals were assessed from the beginning of bimonthly monitoring (August 2016) to December 2019.

DIN has decreased at most sites near the rivers and mid-estuary, but possibly increased at coastal sites and at the estuary mouth. The decreases in DIN at the river and mid-estuary sites is due to changes in both **ammoniacal-N**, which showed strong decreases at Ōpāwaho/Heathcote River mouth and Humphreys Drive, and **NOx-N**, which decreased at Ōtākaro/Avon River, Sandy Point and Penguin Street.

Enterococci counts have increased throughout the estuary including at the Ōtākaro/Avon River mouth and at the coastal sites. However, because of high proportions of censored data, the magnitude of the trend could only be calculated for three sites (Ōtākaro/Avon River, Ōpāwaho/Heathcote River and Humphreys Drive, which showed increases of 13%, 7% and 14% per year, respectively, equating to increases of 2-3 MPN/100mL each year. *E. coli*, which is measured at seven sites, is likely to have increased at most sites, except the Ōtākaro/Avon River, where concentrations have decreased – opposite to the trend for Enterococci. Faecal coliforms, the third faecal indicator bacteria monitored, showed a likely decrease at the Penguin Street site and at Shag Rock during high tide, but no clear trends occurred at the other three sites monitored (Beachville Road, Sandy Point and Shag Rock during low tide).

Water temperature also shows increases over time at most sites, although these trends are not highly likely, probably due to the high variability in water temperature and the short time frame assessed for trends. Higher water temperatures and / or the decrease in turbidity (increase in clarity) may contribute to growth of phytoplankton (increases in chlorophyll-*a*). We suggest that the current ECan strategy of maintaining a consistent time of day for sampling is vital to interpret trends in water temperature.

The **pH** of the water has increased at several sites, except the two river sites where it is very likely or extremely likely to have decreased. The increase in pH may be associated with increased chlorophyll*a* (increased phytoplankton abundance) and dissolved oxygen (at most sites). These indicate increased phytoplankton photosynthesis, which removes carbon dioxide from the water, along with its dissolved form (carbonic acid), making the water less acidic (more alkaline / higher pH). Another possible reason for the increased pH and oxygen could be the observed proliferation of macroalgae in the estuary in recent years.

TSS increased at three sites (Penguin Street, Shag Rock high, Cave Rock), but **turbidity** did not increase at these sites and in fact decreased at the Shag Rock and Cave Rock sites (Table 3-6). The TSS trend appears to be due to a number of higher (>100 mg/L) TSS values measured at these sites. Further monitoring will show whether this trend continues over time or is just due to a few outliers. Turbidity is very likely to have decreased at all sites except Southshore Beach and Penguin Street.

Most of the **metal** results are below the laboratory detection limits, particularly for dissolved metals, and total cadmium and nickel so changes over time were not assessed for these metals. Only the total forms of chromium, copper, lead and zinc have been measurable in at least 20% of the samples. **Total chromium** appears to have increased at South New Brighton Park, Humphreys Drive, Beachville Road and Shag Rock during high tide. Other metals show less change: **total copper** has likely decreased in the Ōpāwaho/Heathcote River, while **total lead** has likely increased in the Ōpāwaho/Heathcote River, while **total lead** has likely increased in the Ōtākaro/Avon River. At most sites, only the direction of change can be calculated: the magnitude of the change is not quantifiable due to the many data below detection limits.

3.5 Nutrients and faecal indicator bacteria can vary seasonally

Some aspects of water quality vary seasonally. Water temperature (as shown in Figure D-1), is one example with higher temperatures in summer and lower temperatures in winter. Dissolved nutrient concentrations (ammoniacal-N, NOx-N and DRP, averaged monthly from 2013 to 2019) vary seasonally at many of the estuary sites and in the two main rivers entering the estuary (Figure 3-9 and Figure 3-10). NOx-N concentrations are highest in the winter and lowest in summer, as are ammoniacal-N and total nitrogen (not shown), though these vary to a lesser extent at many sites. However, DRP concentrations (and to some extent TP) show a different seasonal pattern at the river sites when compared to the coastal sites (Figure 3-11). In the rivers, DRP is highest in autumn and lowest in early spring; whereas in the coastal sites, DRP is highest in winter and lowest in summer. The variation at the sites in the estuary reflect their proximity to the estuary mouth or to the rivers.

Patterns of nutrient concentration follow seasonal patterns largely due to demand from primary producers such as macroalgae and phytoplankton. In winter when growth of these primary producers is limited by the availability of light and low temperatures, water column nutrient concentrations are generally higher. In spring and summer algae take up nutrients from water at higher rates to meet their demand for growth. This process can reduce concentrations of the nutrients down to very low levels. Typically, availability of dissolved inorganic nitrogen (NOx-N or ammoniacal-N) limits primary production in estuaries and coastal waters (Barr 2007, Barr et al. 2013), while phosphorus availability is most limiting to algal growth in Canterbury rivers (Larned et al. 2011). The seasonal patterns of nutrient availability shown in Figure 3-11 are strongly indicative of nitrogen limitation in the saline parts of this estuary, with concentrations reduced to low levels in summer.

There were three sites where seasonal variation in either NOx-N or DRP was not statistically significant (Figure 3-9 and Figure 3-10). At Humphreys Drive, NOx-N was not statistically significant due to the presence of a few high concentrations (outliers), but there does also appear to be a seasonal pattern with higher concentrations in winter. NOx-N concentrations at the Ōtākaro/Avon River mouth show less variation with relatively high concentrations in summer as well as winter. This may be due to low variation in the sources of NOx-N at this site. Finally at Sandy Point, variation in DRP concentrations was not statistically significant. DRP was high at this site compared to all other locations and it is possible that either 1) the source of DRP at this site does not vary seasonally or 2) uptake of DRP by algae does not reduce DRP to the same extent as at other sites.

Faecal indicator bacteria can also demonstrate seasonality in their concentrations in river, estuary and beach environments. Enterococci concentrations did not vary in the estuary or coastal sites, or at Ōtākaro/Avon River, but did vary at Ōpāwaho/Heathcote River (Figure 3-12). The variation by season (months) at this site does not follow any regular pattern, like water temperature or nutrients, and is mainly characterised by very high concentrations in July. On the other hand, *E. coli* concentrations showed more of a seasonal pattern, with lower concentrations in winter/spring, and higher concentrations in summer/autumn (Figure 3-13). This was most apparent at the estuarine sites, rather than the two river mouth sites, which had higher concentrations in July (comparable with the results for enterococci). Higher faecal indicator bacteria concentrations in summer and autumn can be due to additional sources or to bacterial growth in sediments, sands and algal mats.



Figure 3-9: Seasonal variations in NOx-N at all sites. Kruskal Wallis test indicates seasonal differences for all sites except Ōtākaro/Avon River and Humphreys Drive (p-values > 0.05). White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles, and maximum values are shown by dots. Note y-axis scale varies between sites.



Figure 3-10: Seasonal variations in DRP at all sites. Kruskal Wallis test indicates seasonal differences for all sites except Sandy Point. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles, and maximum values are shown by dots. Note y-axis scale varies between sites.



Figure 3-11: Comparison of the seasonal variation in monthly means of NOx-N and DRP at all sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note y-axis scales differ for each site.


Figure 3-12: Seasonal variations in enterococci at all sites. Kruskal Wallis test indicates seasonal differences at only the $\bar{O}p\bar{a}$ waho/Heathcote River site. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles, and maximum values are shown by dots. Note y-axis scale varies between sites.



Figure 3-13: Seasonal variations in *E. coli* at the eight sites regularly monitored. Kruskal Wallis test indicates seasonal differences at South New Brighton Park, Penguin Street and Sandy Point. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles, and maximum values are shown by dots. Note y-axis scale varies between sites.

3.6 Patterns in seasonality have not changed substantially

At the $\bar{O}p\bar{a}waho/Heathcote River site (Figure 3-14, left), the seasonal pattern of NOx-N in 2019 is reflective of that for 2013-2016. However, there was a lower than usual NOx-N measurement at this site in June 2017. The salinity at that time (4.8 ‰) was typical for the Heathcote River but turbidity and total suspended solids concentrations were the lowest ever measured for this site, suggesting unusual conditions at the site or a possible error or mix-up with the samples. The NOx-N concentrations were higher than usual over the 2017/18 and 2018/19 summers, suppressing the usual seasonal variation at this site.$

In Ōtākaro/Avon River (Figure 3-14, right), there was a slightly higher than usual NOx-N concentration measured in April 2018, with lower than usual concentrations measured in March and May 2018. Those two low values are amongst the lowest recorded at this site. All other measurements (salinity, turbidity and total suspended solids) at those times were typical for the site and do not indicate any reason for the lower than usual NOx-N. The NOx-N concentrations measured by Christchurch City Council (CCC) (Marshall & Noakes 2019) two to five days earlier at the Bridge Street site and further upstream do not reflect this pattern, suggesting either sampling or measurement error, or a very short-lived effect.



Figure 3-14: Seasonal variation in NOx-N at Ōpāwaho/Heathcote River (left) and Ōtākaro/Avon River (right) from 2013 to 2019. Y-axis scales are the same for each year but differ between the two sites.

The seasonal patterns for DRP at the two river sites for 2017 and 2018 (Figure 3-15) are unusual with higher than usual concentrations in January and June 2018 and July 2019 at both sites. The July 2019 sampling event was the day after a flood event and this is likely to have affected the water quality. The January 2018 sample (collected 22 January) was also after very large floods, on 5 and 11 January, though it is less likely that these were the cause of the high concentration. The sample collected June 2018 was not affected by high flow and there is no obvious cause for that concentration being higher than usual.



Figure 3-15: Seasonal variation in DRP at Ōpāwaho/Heathcote River (left) and Ōtākaro/Avon River (right) from 2013 to 2019. Y-axis scales are the same for each year but differ between the two sites.

3.7 Summary

The water quality of Ihutai/Avon-Heathcote Estuary has improved since ECan's water quality monitoring programme began in 2007. All forms of nitrogen and phosphorus have decreased at almost all sites. Total suspended solids and volatile suspended solids have also decreased. The diversion of Christchurch City's wastewater treatment plant discharge in 2010 resulted in major improvements, particularly for dissolved reactive phosphorus (DRP), ammoniacal-N and volatile suspended solids. NOx-N did not decrease substantially as the major source of this form of nitrogen is the rivers. Since 2014, there have been further decreases in concentrations of total nitrogen (and at some sites ammoniacal-N and NOx-N) and DRP at most, but not all, sites. However, chlorophyll-*a* has increased at all sites during this period. The indicator bacteria enterococci has also increased at most sites.

4 Water quality is currently poor at many sites

4.1 How do we assess the current state of water quality?

To assess the current state of water quality at the Ihutai/Avon-Heathcote Estuary we looked at data from January 2015 to December 2019. A five-year period provides a good number of data points to be robust (about 60 for monthly monitoring) and generally avoids the effect of trends over time (McBride 2005).

Censored values (those below laboratory detection limits) need to be included in the calculation of summary statistics to avoid biasing results, but they must be handled appropriately. We used the robust Regression on Order Statistical (ROS) method (described by Helsel 2012), which was also used in recent national assessments of coastal water (Dudley & Jones-Todd 2018) and freshwater quality (Larned et al. 2015, Larned et al. 2018). The ROS procedure produces estimated values for the censored data, consistent with the distribution of uncensored values, and these estimates can be used in calculating summary statistics ³. There were also several data points for indicator bacteria where results are over range because water samples were not diluted sufficiently when high concentrations were present. These data are reported as "more than a reporting limit", e.g., >24,000 MPN/100mL⁴. These right-censored data were dealt with using a statistical procedure based on "survival analysis" (Helsel 2012)⁵ to allow the calculation of summary statistics.

We calculated summary statistics and prepared plots that compare the range in concentrations between sites, and to water quality guidelines for ecological health and for human health (see sections 4.3 and 4.4 for details and rationale), where appropriate for estuarine and coastal waters.

4.2 Estuaries are where rivers meet the sea

The salinity of the water at each monitoring site can tell us about the relative influence of the freshwater inputs versus that of the seawater. The water at the river sites has lowest salinity (Figure 4-1), though still above that of freshwater, where it mixes with the estuarine waters. Cave Rock and Southshore Beach, outside the estuary, have a median salinity of 32-33 ppt, about the same as coastal seawater, and do not vary much. Other sites in the estuary are affected more by the freshwater from the two main rivers: South New Brighton Park (near to the Ōtākaro/Avon River) and Humphreys Drive and Sandy Point (near Ōpāwaho/Heathcote River) have lower salinity than Penguin Street, Beachville Road and Shag Rock. The water quality at the South New Brighton Park, Humphreys Drive and Sandy Point sites is therefore more likely to be influenced by the incoming rivers than the sites closer to the mouth.

³ To use the ROS method, a statistical distribution for uncensored data is developed, fitted to probability plotting positions by least squares regression. This relationship is then used to predict the concentrations for the censored values based on their plotting positions (the ordering of the data). The ROS procedure can accommodate multiple censoring limits, as are present in this data set where detection limits have changed over time.

⁴ There were few values like this as follows: 1 value for enterococci; 3 for faecal coliforms and 8 for *E. coli*.

⁵ A parametric distribution is fitted to the uncensored observations and then values for the censored observations are estimated by randomly sampling values larger than the censored values from the distribution.



Figure 4-1: Salinity of river, estuary and coastal monitoring sites. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots.

4.3 Water quality has potential to affect human health

Median values of faecal indicator bacteria, such as enterococci and E. coli, were highest at the two river sites (Figure 4-2) and then generally decreased from the rivers towards the estuary mouth and to Pegasus Bay (for enterococci only). On the other hand, the 95th percentile values for enterococci increased from the river sites to estuary sites, then decreased nearer to the estuary mouth and at coastal sites. For five of the sites (Penguin Street, Beachville Road, Shag Rock during low tide, Cave Rock and Southshore Beach), more than half of the enterococci data were below the detection limit of 10 MPN/100mL.

The Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas ('the guidelines' (MfE/MoH 2003)) recommend measurement of enterococci in marine waters, to assess risks of illness from swimming and other contact recreation activities. They recommend measurement of *E. coli* in freshwater. However, it is not clear from the guidelines which indicator (enterococci or *E. coli*) should be used in brackish or estuarine waters, hence both indicators have been measured at the brackish sites in the HERC sampling to date.⁶

A comparison of the five-year 95th percentile enterococci values against the Microbiological Assessment Category (MAC) definitions for marine waters in the Guidelines (Figure 4-2), the

⁶ A review of brackish water indicators carried out for the regional sector's Coastal Special Interest Group recommended for long residence-time estuaries (greater than three days), enterococci should be chosen. For short residence-time estuaries, *E. coli* is the appropriate choice when near the inflowing river water, but enterococci should be chosen near the mouth. Between these locations, either indicator may be suitable. Accordingly, it appears wise to measure both indicators in low residence time systems and use the more stringent of the two test results for surveillance (McBride et al. 2019)(McBride et al. 2019).

Öpāwaho/Heathcote River mouth site, South New Brighton Park, Humphreys Drive and Sandy Point would be graded in the D (red) category – the poorest category. This category corresponds to (maximum-average) HCGI (highly credible gastrointestinal illness) risks of ≥10%, and ≥3.9% risk of AFRI (Acute Febrile Respiratory Illness) (McBride et al. 2019), indicating significant risks to human health to people swimming or carrying out other forms of contact recreation at these locations. The Ōtākaro/Avon River and Beachville Road sites would be graded in the C (orange) category, and the remaining sites would be graded in the B (yellow) category – indicating lower HCGI and AFRI illness risks at these sites.⁷ Based on comparison of the 95th percentile of *E. coli* data to the MAC definitions for freshwaters in the Guidelines, the Ōtākaro/Avon River and Ōpāwaho/Heathcote River mouth sites, South New Brighton Park, Humphreys Drive and Sandy Point sites were in the D category, Penguin Street was in the C category and Shag Rock low was in the B category. The D category indicates higher gastrointestinal infection risks for people using these waters.⁸

This grading includes data collected over the whole year, rather than the summer bathing season that is generally used by Council and the Land Air Water Aotearoa (LAWA) web site⁹, to assist recreational users in deciding whether it is safe to swim. The gradings are likely to be different as there are frequently higher concentrations of faecal indicator bacteria measured in summer (see section 3.5).



Figure 4-2: Enterococci and *E. coli* counts at monitoring sites. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Lines and boxes below detection limit line are based on estimates only. Shading represents Microbiological Assessment Category definitions from MfE / MoH (2003) for marine (enterococci) and freshwaters (*E. coli*) respectively. Median values of < 10 MPN/100mL are not shown.

⁷ The three boundaries between the four bands (A to D) are assessed as sample 95% iles correspond to (maximum-average) HCGI risks of 1%, 5% and 10%, and 0.3%, 1.9% and 3.9% risk of AFRI (McBride et al .2019). See MfE/MoH (2003), Table H1 for more details.

⁸ Note that the risks to human health are calculated differently for fresh waters compared with marine waters. See MfE/MoH (2003).

⁹ lawa.org.nz

Table 4-1:Summary of indicator bacteria counts and suitability for swimming and other forms of contactrecreation.Red values indicate exceedance of category D ratings of the Microbiological Water QualityGuidelines for Marine and Freshwater Recreational Areas.

| | Enterococci | (MPN/100mL) | E. coli (MPN/100mL) | | Bacterial risk for | |
|-------------------------|-------------------------|--|-------------------------|--|--|--|
| Site | Median concentration | 95 th percentile concentrations ¹ | Median concentration | 95 th percentile concentrations ¹ | swimming and other recreation ² | |
| Avon River | 20 | 403 | 207 | 1,522 | High | |
| Heathcote River | 31 | 1,210 | 209 | 3,103 | High | |
| South New Brighton Park | 15 | 855 | 135 <mark>840</mark> | | High | |
| Humphreys Drive | 20 | 5,264 | 74 702 | | High | |
| Sandy Point | 15 | 7,095 | 107 <mark>8,963</mark> | | High | |
| Penguin Street | <10 | 133 | 20 314 | | Moderate | |
| Beachville Road | <10 | 485 | 20 192 | | Moderate | |
| Shag Rock low | <10 | 192 | 135 <mark>840</mark> | | High | |
| Shag Rock high | 10 | 126 | Not measured | | Low | |
| Cave Rock | <10 | 92 | Not measured | | Low | |
| Southshore Beach | <10 | 51 | Not measured | | Low | |

Note: ¹ Percentiles calculated using Hazen formula, as required by MfE/MoH (2003). ² Based on compliance for both enterococci and *E. coli* guidelines. 3. Based on MAC definitions, see also LAWA¹⁰. High risk indicates >10% risk of GI illness or >5% risk of campylobacter infection; moderate indicates >5% risk of GI illness and 1-5% risk of campylobacter infection; low risk indicates <5% risk of GI illness and <1% risk of campylobacter infection.

Overall, when both indicators are considered (Table 4-1), sites at the river mouths and in the inner estuary have high risks for bacterial infection; whereas sites in the outer estuary have moderate risk (except at low tide, when risks are higher) and sites on the coast have low risk.

Faecal coliform bacteria are used to assess the suitability for shellfish gathering. Based on the data for the five-year period January 2015 to December 2019 (Figure 4-3, Table 4-2), there is a risk for shellfish gathering at three of the locations monitored as either the median concentration exceeds 14 MPN/100 mL or more than 10% of samples exceeds 43 MPN/100 mL (MfE / MoH 2003), or both.¹¹ At Beachville Road, the median is <10, and only 9% of samples exceeded the upper guideline, suggesting shellfish will be safe to eat at this location. At the Shag Rock site, the water is suitable when based on high tide measurements alone, but not on low tide measurements or high and low tide measurements in combination.

¹⁰ https://www.lawa.org.nz/learn/factsheets/what-do-the-swim-icons-mean/

¹¹ The MfE/MoH (2003() guidelines are intended to apply to a shellfish gathering season but a season is not defined in the Guidelines. McBride et al. (2019) suggest that the guidelines should be revised, with the 90th percentile value replaced as a "No sample shall exceed" guideline. McBride et al. (2019) also make a case to consider guidelines based on enterococci, with a requirement that the median is less than 7 enterococci per 100 mL and the maximum does not exceed 22 enterococci per 100 mL.



Figure 4-3: Faecal coliform concentrations at estuary monitoring sites. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Dotted lines indicate that the 5th percentile was based on estimated data (below the detection limit). The upper guideline should not be exceeded by more than 10% of samples and the lower line (14 cfu/100mL) should be compared to median concentrations.

| Site | Median | Percentage samples exceeding guideline | 90 th percentile | Suitability for shellfish gathering | |
|-----------------------|--------|--|-----------------------------|--|--|
| MfE/MoH guideline | 14 | 43 | | | |
| Sandy Point | 80 | 64 | 1064 | × | |
| Penguin Street | 14 | 24 | 210 | × | |
| Beachville Road | <10 | 9 | 39 | \checkmark | |
| Shag Rock (low tide) | 21 | 29 | 141 | × | |
| Shag Rock (high tide) | <10 | 10 | 48 | \checkmark | |
| Shag Rock (overall) | 11 | 22 | 64 | × | |

Table 4-2:Summary of faecal coliform concentrations and suitability for shellfish gathering. Red valuesindicate exceedance of the recreational shellfish gathering guidelines.

4.4 Water quality has potential to affect ecological health

The ecological state of the estuary is influenced by water temperature, nutrient concentrations, turbidity, sediment concentrations, dissolved oxygen and contaminants like metals. In this section we assess possible effects on ecology by comparing the water quality measured over 2014-2019 to a number of different guidelines. The values used (Table 4-3) are different for the river sites, estuary sites and the coastal sites because of the different waters, different types of organisms present and the methods used to derive the guidelines. Where available, we have used standards, limits and guidelines from Canterbury's regional plans (Environment Canterbury 2011, Environment Canterbury 2012, Environment Canterbury 2017) or developed specifically for the Canterbury region (Dudley et al. 2019, Stevenson et al. 2010). Where regional guidelines were not available, we have used national guidelines (ANZECC 2000, ANZG 2018, Plew et al. 2018a) or those developed for coastal and estuarine waters in other regions (Foley 2018, Griffiths 2016), as previously used in water quality reports on Ihutai/Avon-Heathcote Estuary (Bolton-Ritchie (2019).

Many of the guideline values are statistically-derived values from monitoring data (e.g., the Australia and New Zealand Guidelines (ANZG) and Australian and New Zealand Environment and Conservation Council (ANZECC) 2000 guidelines for nutrients in estuarine waters). These provide a useful comparison to assess where concentrations are above those expected for these types of waters. However, where available, we suggest use of biologically-based guideline values, that indicate the potential for effects (e.g., excess algal growth, toxicity) when guidelines are exceeded.

| Dissolved oxygen (% saturation)Estuary and coastal Rivers> 80%New Zealand Government (1991)DINRivers> 70%Environment Canterbury (2011)DINRivers1.5 mg/LEnvironment Canterbury (2017)Ammoniacal-NRivers0.010 mg/LANZG (2018)Estuary0.015 mg/LANZECC (2000); Foley (2018)Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017) | Water quality measurement | Environment | Threshold (must be less than, unless specified) | Source of guideline |
|--|------------------------------|---------------------|---|-------------------------------------|
| (% saturation)Rivers> 70%Environment Canterbury (2011)DINRivers1.5 mg/LEnvironment Canterbury (2017)Ammoniacal-NRivers0.010 mg/LANZEG (2018)Estuary0.015 mg/LANZECC (2000); Foley (2018)Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017) | Dissolved oxygen | Estuary and coastal | > 80% | New Zealand Government (1991) |
| DINRivers1.5 mg/LEnvironment Canterbury (2017)Ammoniacal-NRivers0.010 mg/LANZG (2018)Estuary0.015 mg/LANZECC (2000); Foley (2018)Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017) | (% saturation) | Rivers | > 70% | Environment Canterbury (2011) |
| Ammoniacal-NRivers0.010 mg/LANZG (2018)Estuary0.015 mg/LANZECC (2000); Foley (2018)Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018)Coastal0.31 mg/LDudley et al. (2018)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnvironment Canterbury (2011; 2017) | DIN | Rivers | 1.5 mg/L | Environment Canterbury (2017) |
| Estuary0.015 mg/LANZECC (2000); Foley (2018)Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnvironment Canterbury (2011; 2017) | Ammoniacal-N | Rivers | 0.010 mg/L | ANZG (2018) |
| Coastal0.035 mg/LDudley et al. (2019)NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnvironment Canterbury (2018) | | Estuary | 0.015 mg/L | ANZECC (2000); Foley (2018) |
| NOx-NRivers0.265 mg/LANZG (2018)Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnvironment Canterbury (2011; 2017) | | Coastal | 0.035 mg/L | Dudley et al. (2019) |
| Estuary0.048 mg/LGriffiths (2016)Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnvironment Canterbury (2018) | NOx-N | Rivers | 0.265 mg/L | ANZG (2018) |
| Coastal0.064 mg/LDudley et al. (2019)TNRivers0.913 mg/LANZG (2018)Estuary0.08 mg/LPlew et al. (2018a)Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEnlow (2018) | | Estuary | 0.048 mg/L | Griffiths (2016) |
| TN Rivers 0.913 mg/L ANZG (2018) Estuary 0.08 mg/L Plew et al. (2018a) Coastal 0.31 mg/L Dudley et al. (2019) DRP Rivers 0.016 mg/L Environment Canterbury (2011; 2017) Estuary 0.021 mg/L Enlow (2018) | | Coastal | 0.064 mg/L | Dudley et al. (2019) |
| Estuary 0.08 mg/L Plew et al. (2018a) Coastal 0.31 mg/L Dudley et al. (2019) DRP Rivers 0.016 mg/L Environment Canterbury (2011; 2017) Estuary 0.021 mg/L Environment Canterbury (2018) | TN | Rivers | 0.913 mg/L | ANZG (2018) |
| Coastal0.31 mg/LDudley et al. (2019)DRPRivers0.016 mg/LEnvironment Canterbury (2011; 2017)Estuary0.021 mg/LEoloy (2018) | | Estuary | 0.08 mg/L | Plew et al. (2018a) |
| DRP Rivers 0.016 mg/L Environment Canterbury (2011; 2017) | | Coastal | 0.31 mg/L | Dudley et al. (2019) |
| Estuary 0.021 mg/l Eoloy (2018) | DRP | Rivers | 0.016 mg/L | Environment Canterbury (2011; 2017) |
| | | Estuary | 0.021 mg/L | Foley (2018) |
| Coastal 0.018 mg/L Dudley et al. (2019) | | Coastal | 0.018 mg/L | Dudley et al. (2019) |
| TP Rivers 0.014 mg/L ANZG (2018) | ТР | Rivers | 0.014 mg/L | ANZG (2018) |
| Estuary 0.030 mg/L ANZECC (2000); Griffiths (2016) | | Estuary | 0.030 mg/L | ANZECC (2000); Griffiths (2016) |
| Coastal 0.054 mg/L Dudley et al. (2019) | | Coastal | 0.054 mg/L | Dudley et al. (2019) |

Table 4-3: Water quality guidelines used to assess potential effects on ecological health.

| Water quality measurement | Environment | Threshold (must be less than, unless specified) | Source of guideline |
|---------------------------|-------------------------|---|-----------------------------|
| Chlorophyll-a | Rivers | 5 μg/L | Plew et al. (2018a) |
| | Estuary | 3 μg/L | Plew et al. (2018a) |
| | Coastal | 4.76 μg/L | Dudley et al. (2019) |
| TSS | Rivers & estuary | 25 mg/L | Stevenson et al. (2010) |
| | Coastal | 140 mg/L | Dudley et al. (2019) |
| Turbidity | Rivers | 1.3 NTU | ANZG (2018) |
| | Estuary | 10 NTU | ANZECC (2000); Foley (2018) |
| | Coastal | 21 NTU | Dudley et al. (2019) |
| Copper | Rivers | 0.0018 mg/L | ANZECC (2000) |
| | Estuary and coast | 0.0013 mg/L | ANZECC (2000) |
| Lead | Ōtākaro/Avon River | 0.015 mg/L | ANZECC (2000) |
| | Ōpāwaho/Heathcote River | 0.029 mg/L | ANZECC (2000) |
| | Estuary and coast | 0.004 mg/L | ANZECC (2000) |
| Zinc | Ōtākaro/Avon River | 0.030 mg/L | ANZECC (2000) |
| | Ōpāwaho/Heathcote River | 0.045 mg/L | ANZECC (2000) |
| | Estuary and coast | 0.015 mg/L | ANZECC (2000) |

The water temperature (Figure 4-4) is fairly similar amongst the estuary sites (medians 13-15°C) but were generally a little higher at the river sites and during low tide at Shag Rock. There is more variation over the 2014-2019 sampling period than between sites, as temperatures change with season and climate.

Dissolved oxygen saturation was consistently above 80%, the threshold used for protection of aquatic invertebrates and fish, at all sites except occasionally at the Humphreys Drive site (Figure 4-4). At the two river sites, dissolved oxygen saturation was above the guideline of 70% on almost all occasions measured. Low dissolved oxygen would not expected based on daytime monitoring, due to the presence of photosynthesising algae in the water and on the estuary bed. Dissolved oxygen varied the most at the Penguin Street site, where saturations over 120% have been measured, suggesting that this site is the most affected by algae. Gibson and Marsden (2016) report that seagrass beds are extensive along the South Brighton Spit area, which includes the Penguin Street site.



Figure 4-4: Temperature and dissolved oxygen at monitoring sites relative to guideline values. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots.

High concentrations of nutrients can increase the growth of nuisance macroalgae such as *Gracilaria* and *Ulva* species. The concentrations of DIN (the forms of nitrogen most readily available for plants) are highest at the two river sites and lowest at the estuary mouth and coastal sites (Figure 4-5) indicating that water entering the estuary from both rivers has higher DIN than that coming from the coastal waters. Concentrations at both river sites were below the guideline of 1.5 mg/L in the Canterbury Land and Water Regional Plan, with the exception of a few measurements at $\bar{O}p\bar{a}waho/Heathcote River.$



Figure 4-5: Dissolved inorganic nitrogen at monitoring sites relative to guideline in Canterbury Land and Water Regional Plan. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Dotted lines for whiskers indicate where summary statistics were based on data below the detection limit (shown as dashed grey horizontal line).

There are no guidelines for DIN in estuarine or coastal waters, but there are guidelines for the two components of it: ammoniacal-N and NOx-N. Ammoniacal-N concentrations at Ōtākaro/Avon River, Ōpāwaho/Heathcote River, South New Brighton Park, Humphreys Drive, Sandy Point and Beachville Road are almost always above the recommended guidelines (ANZG 2018), as are concentrations at Shag Rock during low tide (Figure 4-6). However, concentrations are below that guideline about half of the time at Penguin Street and Shag Rock during high tide, and are almost always below the guideline for coastal waters (Dudley et al. 2019)(ANZECC 2000) at Cave Rock and Southshore Beach. NOxN concentrations are above river and estuary guidelines at the same sites with one difference: at Beachville Road NOxN concentrations are below guidelines (Figure 4-6).



Figure 4-6: Nutrients at monitoring sites relative to guideline values as listed in Table 4-3. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Dotted lines for whiskers indicate where summary statistics were based on data below the detection limit (shown as dashed grey horizontal line).

When total nitrogen concentrations are compared to guidelines for rivers, estuarine and coastal waters (Figure 4-6), all sites in the estuary exceed the recommended guideline (Plew et al. 2018a) at all times, but the coastal sites are below guidelines for coastal waters (Dudley et al. 2019). Guideline values at coastal sites are based on monitoring data from the Canterbury region (Dudley et al. 2019), while guideline values for TN at estuary sites are biologically-based and use the threshold between Estuary Trophic Index (ETI) bands B and C for macroalgal growth for New Zealand estuaries (Plew et al. 2018a). These exceedances of estuary guideline values at sites throughout the estuary are in line with the current assessment of nitrogen availability and trophic state in the estuary based on load calculations (see section 5, below). The data for chlorophyll-*a* (measure of phytoplankton) does not suggest exceedances, as concentrations are below the guidelines most of the time at all sites.

There is a similar pattern in guideline exceedances for the two forms of phosphorus measured (Figure 4-6). Sites at both river mouths, in the upper estuary (South New Brighton Park, Humphreys Drive, Sandy Point) and Shag Rock during low tide have concentrations that are always or usually above the guidelines; whereas at sites closer to the estuary mouth (Penguin Street, Beachville Road, Shag Rock during high tide) and the two coastal sites, concentrations are almost always below guidelines. The relatively low TP concentrations in coastal sites compared to guidelines reflect greater mixing with ocean waters at these sites, but also the source of guideline values. For TP the coastal guideline is based on monitoring data from the Canterbury region; whereas the guidelines for rivers and estuaries are national guidelines based on data from around New Zealand (rivers) or from Australian waters (estuaries).

Nutrient concentrations, particularly nitrogen (N) can be taken up so quickly by algae in estuaries (especially in summer) that they can contribute to eutrophication without observable increases in the water concentrations. A better way to assess the impacts of nutrient availability on the trophic state of Ihutai/Avon-Heathcote Estuary is to assess the nutrient *loads* to the estuary rather than *concentrations*. This assessment is undertaken in section 5 and the effect of the nutrients and macroalgal growth on estuarine ecology are discussed further in section 5.3.

Suspended sediment concentrations can decrease light levels at the bed of the estuary, affecting primary producers such as phytoplankton and macroalgae. Suspended sediments also affect filter-feeding invertebrates and fish, potentially affecting their growth and condition (Hewitt et al. 2001, Robertson et al. 2015). TSS measures both inorganic and organic particles in the water and provides a good indication of the amount of sediment in the water. The water at the estuarine and coastal sites was almost always below the guidelines for TSS (based on an annual median), but concentrations at the Ōpāwaho/Heathcote River mouth were consistently above the guideline. Similarly, turbidity was generally within guidelines except at the Ōtākaro/Avon River and Ōpāwaho/Heathcote River sites where it was exceeded at all times. However, these two sites are tidal locations and the guideline used (ANZG 2018) is designed for managing freshwater sites and is therefore not entirely appropriate for these locations. Nonetheless, the results suggest that estuary ecology may be affected by high TSS around the Ōpāwaho/Heathcote River mouth, but not at other sites.



Figure 4-7: Total suspended solids and turbidity at monitoring sites relative to guideline values. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Dotted lines for whiskers indicate where summary statistics were based on data below the detection limit (shown as dashed grey horizontal line).

Metals such as copper, lead and zinc are toxic to plants and animals if present at concentrations that exceed their ability to process them. Total metal concentrations were below the detection limit most of the time for all sites except the Ōpāwaho/Heathcote River, and for total zinc in the Ōtākaro/Avon River (Figure 4-8). Of the sites within the main body of the estuary, Sandy Point had the highest concentrations. Coastal concentrations of copper and zinc were at times higher than the estuarine concentrations. This is likely due to the higher concentrations of solids in the samples (as measured by TSS) at coastal sites. Total lead and zinc concentrations were almost always within the water quality guidelines to protect from toxicity; however, copper concentrations at Ōpāwaho/Heathcote River, Sandy Point and Southshore Beach were above the guidelines in close to half, or even more of the samples, suggesting the potential for toxic effects on biota.

Dissolved metals (rather than total metals which measures both dissolved and particle-attached) are the most available for aquatic organisms and therefore the best measure for assessing possible toxicity. Dissolved copper was above the guideline in 5 out of 21 samples (~24%) at Cave Rock and only 1-2 samples at all other sites. As this data suggests potential for toxic effects on marine biota at the Cave Rock site, there should be further investigation of the copper concentrations at this site.



Figure 4-8: Total copper, lead and zinc at monitoring sites compared to guidelines to protect species from toxicity (ANZECC 2000). Lead guidelines for rivers (0.015 mg/L for Ötäkaro/Avon River and 0.029 mg/L for Öpäwaho/Heathcote River) not shown due to scale. Plot shows distribution of concentrations from January 2015 to December 2019. White horizontal lines within the boxes represent median concentrations, boxes extend from the 25th to 75th percentiles, whisker lines extend from 5th to 95th percentiles and maximum values are shown by dots. Many of the boxes are not fully visible as much of the data are below the detection limit (shown as grey horizontal line).

4.5 Summary

Water quality differs around the estuary, with sites closest to the mouth reflecting that of coastal water, and sites close to the rivers reflecting the riverine inputs. There is a higher risk of gastrointestinal illness or campylobacter infection if swimming or undertaking other contact recreation at sites near the river mouths and inner estuary; a moderate risk near the estuary mouth and low risk at the two coastal sites, based on measurements of *E. coli* (the preferred indicator for freshwater) or enterococci (the preferred indicator for saline waters). Sites in the estuary are not suitable for gathering shellfish, except the Beachville Road site, where indicator bacteria concentrations are within the guidelines.

The water quality could be adversely affecting the estuarine ecology, based on comparing the water quality measured in the last five years to guidelines. Ammoniacal-N, NOx-N, total nitrogen, DRP and TP frequently exceed guidelines at many sites, except those closest to the estuary mouth and coast. Total suspended solids and turbidity at the $\bar{O}p\bar{a}$ waho/Heathcote River mouth exceed the water quality guidelines, though these measurements are lower and within guidelines at other locations. Metal concentrations are generally within guidelines except for copper concentrations at Cave Rock, which have exceeded guidelines to protect from toxicity in over 20% of samples.

5 Nutrients may be leading to eutrophication of Ihutai/Avon-Heathcote Estuary

5.1 Nutrient concentrations only tell part of the story

Nutrient concentrations in Ihutai/Avon-Heathcote Estuary have decreased substantially since 2007, mainly due to the wastewater diversion reducing the input loads. There have been further decreases at most sites, or only slight increases in the period since 2014. Despite these strong improvements, and the resulting ecological improvements observed (Barr et al. 2019, Bolton-Ritchie 2015, Gibson & Marsden 2016, Zeldis et al. 2019), there currently are still areas with high macroalgal growth in the estuary (Bolton-Ritchie 2020).

As identified in section 4.4, nitrogen concentrations (TN, ammoniacal-N, NOx-N) in the estuary are above guideline values at some (but not all) sites. However, nutrients can be taken up so quickly by algae in the estuary (particularly in summer) that they can contribute to eutrophication without presenting as high measured concentrations in the water. The uptake of nutrients during summer is suggested by the seasonal pattern of nutrient values (see section 3.5) showing low values in summer when light and temperature do not limit growth, and high values in winter when growth is limited by light and temperature. Therefore, to assess the nutrient availability and to increase understanding of why macroalgae are still common in the estuary, we need to assess nutrient *loads* entering the estuary, as well as the nutrient concentrations within it.

We used the Estuarine Trophic Index (ETI) Tool 1 (Plew et al. 2020, Zeldis et al. 2017c) to conduct this assessment, as described in detail Appendix E and briefly here. We first calculated nitrogen loads to the estuary from Ōtākaro/Avon and Ōpāwaho/Heathcote rivers using data from Christchurch City Council's monthly monitoring (Marshall & Noakes 2019) and daily flow data based on ECan's flow monitoring location in each river. For several drains where there were no data, we based our estimate on Burge (2007) and Bolton-Ritchie and Main (2005), adjusting those loads for wet years and dry years. We then used the CLUES Estuary approach within the ETI tool, which uses simple dilution models to predict potential nutrient concentrations in the estuary (in the absence of algal uptake). The ETI tool then calculates the eutrophication susceptibility, based on four bands with A being low susceptibility and D being very high eutrophication susceptibility.

5.2 Ihutai/Avon-Heathcote Estuary is in a eutrophic state

The current average annual dissolved inorganic nitrogen (DIN) load to the estuary from rivers and drains is around 200 tonnes/year based on loads calculated from 2015 to 2019 inclusive. This, in combination with information about the estuary size and flushing, places the estuary in a D band of the ETI for macroalgal eutrophication, described as "very high eutrophication susceptibility". The load from the drains is uncertain, and if over-estimated, it is possible the estuary could be in the C band - "high eutrophication". However, for two out of the five years, the contribution from the $\bar{O}t\bar{a}karo/Avon$ and $\bar{O}p\bar{a}waho/Heathcote$ rivers alone was above the C/D threshold.



Figure 5-1: Annual riverine and drain loads of DIN to the estuary. Dashed horizontal lines give threshold values corresponding with ETI Tool 1 bands.

Ecological qualities expected from estuaries that have a high to very high susceptibility to macroalgal eutrophication (bands C and D) are as follows.

- Ecological communities (e.g., bird, fish, seagrass, and macroinvertebrates) are strongly impacted by macroalgae.
- Persistent areas of very high % macroalgal cover (>75%) and/or biomass (>500 g/m² wet weight), including macroalgae partially buried in sediment;
- Degraded sediment quality with sulfidic conditions near the sediment surface.

These qualities are consistent with measurements of trophic state indicators in Ihutai/Avon-Heathcote Estuary. This estuary is classed as a 'Shallow Intertidal-Dominated Estuary' in the ETI classification (Hume 2018), which are vulnerable to eutrophication caused by excessive macroalgal growth, rather than high phytoplankton concentrations which dominate in lakes and deep estuaries, and/or those with long residence times (Plew et al. 2017, Plew et al. 2020). ECan data from the summer of 2019/2020 shows around 40 % of the available intertidal habitat area of the estuary area had >5% coverage by macroalgae, with very high (>75%) macroalgal cover in large areas. Bolton-Ritchie (2020) conducted her assessment of macroalgal cover according to ETI tool 2 methodology (Robertson et al. 2016b) and scored the estuary in band D (Very high eutrophication).

5.3 Algal growth affects sea grass, invertebrates and fish

The primary symptom of estuary eutrophication is high biomass of phytoplankton or macroalgae. Increases in water column chlorophyll-*a* in this study indicate increased phytoplankton growth between 2014 and 2019. However, concentrations of chlorophyll-*a* in Ihutai/Avon-Heathcote Estuary remain comparable to those in nearby ocean water, likely due to the relatively short residence time and shallow, well-mixed water column.

In contrast, macroalgal growth in the estuary is extensive and high relative to other New Zealand estuaries with similar physical properties. In some estuaries with high nutrient loads, macroalgal growth is not high because there are few suitable shallow or intertidal areas available for macroalgae to grow. Ihutai/Avon-Heathcote Estuary is shallow and well mixed, with a large intertidal area suitable for macroalgal growth. The high nitrogen availability in Ihutai/Avon-Heathcote Estuary is enhancing productivity of nuisance macroalgae, including *Ulva* and *Gracilaria*. In nutrient rich conditions, these macroalgae tend to outcompete other benthic plants such as seagrasses (Cardoso et al. 2004). This can cause detrimental effects to estuary ecology, as seagrasses form excellent habitat for a range of juvenile fish and shellfish (Morrison et al. 2014, Morrison et al. 2009). As macroalgae rots, it tends to reduce oxygen available in sediments and can make these sediments less suitable for animal life. As well as its ecological effects, rotting macroalgae can release unpleasant odours that make the estuary less pleasant for people (Barr et al. 2012).

Sediment oxygenation, grain size and organic matter monitored across the estuary since 2011 have shown generally good sediment health, suggesting that the estuary sediments are resilient to eutrophication (Zeldis et al. 2019). However, monitoring of macroinvertebrates has shown sediment communities indicative of poor ecological health in some areas (*Bolton-Ritchie 2015*). There is evidence of impacts by macroalgal cover on seagrass beds on the eastern flats of the estuary (Bolton-Ritchie 2020, Hollever & Bolton-Ritchie 2016).

Overall, measurements of ecological indicators of trophic state in the estuary in recent years are broadly in agreement with predictions of ecological state based on nitrogen loading to the estuary.

6 There are multiple drivers of water quality in the estuary

6.1 The wastewater diversion improved water quality and ecology

Prior to the diversion of the wastewater discharge, the Ihutai/Avon-Heathcote Estuary was placed substantially higher in ETI Band D for susceptibility for macroalgae-driven eutrophication using the CLUES-Estuary approach. Diversion of Christchurch City's wastewater discharge to an ocean outfall in March 2010 resulted in rapid decreases in nutrient concentrations at almost all sites in the estuary. Macroalgal biomass and condition also reduced substantially in the two years following the wastewater diversion (Barr et al. 2019), as did other trophic indicators (Dudley et al. 2019).

Monitoring of macroinvertebrates in the years since the diversion of wastewater from the estuary has shown shifts to less pollution tolerant taxa at sites near the discharge point (Bolton-Ritchie 2015, Dudley et al. 2019, Gibson & Marsden 2016). There is evidence of strong recovery of seagrass beds on the eastern flats of the estuary subsequent to the wastewater diversion (Bolton-Ritchie 2020, Hollever & Bolton-Ritchie 2016). Sediment oxygenation, grain size and organic matter content monitored across the estuary since the wastewater diversion have shown that estuary sediments recovered quickly following the reduction in wastewater nutrient inputs (Bolton-Ritchie 2015, Dudley et al. 2019).

6.2 Nutrient loads from rivers affect estuary water quality

We assessed nitrogen loads and flows to the estuary for each year from 2009 to 2019, as described in Appendix E. The DIN concentrations are consistently higher in the $\bar{O}p\bar{a}waho/Heathcote$ River than the $\bar{O}t\bar{a}karo/Avon$ River, but the average annual mean flow of the $\bar{O}t\bar{a}karo/Avon$ River (based on data from 2007 to 2019) is 1.9 m³/s, 1.7x that of $\bar{O}p\bar{a}waho/Heathcote$ River at 1.1 m³/s. Therefore, in total the load from $\bar{O}t\bar{a}karo/Avon$ River can be higher than that from the $\bar{O}p\bar{a}waho/Heathcote$ River. The relative contribution of each river differs from year to year, depending on the flow and measured concentrations. We note that the loading from drains near the wastewater treatment ponds is not based on recently measured data, but data reported in Burge (2007) and adjusted based on flow (see Appendix E for method), so there is considerable uncertainty associated with this estimate.

There was no apparent increase in riverine loading over the period of interest, but there were fluctuations from year to year largely associated with changes in flow from rivers (Figure 6-1). Years with higher flow tend to have higher nitrogen loading to the estuary, though this flow effect may be somewhat over-estimated (see Appendix E). This yearly fluctuation in loads is observable in the DIN and DRP concentrations at the river sites monitored in the estuary, and at the nearby estuary sites of South New Brighton Park and Humphreys Drive (Figure D-7 and Figure D-9, Appendix D). For example, DIN and DRP concentrations were higher in 2014 than in 2015 and 2016; then somewhat more variable, including higher concentrations, for 2017 through to 2019 (Figure D-7 and Figure D-9).

Whilst we have not conducted trend analysis on offshore coastal water quality data since 2018, no statistically significant increases in ammoniacal-N or NOx-N concentrations were present in the 10 year period from 1 January 2008 to 31 December 2017 at offshore sites around coastal Canterbury (Dudley & Jones-Todd 2018). We see it as unlikely that increases in primary production in Ihutai/Avon-Heathcote Estuary are driven by increases in oceanic nitrogen supply.





6.3 The nutrient contribution from bird life is likely minor

The nutrient and bacterial contributions of each bird species are shown in Figure 6-2 (major contributors only). This indicates that while NZ scaup and gulls contribute over 50% of the nutrients from birds, they contribute around 10% or less of the indicator bacteria. On the other hand, the NZ shoveler, paradise shelduck and other ducks contribute over 75% of the bacterial load associated with birds (Figure 6-2) as they excrete higher daily loads of indicator bacteria compared to other birds.



Figure 6-2: Bird species' contributions to total number, and total daily nutrients and bacteria. Bird numbers are annual-averages, based on monthly bird counts from August 2009 to July 2010 (Crossland 2013). Nitrogen and phosphorus were calculated using Waterbirds version 1.1. Faecal indicator bacteria (number deposited per day) were calculated based on avian faecal loadings in Moriarty et al. (2011).

Based on the above assessment, total annual nutrient loads from the herbivorous bird species are around 3,200 kg/yr of nitrogen, and 260 kg/yr of phosphorus. This forms around 2-3% of the total annual nitrogen load, and 5% of the annual phosphorus load based on our calculations of the combined load from the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers and the drains (Figure 6-3). These values are high compared to values reported for lakes and wetlands internationally (Hahn et al. 2008, e.g., Schernewski 2003) where allochthonous nutrient inputs from waterbirds have been estimated at less than 1% of catchment loads, and closer to those reported for lakes with very dense populations of roosting birds (Marion et al. 1994).



Figure 6-3: Contribution of waterbirds and three rivers to the total daily nutrients and bacteria in the estuary. Rivers based on average of five years of data.

We did not consider nutrient loads from birds in the spatial nutrient modelling (see next section) because:

- 1. nitrogen loading from birds is low compared to loads from rivers
- 2. the location that faecal matter from birds is deposited is unknown, and dispersal of nutrients from birds cannot be represented in the spatial models
- 3. it is likely that these bird-related loads are relatively stable through time, so we see it as unlikely that bird populations are responsible for increases in algal growth in the estuary.

The bacterial load from the birds (all species) is around 2×10^{14} *E. coli* per day. This is well over the contribution from the rivers estimated at 6×10^{11} *E. coli* per day (Figure 6-3). The estimate from birds is highly uncertain, for example we have not considered how much of the faecal contribution from birds is deposited within the estuary, or above the high tide zone (e.g., in roosting areas). However, based on rough estimates, the bacterial contribution from the rivers alone (which equates to ~6-10 E. *coli*/100 mL in the estuary when mixed with coastal waters) does not appear to account for the concentrations of *E. coli* measured in the estuary (median concentrations 20-200 *E. coli*/100 mL,

section 4.3). This suggests that the birds using the estuary could be important contributors to the indicator bacteria counts, as previously identified through faecal source tracking studies (Moriarty & Gilpin 2015). We recommend that:

- more information be obtained regarding the locations of bird deposits
- the influence of the river and bird inputs to the estuary is modelled using a more realistic (non-conservative) approach that accounts for bacteria die-off and resuspension of bacteria from estuary bed sediments
- modelling be undertaken for baseflow (median) conditions, for flood conditions and in different seasons, when migratory birds may be present in higher or lower numbers.

6.4 Some sites are more affected by river water than others

The modelling shows the sites that are most affected by the inputs from the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers, and which sites are more influenced by the ocean water inputs (Figure 6-4) in terms of percent coastal seawater concentration. The South New Brighton Park site is most affected by the Ōtākaro/Avon River but the river water is well-mixed with seawater at the Penguin Street site. The Humphreys Drive and Sandy Point sites are most affected by the Ōpāwaho/Heathcote River, but the Beachville Road site is influenced by water from both rivers and from the ocean. There is good dilution by coastal water in the central and southern areas of the estuary.



Figure 6-4: Percent contribution of coastal seawater to waters of Ihutai/Avon-Heathcote Estuary

The influence of the rivers on each site depends on both the dilution by seawater (as shown above) and on the contaminant load contributed by each river. The modelled results (Figure 6-5, and Table 6-1) are not directly comparable to the measured concentrations as grab-sampling is undertaken at a specific time – high tide for most sites – whereas the model provides temporally averaged data for each location. The model also estimates average concentrations from the water surface to the bed. If there is vertical stratification in the estuary, which has been noted near the Ōtākaro/Avon River mouth, then grab-samples collected from the surface would have higher concentrations than a fully-mixed sample (as modelled).

The modelling does indicate which sites would have highest concentrations of DRP, ammoniacal-N and NOx-N (Figure 6-5 and Table 6-1), based on the river inputs. A comparison of these predicted concentrations to the measured data (Figure 4-6) suggests that there are inputs of ammoniacal-N and DRP at the Sandy Point site, that are additional to those included in the model (Ōtākaro/Avon and Ōpāwaho/Heathcote Rivers and Linwood Canal only). It is likely that these inputs include the drains around the wastewater treatment ponds as identified by Burge (2007). More up to date investigations of the quality and quantity of water in these drains would be useful to understand their influence at the Sandy Point site, and on the estuary as a whole. They are also likely to contribute faecal indicator bacteria. The flow from the Linwood Canal into the estuary was estimated only and is highly uncertain; it is possible that nutrient loading from this drain is also higher than estimated and has more effect on the zones of the estuary than modelled here.

| | Zone 1 | Zone 2 | Zone 3 | Zone 4 | Zone 5 | Zone 6 | Zone 7 | Zone 8 | Whole estuary |
|------------------------------|--------|--------|--------|--------|--------|--------|--------|--------|---------------|
| Ammoniacal | | | | | | | | | |
| Average concentration (mg/L) | 0.023 | 0.020 | 0.015 | 0.054 | 0.025 | 0.015 | 0.013 | 0.012 | 0.010 |
| Percentage from each source: | | | | | | | | | |
| | 4% | 10% | 14% | 91% | 60% | 27% | 16% | 13% | 30% |
| Ōtākaro/Avon | 84% | 64% | 35% | 3% | 10% | 14% | 16% | 12% | 24% |
| Linwood Canal | 0% | 0% | 0% | 0% | 2% | 1% | 0% | 0% | 1% |
| Ocean | 11% | 25% | 51% | 6% | 29% | 58% | 67% | 74% | 46% |
| NOx-N | | | | | | | | | |
| Average concentration (mg/L) | 0.60 | 0.42 | 0.20 | 0.96 | 0.36 | 0.15 | 0.11 | 0.087 | 0.014 |
| Percentage from each source: | | | | | | | | | |
| | 3% | 9% | 19% | 95% | 77% | 51% | 35% | 36% | 42% |
| Ōtākaro/Avon | 96% | 90% | 76% | 5% | 21% | 42% | 56% | 52% | 53% |
| Linwood Canal | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% | 0% |
| Ocean | 1% | 1% | 4% | 0% | 2% | 7% | 9% | 12% | 4% |
| DRP | | | | | | | | | |
| Average concentration (mg/L) | 0.023 | 0.017 | 0.010 | 0.022 | 0.011 | 0.007 | 0.007 | 0.006 | 0.004 |
| Percentage from each source: | | | | | | | | | |
| Ōpāwaho/Heathcote | 2% | 4% | 8% | 85% | 49% | 21% | 12% | 10% | 20% |
| Ōtākaro/Avon | 94% | 83% | 60% | 8% | 24% | 31% | 35% | 28% | 46% |
| Linwood Canal | 0% | 0% | 0% | 0% | 1% | 0% | 0% | 0% | 0% |
| Ocean | 5% | 12% | 32% | 6% | 25% | 48% | 53% | 61% | 33% |

Table 6-1:Contributions of rivers and oceanic nutrient sources to each estuary zone. Estuary zones followthose in Figure E-1, Appendix E).

Percentage of nutrient contributed from each of four sources



Figure 6-5: Modelled concentrations (mg/L) of ammoniacal-N (left), NOx-N (middle) and DRP (right) in Ihutai/Avon-Heathcote Estuary based on median flow and nutrient concentrations from 2014-2019.

6.5 Effects of estuary water residence time and temperatures

The Canterbury earthquakes in 2010 and 2011 caused decreases of about 14% in the overall volume (and therefore tidal prism) of Ihutai/Avon-Heathcote Estuary (Measures & Bind 2013, Orchard & Measures 2016). This would mean a higher flushing rate by rivers and a decreased mean residence time of water in the estuary, and lead to reduced phytoplankton as they have less time to grow and reproduce before being flushed from the estuary (Plew et al. 2020). An overall reduction in residence time is consistent with the trend analyses findings of decreased chlorophyll-a concentrations at most estuarine sites (Section 3.2). However, decreases in volume did not occur evenly throughout the estuary and there were some locations where the volume increased, so changes to residence time, and therefore phytoplankton growth are not expected to occur evenly throughout the estuary.

Trend analysis in this study has shown the likelihood of warming temperatures at many of the sites sampled. Seasonal comparisons indicate that this is largely due to warmer than normal winter temperatures. New Zealand's marine environment has seen a warming of about 0.1°C-0.3°C per decade in the last 30 years, leading to unprecedented marine heat waves in four of five summers since 2016 (Patston 2019), with some of the highest temperature anomalies focused on the eastern South Island. The increase in seawater temperatures across the region has resulted in particularly warm winter temperatures (Schiel et al. 2016). If the trend analysis results in this study showing recent increases in temperatures are confirmed, these could be, at least in part, reflective of regional patterns of warming associated with global climate change.

Temperature changes are likely to result in changes in primary production in the estuary. With enough light, macroalgal growth rate is controlled by nutrients and temperature in non-linear fashion, with low growth rapidly increasing as nutrients increase from low to moderate levels, reaching a saturated rate at high nutrient levels. Increased temperature has a controlling effect on the shape of this curve, potentially 'up-shifting' growth at equivalent nutrient levels, meaning that the nutrient threshold at which high growth is reached is lowered. Algal growth is seasonally limited by light and temperature (in winter), and so increases in winter temperature could increase winter algal growth rate as light becomes non-limiting in late winter. It is therefore possible that warming is at least partly responsible for observed recent increases in algal abundance in the estuary, with warmer winter temperatures increasing the capacity of algae to use inorganic nitrogen loads from land (i.e. DIN inputs from the rivers), increasing overwintering populations of algae in the estuary and increasing productivity in early spring and summer. This in turn affects other aspects of water quality including dissolved oxygen and pH.

6.6 Summary

The diversion of the Christchurch City's wastewater discharge in March 2010 resulted in almost immediate improvements in the water quality, macroalgae and sediment in Ihutai/Avon-Heathcote Estuary. Improvements in macroinvertebrate communities occurred more slowly. The nutrient loads from the rivers affect nutrient concentrations in the estuary, particularly at sites closest to the river mouths. There are also locations in the estuary (e.g., Sandy Point) where nutrient concentrations are much higher than expected based on riverine inputs – indicating additional sources, likely to include the drains which are not well-quantified. The wild birds using the estuary appear to be adding little to the nutrient content of the estuary but are probably major contributors to its faecal indicator bacteria levels. Decreases in overall estuary water residence time may have had a decreasing influence on algal productivity at some sites, while water temperature increase (if confirmed) may be exacerbating algal growth and blooms.

7 The current monitoring programme is fit for purpose

7.1 How did we approach this?

Water quality monitoring in the Ihutai/Avon-Heathcote Estuary is carried out as part of the "Healthy Estuary and Rivers of the City" monitoring programme for a range of purposes, including water quality and ecosystem health monitoring (Batcheler et al. 2009). As part of this state and trends assessment, ECan sought a review of the suitability of its current water quality monitoring programme for measuring state and trend in the estuary, and for identifying issues as they arise.

There are four key components this review addresses:

- Do the current number and location of sampling points within the Ihutai/Avon-Heathcote Estuary adequately represent the water quality in the Estuary? Should sampling points be moved and/or added?
- Are the current suite of monitored water quality variables and the metadata collected sufficient?
- Are the current sampling methods, including sampling platform, frequency and timing with respect to tide, robust and in line with the standards of comparable monitoring programmes around New Zealand?
- What monitoring is required to make connections between sources of pollutants, freshwater quality and estuarine ecological health? What further data sets are required to investigate issues further, such as measurements to be collected to improve application of the ETI tools, or consider effects of climate change?

A brief outline of the legislative context, environmental context, and recent national developments of relevance to ECan's monitoring programme are provided in Appendix E, section 4. We then address each of the four key components of the review as outlined above.

7.2 Number and locations of monitored sites

In this section we consider the number and location of monitoring sites within Ihutai/Avon-Heathcote Estuary with respect to measuring water quality state and trends in the estuary, and requirements for the ETI dilution modelling approach to assessing estuary susceptibility to nitrogen loads.

The ETI dilution modelling approach calculates 'potential' nutrient concentrations of estuarine water and requires local oceanic (i.e. open coast) total nitrogen (TN) concentration, TN concentrations in fresh water flows to estuaries, and freshwater flow rates (Plew et al. 2020, Plew et al. 2018b). These data are important for modelling loads of N entering estuaries from land that correspond with changes in estuarine trophic state (e.g., Dudley & Plew 2017, Plew & Dudley 2018) and linking these predicted changes with observed data (e.g., Robertson and Stevens 2016). Therefore, as outlined below, we recommend that water column nutrients are monitored in all major terminal river reaches entering the estuary (at locations unaffected by tidal state), as well as within the estuary and on the adjacent coast (Zaiko et al. 2018, Zeldis et al. 2017b).

7.2.1 Estuary sites

As shown in sections 3, 4 and 6, spatial variability in water quality within Ihutai/Avon-Heathcote Estuary means that water quality at a single site is unlikely to represent average water quality conditions for the whole estuary. The current monitoring network provides a high density of sites for an estuary the size of Ihutai/Avon-Heathcote, when compared to other council sampling programmes nationally (Zeldis et al. 2017b). These sites show the spatial variability in water quality (section 3.6) and, based on distributions of river water shown in 6.4 provide good spatial coverage of the continuum between areas of the estuary with greater and less freshwater influence. Furthermore, these sites also provide good coverage of the major freshwater inputs to the estuary: the Ōtākaro/Avon and Ōpāwaho/Heathcote Rivers, and drains on the Northern and Western shores. Hence, we would encourage maintenance of all current site locations.

7.2.2 River sites

The monitoring programme includes sites at the mouth of each of the two main rivers feeding the estuary, the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers. These sites are useful for indicating the sources of contaminants, particularly in relation to nearby sites. However, they are not useful for *modelling* of inputs into the estuary, as the water at these sites is often mixed with coastal waters, rather than being freshwater and flows at these sites are influenced by tides.

Data upstream of the saline influence with flows minimally affected by tides is needed for modelling loads and estuary dilution (Plew et al. 2018b). This modelling is useful to measure change in overall upstream catchment pressure (e.g., changes in nutrient or sediment loads) and examine how specific sources of water (e.g., rivers, drains, and other point sources) contribute to changes in the ecological health and water quality of the estuary. To undertake these analyses (section 5) we used water quality data from CCC's monitoring programme, and flow data from ECan's river flow monitoring programme. The data from these sources was generally appropriate for these analyses, and if CCC continue to monitor river water quality there is no need for ECan to repeat this in the HERC programme.

However, there were no flow data for the outlet of Linwood Canal, or nutrient, faecal indicator bacteria or flow data from the drains near the wastewater treatment ponds. We calculated Linwood Canal flow data from upstream flows and model outputs (NZ Rivermaps), and drain nutrient and flow data from relatively old reports (Bolton-Ritchie & Main 2005, Burge 2007).Our load calculations indicate that these drains are likely to exert significant influence on the water quality and ecology of the estuary, and accurate estimates of inputs from these drains are imperative to determine accurate load estimates for the estuary.

7.2.3 Oceanic sampling

There are two sites in the HERC programme that are predominantly coastal water: Cave Rock and the Southshore Beach (Caspian Street). While these sites provide an indication of the quality of coastal water, they may still be influenced by the rivers and estuary. For water quality modelling of the estuary, oceanic data are needed to calculate mixing within the estuary. This will require sampling of the open coast (i.e., from outside the estuary and outside the influence of freshwater plumes from land). Such sampling is not included in the HERC programme, but is included in ECan's regional coastal water quality sampling programme. That programme, recently reviewed by Dudley et al. (2019), provides excellent data for use with the ETI tool 1 (section 5) and in numerical modelling (section 6.4); coastal hydrodynamic models suggested that while all of the coastal sites in the CRC coastal water quality monitoring network were influenced to some extent by river plumes, sites 3 km

offshore sites were the least-influenced and were suitable for estuary modelling purposes. Recent estuary and coastal water quality assessments of Ihutai Estuary and the nearby coast (e.g. Dudley et al. (2019), Plew et al. (2017), this study) used yearly or seasonally averaged nutrient values for the oceanic water, so current quarterly coastal sampling is suitable for comparison with estuary values sampled monthly. Therefore, there is no need to conduct further off-shore monitoring under the HERC. The position of coastal sites near the estuary also appear appropriate for examining the effects of outflow water from the estuary on nearshore coastal ecosystems.

7.3 Monitored variables

In this section we address whether the current suite of water quality variables and metadata collected by ECan are optimal, based on the environmental and legislative context described in Appendix E. We compare the list of variables monitored in this programme to those recommended for estuary water quality monitoring in recent reports nationally. We also consider variables needed for estuarine catchment modelling purposes.

Table 7-1 lists all variables recommended for a range of coastal or estuarine water quality monitoring in a selection of recent relevant reports. The relevant reports are:

- Cornelisen (2010) recommended biological indicators for monitoring environmental conditions in coastal waters
- Zaiko et al. (2018) identified estuarine attributes suitable for the establishment of national thresholds on which to manage upstream environments
- Zeldis et al. (2017b) recommended water quality variables for regional SoE monitoring that, if adopted uniformly across councils, would improve national level SoE analyses
- Zeldis et al. (2017b) listed indicators used in assessment of the trophic state of estuaries in ETI tool 2.

Table 7-1:Recommended water quality variables for SoE and recreational water quality monitoring of
marine and estuarine water quality from selected recent reports.Where recommendations differ between
estuarine and fully marine waters, E = Estuarine, M = Marine.

| Variable | MfE (Dudley et al. 2017) Core | MfE (Dudley et al. 2017) Support | Horizons (Cornelisen 2010) | MfE (Zaiko et al. 2018) | ETI tool 2 indicator | NEMS method available | Currently monitored in Ihutai Estuary by ECan | | |
|--|---|---|----------------------------------|----------------------------------|-------------------------|-----------------------------|--|--|--|
| Major physico-chemical variables | | | | | | | | | |
| Salinity | \checkmark | No | E = Yes M = Yes | No | No | \checkmark | \checkmark | | |
| Temperature | \checkmark | No | E = Yes M = Yes | No | No | \checkmark | \checkmark | | |
| Dissolved oxygen | \checkmark | No | E = Yes M = No | No | \checkmark | \checkmark | \checkmark | | |
| рН | \checkmark | No | E = Yes M = Yes | No | No | \checkmark | \checkmark | | |
| Optical variables | | | | | | | | | |
| Visual clarity | \checkmark | No | E = Yes M = No | No | No | \checkmark | No | | |
| Turbidity | No | \checkmark | E = Yes M = Yes | No | No | \checkmark | \checkmark | | |
| Total Suspended Solids | \checkmark | No | E = Yes M = No | \checkmark | No | \checkmark | \checkmark | | |
| Light penetration | No | \checkmark | E = No M = No | No | No | \checkmark | No | | |
| Coloured dissolved organic matter | No | \checkmark | E = Yes M = No | No | No | \checkmark | No | | |
| Munsell Colour | No | \checkmark | E = No M = No | No | No | \checkmark | No | | |
| Nutrients | | | | | | | | | |
| Total nutrients (TN, TP) | \checkmark | No | E = Yes M = Yes | \checkmark | No | \checkmark | \checkmark | | |
| Dissolved nutrients (NOXN, NH4N, DRP) | √* | No | E = Yes M = Yes | No | No | \checkmark | \checkmark | | |
| Dissolved organic nutrients (DON, DOP) | No | No | E = Yes M = Yes | No | No | No | No | | |
| Microbiological indicators | | | | | | | | | |
| Enterococci | \checkmark | No | E = No M = Yes | | No | \checkmark | \checkmark | | |
| Faecal coliforms | No | \checkmark | E = No M = No | √ ** | No | \checkmark | Some locations | | |
| E. coli | No | \checkmark | E = Yes M = No | | No | \checkmark | Some locations | | |
| Chlorophyll-a | \checkmark | No | E = Yes M = Yes | No | \checkmark | \checkmark | \checkmark | | |
| Phytoplankton assemblage | No | \checkmark | E = No M = No | No | No | No | No | | |
| Other toxicants | No | No | E = Yes M = No | No | No | ✓ (metals) | ✓ (metals) | | |

* DRP deemed a supporting variable in fully marine (oceanic) waters

** The recommended microbiological indicator is not specified

7.3.1 State of Environment (SoE) monitoring

Water column monitoring in estuaries – including the variables measured – should align with the methods and timing of monitoring taking place at sites immediately upstream and in the open coast. This alignment aids in attributing changes in estuaries to processes and activities in upstream catchments and nearby marine systems. Standardisation of variables measured across the "mountains to sea" (ki uta ki tai) continuum is echoed in recent MfE reports (Zaiko et al. 2018, Zeldis et al. 2017b), and is consistent with the concept of integrated management required by both the NZCPS and the NPS-FM (refer Section 7.2).

In addition to the various water quality variables listed in Table 7-1, Dudley et al. (2017) also recommended inclusion of an integrated index of estuarine ecological health to facilitate setting water quality thresholds (i.e., boundaries between bands of environmental state). Because ECan is already undertaking monitoring in line with ETI methodologies (which provides this index), we pay special attention to water quality variables that can be included in ETI calculations.

We note that the variables included in the NEMS (2020) for discrete coastal water quality sampling are not a list of recommended variables, but a list of variables typically measured as part of long-term SoE programmes for coastal waters. Rationale for variables in Table 7-1 that are not monitored by ECan is not provided here, but can be found in recent publications specific to New Zealand estuarine water quality monitoring (NEMS 2019, Zaiko et al. 2018, Zeldis et al. 2017b).

In Table 7-1, we note that of the recommended 'core' variables listed in Dudley et al. (2017) only visual clarity is missing from current monitoring at Ihutai/Avon-Heathcote Estuary sites. Visual clarity and turbidity typically co-vary strongly in coastal waters; the relative benefits of both as measures of light attenuation are discussed in Davies-Colley and Smith (2001). The NEMS Water Quality (NEMS 2019) provides guidance on a range of options for visual clarity measurements, including the use of a Stream Health Monitoring Kit (SHMAK) tube where waters are very sediment-laden.

The Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas (MfE/MoH 2003) require measurement of enterococci in marine waters as an indicator for the risk of illness from swimming and other contact recreation activities. However, where the waters are used for recreational shellfish gathering, faecal coliforms are the recommended indicator¹². For brackish or estuarine waters, it is unclear in the guidelines whether enterococci or the freshwater indicator, *E. coli*, should be monitored.

A recent review of microbial monitoring for marine recreational areas (McBride et al. 2019) provides the advice to monitor enterococci in estuaries with long residence times (>3 days). For estuaries with a shorter residence time, *E. coli* is the appropriate choice when near the inflowing river water, but enterococci should be monitored near the estuary mouth. What to monitor between these locations still needs consideration and it appears both indicators should be measured (McBride et al. 2019). Based on that report, given the residence time of Ihutai/Avon-Heathcote Estuary (ca. 5.5 days (Plew et al. 2017)), data showing large variation in salinity between sites near rivers and sites near the estuary mouth, and because sites near river mouths are monitored at low tide, we recommend continuing with the current approach, monitoring both *E. coli* and enterococci at all sites except those at the estuary mouth and coast where only enterococci is monitored.

¹² It is hoped in the future that these guidelines might be based on *E. coli* or enterococci. McBride et al. (2019) propose the use of a new risk-based shellfish uptake-and-depuration model which could be based on enterococci. This would reduce laboratory costs for analysis of coastal water samples.

7.3.2 Variables for estuary dilution modelling

Both the ETI tool 1 dilution modelling approach of Plew et al. (2020) and the more detailed 3dimensional numerical modelling approach used in section 6.4 require an understanding of the nitrogen loads carried into the estuary from freshwater flows and the ocean. Plew et al. (2020) also use salinity data collected at high tide within the estuary to validate estimates of mixing of fresh water and ocean water. These approaches therefore require monitoring of total nitrogen, NOx-N, and salinity at terminal river reaches entering estuaries, as well as within estuaries and in nearby ocean water.

The results of trend analyses on nutrient and chlorophyll-*a* data from 2014 to 2019 inclusive highlight the importance of accurate quantification of nutrient loads entering the Ihutai/Avon-Heathcote Estuary. Trend analyses showed surface water chlorophyll-*a* concentrations increased at most sites, while nutrient concentrations increased only slightly, or declined. Increases in seawater temperature are also apparent during this period. We plotted date of sampling against time of sampling for each site and observed that time of day had been kept consistent through the time series for each site, generally to within 2-3 hours. This suggests that changes in parameters are not due to time of sampling. As discussed, a possible reason for declining nutrient concentrations is that much of the dissolved nutrient load entering the estuary is taken up into algae, such as phytoplankton, *Ulva* and *Gracilaria*. The "potential" nutrient concentrations calculated by dilution modelling do not consider uptake of nutrients and so may provide a better measure (than measured seawater nutrients) with which to assess the relative contributions of nutrient loading and climate change to ecological change in the estuary. However, the value of dilution modelling is dependent on the quality of input data, and therefore obtaining quality data for the drains should be a priority.

ETI tool 1 can be used to predict trophic conditions within the estuary and N-loads from land that correspond to changes in these trophic conditions. The predicted trophic condition from ETI tool 1 can be validated by assessing trophic state using ETI tool 2 (Robertson et al. 2016b, Zeldis et al. 2017b) using indicators measured within the estuary. These indicators include chlorophyll-*a* and dissolved oxygen in estuary water, which are among the water quality variables included in current ECan monitoring.

7.4 Sampling and measurement methods

In this section we review the current sampling and laboratory methods, including sampling platform, frequency and timing with respect to tide.

7.4.1 Detecting changes in water quality

Frequency and timing of monitoring

The current monthly sampling frequency is sufficient and consistent with many other long term or SoE-based water quality monitoring programmes. Because of the variability of estuarine water quality at short time scales (including strong seasonal variation for many measurements), long and relatively intensively sampled time series are required to detect changes in estuarine water quality. More frequent sampling (e.g., monthly instead of quarterly) increases the power of statistical tests to detect trends. Water quality trend analysis techniques typically rely on multi-year to multi-decadal data series with few missing data points. For example, recent national water quality trend analyses used 8–20 year, monthly or quarterly-sampled datasets and required data to be present for 80% of the sampling dates and evenly distributed across the years examined (Larned et al. 2015, Zeldis et al. 2017b).

Maintenance of existing sites, sampling and measurement methods is important for the detecting changes in water quality over time. We recommend continuing to sample at the same monthly frequency to maintain the ability to compare trends in water quality between river mouth and nearshore ocean water sites. The uninterrupted regular time series data collected to date provides an ideal dataset for examining trends in water quality over time and for detecting changes in water quality in Ihutai/Avon-Heathcote Estuary, and is also suitable for assigning that change to a specific nutrient source (e.g., partitioning total loads by river).

Tidal state

One of the major sources of variability in estuarine water sampling is tidal state. This is largely because at high tide there is greater dilution of the freshwater inflows from land by ocean water than at low tide. Tidal dilution therefore creates problems for SoE sampling which has the twin goals of being representative of water quality state within an estuary and detecting trends in water quality through time. For a monitoring programme that seeks to assess estuarine water quality state, it would be most appropriate to randomise for tide, stratify sampling by tide, or simply ignore tide in planning but record it at the time of sampling. All these approaches would be most appropriate to characterise "average" water conditions. However, if the primary monitoring aim is to detect changes in water quality through time, it would be most appropriate to sample consistently at a single tidal state to minimise the effect of tide and increase statistical power. Three potentially appropriate monitoring approaches that fit both these `conflicting' monitoring purposes are:

- sample regularly (e.g., quarterly or monthly) at both high and low tide;
- sample regularly (e.g., quarterly or monthly), without regard to tidal state and record time and tidal conditions at the time of sampling; or
- sample consistently at mid-tide to capture a "middling" mix of ocean and fresh waters.

The first approach has been used successfully in New Zealand (e.g., Invercargill City Council data described in Zeldis et al. (2017b)). This approach allows trend analysis on both high tide and low tide datasets, and when data are considered together should give a reasonable average condition for estuary water. However, this approach may not be practical where travel times between sites are great and has substantially greater cost. The second approach sacrifices statistical power in trend analysis; sampling may need to be more frequent to detect trends in water quality through time. The last approach also provides a reasonable average condition for estuary water but may also be difficult to maintain successfully where travel times between sites are great.

Based on data provided by ECan, we consider that the current approach of consistent sampling with respect to tide (sampling on high tide at the estuary and coastal sites, low tide at the two river mouths, and both low and high tide at Shag Rock) is appropriate for detecting long term change, and we note that this timing has been well-maintained over the now greater than 10 year time series. We would not advise changing the sampling timing as this would introduce a source of variation to long-term datasets that would make interpretation of trends difficult. We also observed by plotting date of sampling against time of sampling for each site (data not shown) that in general sampling has been strictly maintained to within a given time-of-day band for each site, though there has been a slight trend in sampling time at some sites. We recommend that ECan assess the trends in sampling times at each of their sites, and adjust future sampling times as required (i.e., to make sure they stay consistent over the time series) to prevent changes in sampling time from confounding interpretation of trends.

Sample collection

In general Zeldis et al. (2017b) and Zaiko et al. (2018) recommend use of NEMS (2020) methods for water quality sampling in coastal waters, as well as use of NEMS protocols with regard to metadata collection, reporting of measurement uncertainty, and quality coding. Use of NEMS protocols is particularly beneficial for national-scale reporting, where consistent methods across all regional authorities facilitates comparison of water quality across regions. Below, we give a brief comparison of methods currently used for water sampling (as provided with data supplied to NIWA in 2016) with those listed in NEMS. We note that NEMS suggests all sampling and measurement procedures should be fully documented in a Field and Office Manual (or equivalent). For some assays such as chlorophyll-*a* and nitrogen ions (e.g., ammoniacal-N and NOx-N), microbiological activity should be stopped soon after sampling to ensure consistency between sampling periods. Ice packs (e.g., slicka pads, as mentioned in CRC sample collection protocols provided to NIWA in 2017) can be insufficient for this task – particularly in summer. NEMS recommends immediate stabilisation of samples is carried out using crushed ice.

Field meters

In line with NEMS, we recommend that field records be regularly kept of field meter specifications, and calibration and validation details. Details on how to do this, including an example calibration form, are provided in (NEMS 2019).

Sampling point

The locations for field measurements and water sample collection are well marked in ECan's maps. Maintaining consistency or "stationarity" (NEMS 2020) in sampling point locations is important to reduce erroneous variation in time series of water quality measurements.

Some aspects of sample collection are not mentioned in the sampling details provided to NIWA; we suggest that NEMS recommendations for sampling depth (i.e. 30 cm below the water surface), are specified.

Laboratory measurements on water samples

Based on data provided to NIWA in 2016, laboratory methods for some analytes differed to those recommended in Table 5 of NEMS (2020). These analytes include turbidity, TSS, NOx-N, and ammoniacal-N. We recommend that where NEMS-recommended analytical methods can provide results comparable to previous methods, but with improved detection limits, the NEMS methods are considered. However, the necessity for lower detection limits depends to some extent on analyte concentrations at a given site. For example, if concentrations of nutrients are commonly below laboratory detection limits, this may restrict our ability to detect trends in water quality. In such a case, a method with a lower detection limit may be preferable.

The relatively high detection limit for faecal indicator bacteria analysis (10 MPN/100mL) means many measurements of enterococci were below detection. This made trend detection less powerful as the magnitude of trends in enterococci could not be calculated at most sites (section 3.4). If calculating trend magnitude is of high importance for ECan then we recommend bacterial analyses use methods with lower detection limits. Similarly, detection limits for metals analyses were relatively high compared to the concentrations present, and as a result of the many data points below the limit of detection, trend magnitudes could not be calculated. If possible, we would recommend metals analyses use methods with lower detection limits.
7.4.2 Monitoring for dilution modelling

As outlined in Section 6.2.2, the NPS-FM requires freshwater quality and quantity limits to be set with consideration of impacts on downstream water bodies (New Zealand Government 2017). The dilution modelling approach used in ETI tool 1 facilitates this process, because it permits calculation of bands of N-loading to estuaries that correspond with bands of estuarine trophic condition (Dudley & Plew 2017, Plew et al. 2018b, Zeldis et al. 2017c). In addition to the water quality information requirements for this approach laid out above, the following data are required for estuary-by-estuary assessment of N-load bandings using ETI tool 1:

- tidal prism of the estuary at spring tide (i.e. the difference in volume of water in an estuary between spring high tide and spring low tide);
- volume of the estuary at spring high tide;
- mean annual freshwater inflow to the estuary;
- volume-averaged salinity at high tide to calculate dilution;
- salinity of ocean water outside the estuary; and
- intertidal area.

The DELFT3D/DELWAQ model constructed for the Ihutai/Avon-Heathcote Estuary and described in Section 6.4 has similar data requirements. Of the data requirements above, only regular measurements of freshwater inflow rates and nutrient concentrations (used to calculate contaminant loads) from city drains and stormwater outflows could be added to improve modelling outcomes.

7.5 Additional information and metadata to explain patterns of water quality and ecological issues

As previously mentioned, interactions between climate and nutrient loading from land via rivers and drains are likely to control the ecological state of Ihutai/Avon-Heathcote Estuary. A key response to these drivers is build-up of nuisance macroalgae in the intertidal zones of estuaries (Bolton-Ritchie 2020, Plew et al. 2020). Macroalgae could also be influencing persistence of faecal indicator bacteria as high concentrations of these have been found associated with algal mats in other locations. Monitoring of macroalgal cover, nutrient content and biomass of nuisance macroalgae provides vital data to measure this response. However, these data are laborious and expensive to collect. Remote sensing technologies may provide a lower-cost option to collect data on macroalgae populations at high spatial and temporal resolution. We would recommend investigating this possibility.

Water temperature and turbidity are useful supporting variables to measure alongside microbiological water quality indicators. In addition, the collection of metadata, notably tidal height and state, rainfall, and wind direction and intensity, are important for meaningful interpretation of the microbiological water quality data. We note that all of these variables have been recorded alongside microbial monitoring in Ihutai/Avon-Heathcote Estuary since 2007 (Zeldis et al. 2017b). Regular (at least five-yearly) catchment assessments to check the condition of urban infrastructure and changes in land use are also important in understanding and managing risks to human health from microbiological contamination (MfE/MoH 2003).

7.6 Summary

The existing ECan estuary water quality monitoring programme is providing robust data for the purposes of general SoE monitoring. Data (such as water temperature, pH and dissolved oxygen) collected through this programme are very useful for deducing the reasons for changes in ecological state of the estuary. We see few areas where the water quality monitoring programme could be easily improved. However, we would suggest ECan at least considers adding visual clarity measurements to regular monitoring and using crushed ice to rapidly halt biological processes in water samples after collection if this is not currently done. This latter change is particularly important for nutrient and chlorophyll-*a* analyses, and most necessary in summer. In addition, we suggest considering laboratory analytical methods with lower detection limits for faecal indicator bacteria and metals, if a priority is to detect trends in these analytes through time.

Trend analyses in this study showed recent increases in water temperature and water column chlorophyll-*a*. Similarly, benthic macroalgal biomass appears to have increased in recent years (Bolton-Ritchie 2020). Estimates of nutrient loading, and estuary nutrient concentration measurements do not suggest that increases in nitrogen availability alone is driving increases in algal growth. Instead, we suggest that nutrient loading from the upstream catchment may be interacting with changes in climate to alter trophic state of the estuary. Because increases in primary production (e.g., due to increased water temperature and algal biomass) will draw down seawater nutrients, we would encourage a focus on accurate measurement of nitrogen loading to the estuary to explain changes in nitrogen availability through time. The portion of the nitrogen load we are least sure of, currently, is that coming from drains near the wastewater treatment ponds. Our current estimate is that this forms around one third of the total nitrogen load from the upstream catchment.

A key response to changes in nutrient availability and climate is build-up of nuisance macroalgae in the intertidal zones of Ihutai/Avon-Heathcote Estuary. We see regular monitoring of macroalgal populations as a cornerstone of managing the heath of this estuary.

7.7 Recommendations

We recommend, in order of highest to lowest priority:

- 1. Retention of existing estuarine water quality sampling site locations to maintain the high-value of water quality time series.
- 2. Focus on collecting regular flow, nutrient and faecal indicator bacteria data from drains near the wastewater treatment ponds from the city to the estuary, as well as continuation of the current monitoring of terminal river reach sites and Canterbury's open coastal waters.
- 3. Regular monitoring of intertidal macroalgal populations as a key 'bioindicator' of biological response to climate and nutrient conditions in the estuary.
- 4. Consideration of metals and bacterial analyses methods with lower detection limits to improve trend detection.
- 5. Using crushed ice to rapidly halt biological processes in water samples after collection.
- 6. Consideration of adding visual clarity measurements at Ihutai/Avon-Heathcote Estuary monitoring sites.

8 Summary of key questions

In this section we summarise the answers to the eight key questions ECan asked to be addressed in this water quality assessment.

1. Are parameter concentrations/values at each monitored site changing (increasing or decreasing), i.e. are there trends over time, or are parameter values constant over time (within the realms of natural variability)? If they are increasing or decreasing, please identify the direction of the trend and quantify the trend. The main focus is to be on the most recent 6-year period (Jan 2014-Dec 2019), however, where possible also complete trend analysis for the complete time period.

This question was addressed in section 3. Nutrient concentrations (almost all forms at all sites) and suspended solids decreased over time from 2007 to 2019, whereas other variables including indicator bacteria, showed increases at some sites and decreases at others. In the most recent period (2014-2019) there are likely further decreases in nutrient concentrations at most sites, but chlorophyll-*a* has increased at all sites. Enterococci have also increased at all sites.

2. Determine the changes in parameter concentrations/values as a result of the removal of wastewater discharge from the estuary in March 2010.

This question was addressed in section 3.3. Nutrient and volatile suspended solids concentrations decreased significantly after the wastewater was diverted, except for NOx-N at the $\bar{O}p\bar{a}waho/Heathcote$ River and nearby sites and at the estuary mouth. Chlorophyll-*a* also decreased at almost all sites. Other variables showed little change.

3. What is the likely impact of the measured nutrient, oxygen, turbidity, TSS, temperature, metals, and faecal indicator bacteria concentrations on the ecological functioning (ecosystem health) of the estuary and human health? That is, assess measured values against relevant guideline values for ecosystem and human health. Consider macroalgae growth and hence trophic state and seafood safe to eat.

This question was addressed in sections 3.6 and 5. The concentrations and loads of nutrients currently entering the estuary may be leading to eutrophication. Indicator bacteria concentrations suggest that sites near the river mouths and in the estuary have higher illness and infection risks for swimming and shellfish gathering; except the Beachville Road site where there are low risks for shellfish gathering; and the coastal sites have low illness and infection risks for swimming.

4. Are seasonal patterns in parameter concentrations changing?

This question was addressed in section 3.6. The seasonal patterns do not appear to have changed but have been affected by sampling during flood events and unusually low concentrations on some occasions.

5. The estuary supports a diversity and abundance of birds. What contribution are the birds likely making to the nutrient and micro-organism concentrations and hence water quality within the estuary?

This is addressed in section 6.3 and shows that it is likely that addition to nutrients by birdlife is likely minor but more important for microbes. Due to their high numbers, birds do contribute to

the nutrients in the estuary (2-5% of the total load), and to the bacteria counts (up to 100%), but it is not clear how this affects the estuary water quality overall.

6. Identify the possible current drivers of water quality issues in the estuary and the likely ecological effects of these.

We examined drivers in section 6 and found that the rivers (mainly), drains, birds and residence time may be affecting water quality. In addition, changes to water temperature may be affecting macroalgal growth, which in turn influences water quality.

7. An assessment of the adequacy of the current water quality monitoring programme for measuring state and trend in the estuary, and for identifying issues as they arise.

The current monitoring programme (reviewed in section 7) is largely fit-for-purpose with a suitable number of sites and frequency and variables measured.

8. Recommendations for further data sets required to investigate issues further.

In addition to the recommendations made regarding the monitoring programme (sections 7.7), we suggest the following:

- Regular monitoring of intertidal macroalgal populations
- Investigate the water quality and quantity of the drains near Sandy Point to determine the source(s) of high DRP and ammoniacal-N.
- Extend the existing DELFT3D/DELWAQ model to include non-conservative processes (bacterial die-off, resuspension from sediment) and model the inputs of bacteria, including from bird deposits if the necessary spatial information can be obtained.

9 Acknowledgements

Thanks to Katie Noakes and Winsome Marshall of Christchurch City Council for the supply of water quality monitoring data for the streams of Christchurch City.

We acknowledge Dr John Zeldis, Juliet Milne, and Dr Rebecca Stott (all NIWA) for providing reviews and technical guidance which greatly improved this report.

Thanks to Lesley Bolton-Ritchie and Michele Stevenson of Environment Canterbury for their useful reviews.

10 Glossary of abbreviations and terms

| Ammoniacal-N | A measure of ammonia dissolved in water, which includes the two forms present in water: free ammonia (NH ₃) and the ammonium ion (NH ₄ ⁺); and expressed as the concentration of nitrogen. |
|-----------------------|--|
| Chlorophyll- <i>a</i> | Chlorophyll- <i>a</i> is a measure of the green algae (phytoplankton) in the water, which grows in the presence of nitrogen and phosphorus. |
| cfu | Colony forming units, a measure of the number of bacteria in a water sample |
| DO | Dissolved oxygen, oxygen that is dissolved within water |
| DRP | Dissolved reactive phosphorus, a measure of the forms that are available for aquatic plant growth |
| ETI | Estuary trophic index |
| MPN | Most probable number, a measure of the number of bacteria in a water sample |
| NH4-N | Abbreviation for ammoniacal-nitrogen |
| NOx-N | Abbreviation for nitrate+ nitrite nitrogen. The sum of two forms of nitrogen found dissolved in water: nitrate (NO ₃ ⁻) and nitrite (NO ₂ ⁻); expressed in mg of nitrogen in the water |
| Residence time | The length of time a particle or piece of water stays within a water body. Estuaries that flush rapidly have a short residence time. |
| TN | Total nitrogen, all the forms of nitrogen in a water sample, including dissolved and organic forms (e.g., inside degrading biological material) |
| ТР | Total phosphorus, all the forms of phosphorus in a water sample, including that stuck to sediment and dissolved forms. |

11 References

ANZECC (2000). Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand *Volume 1: The guidelines*.

ANZG. (2018). Australian and New Zealand guidelines for fresh and marine water quality. from <u>https://www.waterquality.gov.au/anz-guidelines/</u>

Barr, N., Zeldis, J., Scheuer, K., Schiel, D. (2019). Macroalgal Bioindicators of Recovery from Eutrophication in a Tidal Lagoon Following Wastewater Diversion and Earthquake Disturbance. *Estuaries and Coasts*: 1-16.

Barr, N., Zeldis, J.R., Gongol, C., Drummond, L., Scheuer, K. (2012). Effects of the Canterbury earthquakes on Avon-Heathcote Estuary / Ihutai macroalgae. Environment Canterbury *Technical Report Investigations and Monitoring Group*. Christchurch.

Barr, N.G. (2007). *Aspects of nitrogen metabolism in the green alga Ulva: developing an indicator of seawater nitrogen loading*. PhD thesis. Auckland University.

Barr, N.G., Dudley, B.D., Rogers, K.M., Cornelisen, C.D. (2013). Broad-scale patterns of tissue- δ^{15} N and tissue-N indices in frondose Ulva spp., Developing a national baseline indicator of nitrogenloading for coastal New Zealand. *Marine pollution bulletin 67(1)*: 203-216.

Batcheler, L., Bolton-Ritchie, L., Bond, J., Dagg, D., Dickson, P., Drysdale, A., Handforth, D., Hayward, S. (2009). *Healthy Estuary and Rivers of the City: Water quality and ecosystem health monitoring programme of Ihutai*. Environment Canterbury Report No RO9/8. 60pp.

Bolton-Ritchie, L. (2015). *The sediments and biota within the estuary of the Heathcote and Avon Rivers/Ihutai 2007-2013*. Environment Canterbury Regional Council.

Bolton-Ritchie, L. (2019). Water quality of the Estuary of the Heathcote and Avon Rivers/Ihutai. Summary report on data collected in 2018. Environment Canterbury *Healthy Estuary and Rivers of the City. Water quality and ecosystem health monitoring programme of Ihutai.* . Environment Canterbury, Christchurch.

Bolton-Ritchie, L. (2020). *Macroalgae abundance and trophic state of the Estuary of the Heathcote and Avon Rivers/Ihutai in the summer of 2019/2020*. Unpublished Environment Canterbury Technical Report.

Bolton-Ritchie, L., Main, M. (2005). *Nutrient water quality Avon-Heathcote Estuary/Ihutai, Inputs, concentrations and potential effects*. Environment Canterbury Report No. U05/71.

Burge, P. (2007). Sedimentation in the Avon-Heathcote Estuary/Ihutai–an analysis of past and present studies. Report prepared for Environment Canterbury Regional Council and the Avon-Heathcote Estuary Ihutai Trust.

Cardoso, P., Pardal, M., Lillebø, A., Ferreira, S., Raffaelli, D., Marques, J. (2004). Dynamic changes in seagrass assemblages under eutrophication and implications for recovery. *Journal of Experimental Marine Biology and Ecology 302(2)*: 233-248.

Cornelisen, C.D. (2010). *Recommendations for monitoring the coastal marine area in the Manawatu-Wanganui Region*. Prepared for Horizons Regional Council. Report No. 1752. 40 pp.

Crossland, A.C. (2013). Wetland bird monitoring at the Avon-Heathcote Estuary and Bromley Oxidation Ponds, Christchurch: August 2009 to July 2010. *Notornis 60(2)*.

Davies-Colley, R., Smith, D. (2001). Turbidity suspended sediment, and water clarity: a review JAWRA Journal of the American Water Resources Association 37(5): 1085-1101.

Dudley, B., Jones-Todd, C. (2018). New Zealand Coastal Water Quality Assessment Update. Ministry for the Environment *NIWA Client Report 2018096CH*. Ministry for the Environment.

Dudley, B.D., Plew, D., Whitehead , A., Zeldis, J. (2019). Canterbury coastal and harbour water quality investigation. *NIWA Client Report 2019053CH*. Report to Canterbury Regional Council.

Dudley, B.D., Plew, D.R. (2017). Te Akaaka eutrophication susceptibility assessment. NIWA report to Environment Canterbury. *NIWA Client Report No: 2017041CH*.

Dudley, B.D., R. Burge, O., Plew, D., Zeldis, J. (2020). Effects of agricultural and urban land cover on New Zealand's estuarine water quality. *New Zealand Journal of Marine and Freshwater Research*: 1-21.

Environment Canterbury (2011). Canterbury Natural Resources Regional Plan. Environment Canterbury Regional Council *Environment Canterbury Report 11/19*. Christchurch.

Environment Canterbury (2012). *Regional Coastal Environment Plan*. Environment Canterbury Regional Council. Christchurch.

Environment Canterbury (2017). *Canterbury Land and Water Regional Plan*. Environment Canterbury Regional Council. Christchurch.

Foley, M. (2018). Updating the methodology used to calculate overall water quality scores at marine water quality sites in the Auckland region. Auckland Council *Auckland Council Report 2018/027*. Auckland.

Gibson, K., Marsden, I.D. (2016). Seagrass *Zostera muelleri* in the Avon-Heathcote Estuary/Ihutai, summer 2015–2016. University of Canterbury, Christchurch *Estuarine Research Report 44 44*. 40 p. Griffiths, R. (2016). Recommended coastal water quality standards for Northland. Northland Regional Council *Northland Regional Council report*. Whangarei.

Hahn, S., Bauer, S., Klaassen, M. (2007). Estimating the contribution of carnivorous waterbirds to nutrient loading in freshwater habitats. *Freshwater Biology* 52(12): 2421-2433.

Hahn, S., Bauer, S., Klaassen, M. (2008). Quantification of allochthonous nutrient input into freshwater bodies by herbivorous waterbirds. *Freshwater Biology* 53(1): 181-193.

Helsel, D.R. (2012). Statistics for censored environmental data using Minitab and R. Wiley, New York.

Hewitt, J.E., Hatton, S., Safi, K.A., Craggs, R. (2001). Effects of suspended sediment levels on suspension-feeding shellfish in the Whitford embayment. Auckland Regional Council. *NIWA Client Report*.

Hollever, J., Bolton-Ritchie, L. (2016). *Broad scale mapping of the Estuary of the Heathcote and Avon Rivers/Ihutai*. Environment Canterbury, Regional Council: Christchurch, New Zealand.

Hume, T. (2018). Fit of the ETI trophic state susceptibility typology to the NZ coastal hydrosystems classification. *NIWA Client Report*.

Larned, S., Hamilton, D., Zeldis, J., Howard-Williams, C. (2011). *Nutrient-limitation in New Zealand rivers. lakes and estuaries*. 19 pp.

Larned, S., Snelder, T., Unwin, M., McBride, G., Verburg, P., McMillan, H. (2015). Analysis of Water Quality in New Zealand Lakes and Rivers. *NIWA Client Report.*

Larned, S.T., Whitehead, A., Snelder, T., Fraser, C., Yang, J. (2018). Water quality state and trends in New Zealand rivers. Ministry for the Environment *NIWA Client Report*.

Marion, L., Clergeau, P., Brient, L., Bertru, G. (1994). The importance of avian-contributed nitrogen (N) and phosphorus (P) to Lake Grand-Lieu, France. *In*: Aquatic Birds in the Trophic Web of Lakes: 133-147. Springer.

Marshall, W., Noakes, K. (2019). Surface water quality monitoring report for Christchurch City waterways: January - December 2018. Christchurch City Council and Aquatic Ecology Limited *Christchurch City Council Report*. Council, C.C., Christchurch, New Zealand.

McBride, G. (2005). Using statistical methods for water quality management: issues, problems and solutions. *Wiley Series In Statistics In Practice*. John Wiley & Sons Inc, New Jersey.

McBride, G., Yalden, S., Milne, J.R. (2019). National microbiological water quality guidelines for marine recreational areas: Implications from a review of recent research. *NIWA Client Report* 2018-333HN: 93.

McBride, G.B. (2019). Has Water Quality Improved or Been Maintained? A Quantitative Assessment Procedure. *Journal of Environmental Quality 48*: 412-420.

Measures, R. (2016). New River Estuary Hydrodynamic Modelling: Model build, calibration and tracer simulations. Environment Southland *NIWA Client Report*.

Measures, R., Bind, J. (2013). Hydrodynamic model of the Avon Heathcote Estuary: model build and calibration. Environment Canterbury *NIWA Client Report*.

Measures, R., Hicks, M., Shankar, U., Bind, J., Arnold, J., Zeldis, J. (2011). Mapping earthquake induced topographical change and liquefaction in the Avon-Heathcote Estuary. Prepared for Environment Canterbury and Christchurch City Council. NIWA *NIWA Client Report*. Christchurch.

Ministry for the Environment/Ministry of Health (2003). *Microbiological water quality guidelines for marine and freshwater recreational areas*. Ministry for the Environment, Wellington, New Zealand.

Moriarty, E.M., Gilpin, B.J. (2015). *Faecal Sourcesin the Avon River/Otakaro, Heathcote River/Opāwaho and the Estuary of the Heathcote & Avon Rivers/Ihutai 2015*. Prepared for Environment Canterbury, Community & Public Health, Christchurch City Council and the Ministry of Health. ESR. ESR, Christchurch.

Moriarty, E.M., Karki, N., Mackenzie, M., Sinton, L.W., Wood, D.R., Gilpin, B.J. (2011). Faecal indicators and pathogens in selected New Zealand waterfowl. *New Zealand Journal of Marine and Freshwater Research* 45(4): 679-688.

Morrison, M., Jones, E.G., Consalvey, M., Berkenbusch, K. (2014). *Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge*. Ministry for Primary Industries.

Morrison, M.A., Lowe, M., Parsons, D., Usmar, N., McLeod, I. (2009). A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand aquatic environment and biodiversity Report 37*: 100.

NEMS (2019). *National Environmental Monitoring Standards for Discrete Water Quality*, Part 4 of 4: Sampling, Measuring, Processing and Archiving of Discrete Coastal Water Quality Data New Zealand. Ministry for the Environment.

New Zealand Government (1991). Resource Management Act. Wellington, New Zealand.

New Zealand Government (2017). *National Policy Statement for Freshwater Management 2014, amended 2017*. Wellington, New Zealand.

Orchard, S., Measures, R. (2016). *Development of a fine-scale salinity model for the Avon Heathcote Estuary Ihutai*. Resilient Shorelines Report Prepared for Brian Mason Scientific and Technical Trust. University of Canterbury and NIWA. Christchurch.

Patston, L. (2019). Our Changing Oceans <u>https://www.niwa.co.nz/news/our-changing-oceans</u>.

Plew, D., Dudley, B.D. (2018). Eutrophication susceptibility assessment of Pounawea (Catlins) Estuary. Prepared for Otago Regional Council. *NIWA Client Report* No. 2018232CH.

Plew, D., Dudley, B.D., Shankar U., Zeldis, J. (2018a). *Assessment of the eutrophication susceptibility of New Zealand Estuaries*. *NIWA Client Report* to Ministry for the Environment: 2018206CH.

Plew, D.R., Dudley, B.D., Bind, J. (2017). Canterbury region estuary eutrophication susceptibility assessment. *NIWA Client Report* to Environment Canterbury: 2017154CH.

Plew, D.R., Zeldis, J.R., Dudley, B.D., Whitehead, A.L., Stevens, L.M., Robertson, B.M., Robertson, B.P. (2020). Assessing the eutrophic susceptibility of New Zealand Estuaries. *Estuaries and Coasts* <u>https://doi.org/10.1007/s12237-020-00729-w</u> in press.

Plew, D.R., Zeldis, J.R., Shankar, U., Elliott, A.H. (2018b). Using Simple Dilution Models to Predict New Zealand Estuarine Water Quality. *Estuaries and Coasts*.

Robertson, B., Stevens, L., Robertson, B., Zeldis, J., Green, M., Madarasz-Smith, A., Plew, D., Storey, R., Hume, T., Oliver, M. (2016a). NZ Estuary Trophic Index Screening Tool 1. Determining eutrophication susceptibility using physical and nutrient load data. *Envirolink Tools Project*.

Robertson, B., Stevens, L., Robertson, B., Zeldis, J., Green, M., Madarasz-Smith, A., Plew, D., Storey, R., Hume, T., Oliver, M. (2016b). NZ Estuary Trophic Index Screening Tool 2. Determining Monitoring

Indicators and Assessing Estuary Trophic State. *Prepared for Envirolink Tools Project: Estuarine Trophic Index, MBIE/NIWA Contract No: C01X1420.*

Robertson, B.P., Gardner, J.P.A., Savage, C. (2015). Macrobenthic–mud relations strengthen the foundation for benthic index development: A case study from shallow, temperate New Zealand estuaries. *Ecological Indicators 58*: 161-174.

Schernewski, G. (2003). Nutrient budgets, dynamics and storm effects in a eutrophic, stratified Baltic Lake. *Acta hydrochimica et hydrobiologica 31(2)*: 152-161.

Schiel, D.R., Lilley, S.A., South, P.M., Coggins, J.H.J. (2016). Decadal changes in sea surface temperature, wave forces and intertidal structure in New Zealand. *Marine Ecology Progress Series 548*: 77-95.

Snelder, T., Fraser, C. (2019). *The LWP_Trend libary version 1901: LWPTrends_V1901.r*. Christchurch, New Zealand, LWP Ltd.

Snelder, T.H., Larned, S.T., McDowell, R.W. (2018). Anthropogenic increases of catchment nitrogen and phosphorus loads in New Zealand. *New Zealand Journal of Marine and Freshwater Research 52(3)*: 336-361.

Snelder, T.H., Whitehead, A.L., Fraser, C., Larned, S.T., Schallenberg, M. (2020). Nitrogen loads to New Zealand aquatic receiving environments: comparison with regulatory criteria. *New Zealand Journal of Marine and Freshwater Research*.

Stevenson, M., Wilks, T., Hayward, S. (2010). An overview of the state and trends in water quality of Canterbury's rivers and streams. Environment Canterbury Regional Council *Environment Canterbury Report R10/117*. Christchurch.

Stocker, T., Qin, D.Q., Plattner, G.-K. (eds). (2014). *Climate Change 2013: The Physical Science Basis: Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press.

Whitehead, A.L., Booker, D.J. (2018). Communicating biophysical conditions across New Zealand's rivers using an interactive webtool. *New Zealand Journal of Marine and Freshwater Research*: 1-10.

Zaiko, A., Berthelsen, A., Cornelisen, C., Clark, D., Bulmer, R., Hewitt, J., Stevens, L., Stott, R., McBride, G., Hickey, C., Banks, J., Hudson, N. (2018). Managing Upstream: Estuaries State and Values – Methods and Data Review. Stage 1B report Prepared for The Ministry for the Environment. *NIWA Client Report No*: 2017415HN. p.

Zeldis, J., Storey, R., Plew, D., Whitehead, A., Madarasz-Smith, A., Oliver, M., Stevens, L., Robertson, B., Dudley, B. (2017a). The New Zealand Estuary Trophic Index (ETI) Tools: Tool 3 - Assessing Estuary Trophic State using a Bayesian Belief Network. From <u>https://shiny.niwa.co.nz/Estuaries-Screening-Tool-3/</u>

Zeldis, J., Whitehead, A., Plew, D., Madarasz-Smith, A., Oliver, M., Stevens, L., Robertson, B., Storey, R., Burge, O., Dudley, B. (2017b). The New Zealand Estuary Trophic Index (ETI) Tools: Tool 2 - Assessing Estuary Trophic State using Measured Trophic Indicators. From https://shiny.niwa.co.nz/Estuaries-Screening-Tool-2/

Zeldis, J., Whitehead, A., Plew, D., Madarasz-Smith, A., Oliver, M., Stevens, L.M., Robertson, B., Storey, R., Burge, O., Dudley, B. (2017c). The New Zealand Estuary Trophic Index (ETI) Tools: Web Tool 1 - Determining Eutrophication Susceptibility using Physical and Nutrient Load Data. *Ministry of Business, Innovation and Employment Envirolink Tools Contract: C01X1420*

Zeldis, J.R., Depree, C., Gongol, C., South, P.M., Marriner, A., Schiel, D.R. (2019). Trophic indicators of ecological resilience in a tidal lagoon estuary following wastewater diversion and earthquake disturbance. *Estuaries and Coasts*: 1-17.

Appendix A Key questions addressed in this report

ECan requested that this report provide answers to the eight questions listed below:

- 1. Are parameter concentrations/values at each monitored site changing (increasing or decreasing), i.e. are there trends over time, or are parameter values constant over time (within the realms of natural variability)? If they are increasing or decreasing, please identify the direction of the trend and quantify the trend. The main focus is to be on the most recent 6-year period (Jan 2014-Dec 2019), however, where possible also complete trend analysis for the complete time period.
- 2. Determine the changes in parameter concentrations/values as a result of the removal of wastewater discharge from the estuary in March 2010.
- 3. What is the likely impact of the measured nutrient, oxygen, turbidity, TSS, temperature, metals, and faecal indicator bacteria concentrations on the ecological functioning (ecosystem health) of the estuary and human health? That is, assess measured values against relevant guideline values for ecosystem and human health. Consider macroalgae growth and hence trophic state and seafood safe to eat.
- 4. Are seasonal patterns in parameter concentrations changing?
- 5. The estuary supports a diversity and abundance of birds. What contribution are the birds likely making to the nutrient and micro-organism concentrations and hence water quality within the estuary?
- 6. Identify the possible current drivers of water quality issues in the estuary and the likely ecological effects of these.
- 7. An assessment of the adequacy of the current water quality monitoring programme for measuring state and trend in the estuary, and for identifying issues as they arise.
- 8. Recommendations for further data sets required to investigate issues further.

Appendix B Seasonality analysis for trend assessment

 Table B-1:
 Results of seasonality analysis for data from 2007 to 2019.
 Seasonality is based on the results of the Kruskal-Wallis test: if p-value <0.05, data show seasonal variation.</th>

| | Ōtākaro / Avon River | Ōpāwaho / Heathcote River | South New Brighton Park | Humphreys Drive | Sandy Point | Penguin Street | Beachville Road | Shag Rock Iow | Shag Rock high | Cave Rock | Southshore Beach |
|---------------------------------------|-------------------------|---------------------------------|----------------------------------|--------------------|-------------|-------------------|--------------------|------------------|-------------------|------------|---------------------|
| Salinity | Nonseasona | l Seasonal | Seasonal | Nonseasonal | Nonseasona | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasonal | Seasonal | Nonseasonal |
| Water temperature | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Dissolved oxygen saturation | Seasonal | Seasonal | Seasonal | Nonseasonal | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal | Seasonal | Nonseasona | Il Nonseasonal |
| Dissolved oxygen concentration | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| рН | Seasonal | Seasonal | Seasonal | Seasonal | Nonseasona | Nonseasonal | Seasonal | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Ammoniacal-nitrogen | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasona | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Nitrate-N + nitrite-N (NOx-N) | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Dissolved Inorganic Nitrogen (DIN) | Seasonal | Seasonal | Seasonal | Nonseasonal | Nonseasona | l Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Total Nitrogen | Seasonal | Seasonal | Seasonal | Nonseasonal | Nonseasona | Nonseasonal | Seasonal | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Dissolved Reactive Phosphorus | Seasonal | Nonseasonal | Seasonal | Nonseasonal | Seasonal | Nonseasonal | Seasonal | Nonseasonal | Seasonal | Seasonal | Seasonal |
| Total Phosphorus | Nonseasona | Nonseasonal | Nonseasona | l Seasonal | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasonal | Seasonal | Nonseasonal |
| Chlorophyll-a | Seasonal | Seasonal | Nonseasona | l Nonseasonal | Nonseasona | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Total Suspended Solids (TSS) | Seasonal | Seasonal | Nonseasona | lNonseasonal | Nonseasona | Nonseasonal | Nonseasona | lNonseasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Volatile Suspended Solids (VSS) | Nonseasona | Nonseasonal | Nonseasona | lNonseasonal | Nonseasona | Nonseasonal | Nonseasona | lNonseasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Turbidity | Seasonal | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasona | Nonseasonal | Nonseasona | l Nonseasonal | Nonseasonal | Nonseasona | l Nonseasonal |
| Enterococci | Nonseasona | l Seasonal | Nonseasona | l Seasonal | Nonseasona | Nonseasonal | Nonseasona | l Nonseasonal | Seasonal | Nonseasona | l Nonseasonal |
| E. coli | Nonseasona | Nonseasonal | Seasonal | Seasonal | Seasonal | Seasonal | NM | Nonseasonal | NM | NM | NM |
| Note: NM: not measured at the | at particular site | 2. | | | | | | | | | |

Table B-2: Results of seasonality analysis for data from 2014 to 2019. Seasonality is based on the results of the Kruskal-Wallis test: if p-value <0.05, data show seasonal variation.</th>

| | Ōtākaro / Avon River | Ōpāwaho / Heathcote River | South New Brighton Park | Humphreys Drive | Sandy Point | Penguin Street | Beachville Road | Shag Rock Iow | Shag Rock high | Cave Rock | Southshore Beach |
|---------------------------------------|-------------------------|---------------------------------|----------------------------|--------------------|-----------------|-------------------|--------------------|------------------|-------------------|--------------|---------------------|
| Salinity | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasonal | Nonseasonal | Seasonal | Nonseasonal | Nonseasona | l Seasonal | Nonseasonal |
| Water temperature | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Dissolved oxygen saturation | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Nonseasonal | Seasonal | Nonseasonal | Seasonal | Nonseasona | l Nonseasonal |
| Dissolved oxygen concentration | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| рН | Seasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Nonseasona | l Nonseasona | INonseasonal |
| Ammoniacal-nitrogen | Seasonal | Nonseasona | Seasonal | Seasonal | Nonseasona | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Nonseasonal |
| Nitrate-N + nitrite-N (NOx-N) | Nonseasonal | Seasonal | Seasonal | Nonseasona | l Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Dissolved Inorganic Nitrogen (DIN) | Nonseasonal | Seasonal | Seasonal | Seasonal | Nonseasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Total Nitrogen | Nonseasonal | Seasonal | Seasonal | Nonseasona | Nonseasona | Seasonal | Seasonal | Seasonal | Nonseasona | l Seasonal | Nonseasonal |
| Dissolved Reactive | | | | | | | | | | | |
| Phosphorus | Seasonal | Seasonal | Seasonal | Seasonal | Nonseasona | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal | Seasonal |
| Total Phosphorus | Nonseasonal | Nonseasona | Seasonal | Seasonal | Nonseasona | Nonseasonal | Seasonal | Nonseasonal | Nonseasona | l Seasonal | Nonseasonal |
| Chlorophyll-a | Nonseasonal | Seasonal | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Seasonal | Nonseasona | l Nonseasonal |
| Total Suspended Solids (TSS) | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Nonseasona | l Nonseasona | l Nonseasonal |
| Volatile Suspended Solids | | | | | | | | | | | |
| (VSS) | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Nonseasona | Nonseasona | lNonseasonal |
| Turbidity | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Seasonal | Nonseasonal | Seasonal | Nonseasona | lNonseasonal |
| Enterococci | Nonseasonal | Seasonal | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Nonseasona | l Nonseasona | l Nonseasonal |
| E. coli | Nonseasonal | Nonseasona | Seasonal | Nonseasona | l Seasonal | Seasonal | NM | Nonseasonal | NM | NM | NM |
| Faecal coliforms | NM | NM | NM | NM | Seasonal | Seasonal | Nonseasona | al Nonseasonal | Seasonal | NM | NM |
| Total copper | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasonal | Nonseasonal | NA | Nonseasonal | NA | Nonseasona | l Nonseasonal |
| Total chromium | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | l Nonseasona | Nonseasonal | Nonseasona | al Nonseasonal | Nonseasona | l Nonseasona | l Nonseasonal |
| Total lead | Nonseasonal | Nonseasona | Nonseasona | I NA | Nonseasona | Nonseasonal | NA | NA | NA | Nonseasona | lNonseasonal |
| Total zinc | Nonseasonal | Nonseasona | Nonseasona | l Nonseasona | Nonseasonal | Nonseasonal | NA | Nonseasonal | NA | Nonseasona | lNonseasonal |
| Note: NM: not measured at that pa | articular site. NA | : trend could i | not be assesse | d due to high p | roportions of c | ensored data. | | | | | |



Appendix C Time series plots of variables not shown in section 3.2





Figure C-2: Dissolved oxygen concentrations and pH from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal) or coastal (green). Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.



Figure C-3: *E. coli* and faecal coliform bacteria from January 2007 to December 2019 at estuary monitoring sites. Sites are coloured by type: river (blue), estuary (teal). Coastal sites not monitored for these bacteria. Note that y-axis scales differ for each site to improve visibility of data over time. Dashed vertical line indicates date when wastewater discharge was diverted from the estuary to an ocean outfall. The grey shaded area indicates the time when there were temporary discharges of untreated wastewater into the Avon and Heathcote Rivers and directly into the estuary, due to earthquake damage.



Appendix D Water quality trends from 2014 to 2019

Trend likelihood - Extremely likely increase - Very likely increase - Likely increase - As likely to increase as decrease

Figure D-1: Changes in water temperature at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend).



Figure D-2: Changes in salinity at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes differ for each site.



Figure D-3: Changes in pH at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend).



Figure D-4: Changes in dissolved oxygen saturation at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes differ for each site.



Figure D-5: Changes in ammoniacal-N (NH₄-N) at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site. Line at Southshore Beach is horizontal as slope (trend magnitude) could not be calculated due to high proportion of censored data.



Figure D-6: Changes in NOx-N at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-7: Changes in DIN at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-8: Changes in total nitrogen at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-9: Changes in dissolved reactive phosphorus (DRP) at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-10: Changes in total phosphorus at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-11: Changes in chlorophyll-*a* (measured in µg/L) at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-12: Changes in total suspended solids (TSS) at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-13: Changes in volatile suspended solids (VSS) at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site. Many data were below the detection limit of 3 mg/L and are shown as open circles. Horizontal lines at bottom of plots indicate slope (trend magnitude) could not be calculated due to high proportions of censored data.



Figure D-14: Changes in turbidity at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site.



Figure D-15: Changes in enterococci at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site. Many data were below the detection limit of 10 MPN/100mL and are shown as open circles. Note that all trends for enterococci were assessed using nonseasonal trend methods.



Figure D-16: Changes in *E. coli* at all sites during January 2014 – December 2019. Trend lines are coloured by likelihood of trend (see legend). Note y-axes are log-scales and differ for each site. Many data were below the detection limit of 10 MPN/100mL and are shown as open circles.



Trend likelihood — As likely to increase as decrease — Likely decrease — Very likely decrease





Figure D-18: Changes in total copper at all sites during January 2014 – December 2019. Many data were below the detection limit and are shown as open circles. Note y-axes are log-scales and differ for each site. Trend lines are calculated from August 2016 to December 2019 and are coloured by likelihood of trend (see legend). Horizontal lines at bottom of plots indicate slope (trend magnitude) could not be calculated due to high proportions of censored data.



Figure D-19: Changes in total chromium at all sites during January 2014 – December 2019. Many data were below the detection limit and are shown as open circles. Note y-axes are log-scales and differ for each site. Trend lines are calculated from August 2016 to December 2019 and are coloured by likelihood of trend (see legend). Horizontal lines at bottom of plots indicate slope (trend magnitude) could not be calculated due to high proportions of censored data.



Figure D-20: Changes in total lead at all sites during January 2014 – December 2019. Many data were below the detection limit and are shown as open circles. Note y-axes are log-scales and differ for each site. Trend lines are calculated from August 2016 to December 2019 and are coloured by likelihood of trend (see legend). Horizontal lines at bottom of plots indicate slope (trend magnitude) could not be calculated due to high proportions of censored data.



Figure D-21: Changes in total zinc at all sites during January 2014 – December 2019. Many data were below the detection limit and are shown as open circles. Note y-axes are log-scales and differ for each site. Trend lines are calculated from August 2016 to December 2019 and are coloured by likelihood of trend (see legend). Horizontal lines at bottom of plots indicate slope (trend magnitude) could not be calculated due to high proportions of censored data.

Appendix E Methods used in this assessment

E.1 Methods for calculating nutrient loads from the rivers

We assessed nitrogen loads to the estuary and their likely effects on macroalgae as follows:

- 1. Mean flows at the terminal reaches of the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers were calculated from daily measured flows at sites upstream from tidal influences. These were adjusted by the ratio between modelled flow at the measurement site and modelled flow at the terminal river reach, where the modelled flows were taken from the NZ River Maps database (Whitehead & Booker 2018).
- 2. We used concentrations in the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers from monthly sampling at sites upstream from tidal influences as measured by CCC (Marshall & Noakes 2019). Concentrations were compared between these upstream sites and terminal reaches, and indicated concentrations were similar or lower at the terminal reaches and that there were no major inputs further downstream that would be omitted by using data from upstream.
- 3. For load calculations from rivers we first checked for relationships between flow and nutrient concentrations. We found no significant relationships, so we estimated the concentrations for each day based on the monthly data¹³ and multiplied those daily concentrations by the daily flows to generate daily riverine loads. These daily loads were then summed to get annual loads. It has been previously noted that NOx-N concentrations in the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers can be lower during rainfall, due to the dilution of high NOx-N groundwater with lower NOx-N runoff (M. Stevenson, pers. comm). If this is the case then the annual loads we calculated will somewhat over-estimate the load.
- 4. In addition to the river inputs, there are inputs from several drains contributing nutrients to the inner estuary. In the absence of recent flow and nutrient concentration data for these drains, we based our estimate of N loading from these sources using summaries in Burge (2007) and Bolton-Ritchie and Main (2005). We used the DIN loading estimates of Burge (2007), totalling 76,000 kg/y (estimated between 1992 and 1997), and adjusted them by flow in the Ōtākaro/Avon and Ōpāwaho/Heathcote rivers, to account for the likelihood that loads from drains in wet years were likely to be higher than in dry years.
- 5. We then used the CLUES-Estuary tool to estimate eutrophication susceptibility and assess the impacts of nutrient availability on the trophic state of Ihutai/Avon-Heathcote Estuary. These general methods followed those of Plew et al. (2020), using values for input parameters from (Plew et al. 2017), and inputs for riverine flow and nutrient loads described above.

¹³ We used a linear interpolation between the monthly monitoring data.

E.2 Methods for calculating nutrient and bacteria loads from birds

We acquired data on the number of birds using Ihutai/Avon-Heathcote Estuary from surveys undertaken by Andrew Crossland of the Christchurch City Council Parks Department (Crossland 2013). We used monthly count data to estimate an average number of birds at any one time.

We then used the model Waterbirds 1.1 (Hahn et al. 2007, Hahn et al. 2008) to estimate nutrient inputs of various bird species in the estuary. The model estimates nutrient outputs from 40 bird species, depending on each species' body mass, energy expenditure, food type and season. As this model was developed for freshwater habitats in Europe, we selected species closest to those found in the estuary and scaled the nutrient outputs from the model by the relative body mass for the species being represented. For example, we scaled the nutrient loads for great cormorants (average body mass of 2254 g) by 0.75 to represent the pied cormorant (average body mass 1700 g).

We estimated the input of faecal indicator bacteria from birds based on the same bird count data, and used estimates of daily microbial output of bacteria from Moriarty et al. (2011) which provided estimates for four common waterfowl: black swan, Canada goose, duck and gull. Again, we selected the most similar species and scaled the estimated daily microbial outputs for each of the four birds by the body weight of the waterfowl species found in Ihutai/Avon-Heathcote Estuary. From these calculations we estimated the average input of nutrients and bacteria from all bird species.

For calculation of bird contributions to nutrient loads, it is important to distinguish between "allochthonous" nutrients brought into the estuary from outside (e.g., consumption of vegetation from riverbanks, and subsequent release of those nutrients into the estuary), and "autochthonous" cycling of estuarine nutrients (e.g., consumption of small fish or invertebrates from within the estuary and subsequent release of nutrients from these food sources back into the estuary). Hahn et al. (2008) note that it is primarily herbivorous waterbirds that are responsible for importing nutrients from outside the estuary. Hence, for loading calculations we have only considered inputs from birds considered to be largely herbivorous. This could potentially over-estimate the nutrient loads as the birds may also be feeding on macroalgae, and therefore recycling estuarine nutrients.

E.3 Methods for three-dimensional modelling of estuary water quality

We modelled the estuary using a Delft3D/DELWAQ three-dimensional numerical model to understand the way the water from the rivers and sea mix within the estuary. This helps us understand which of the monitoring sites are affected most by the river inputs and therefore what could be driving water quality at these sites. Initially developed for ECan in 2013 (Measures & Bind 2013), the model extends up the Ōtākaro/Avon River to Gloucester Street, up the Ōpāwaho/Heathcote River to Buxton Terrace, and ~16.5 km offshore. The model uses a curvilinear grid with variable cell sizes to capture higher resolution in key areas, but lower resolution in other areas to speed up the model processing. Five vertical layers are included to capture density gradient effects. The model was calibrated to water level data for a 12 day period in April 2012. Additional details of model development and calibration are available in Measures and Bind (2013).

The estuary model was modified to include water particle tracers for the freshwater sources (Ōtākaro/Avon River, Ōpāwaho/Heathcote River, and Linwood drain only, not the drains near the wastewater treatment ponds as recent flow and nutrient data were not available for these) and ocean water. The model was run for ~90 days to generate an equilibrium initial condition, and for a subsequent 30 days to generate results and capture a spring-neap tide cycle. Input from freshwater sources was assumed to be average-low water flow and astronomic tide forcing was assumed at the offshore boundary. The estuary was divided into eight discrete zones (see Figure 6-4) with the average (both time and spatial) contribution of each water source computed for each zone. Estimates of nutrient loads for each estuary zone were calculated by multiplying water tracer concentrations (to indicate water contribution from each source) by observed source water nutrient concentrations using a processing method established by Measures (2016) for the New River Estuary in Southland. Note that the modelling analysis ignores non-conservative processes such as settlement of nutrients or nutrient cycling. Additionally, wind effects were neglected in the analysis.



Figure E-1: Zones used in modelling nutrient contributions of rivers, Linwood Drain, and ocean water to Ihutai/Avon-Heathcote Estuary.

E.4 Legislative context, environmental context and national initiatives for monitoring

This section provides a brief overview of key resource management legislation and recent national initiatives for consideration in the review of ECan's estuarine water quality monitoring programme.

New Zealand Coastal Policy Statement 2010

The New Zealand Coastal Policy Statement 2010 (NZCPS) is the principal document for managing coastal and estuarine waters. In terms of water quality, the primary NZCPS policies of relevance to the review of ECan's monitoring are:

Policy 21: Enhancement of water quality

Where the quality of water in the coastal environment has deteriorated so that it is having a significant adverse effect on ecosystems, natural habitats, or water-based recreational activities, or is restricting existing uses, such as aquaculture, shellfish gathering, and cultural activities, give priority to improving that quality by:

- (a) identifying such areas of coastal water and water bodies and including them in plans;
- (b) including provisions in plans to address improving water quality in the areas identified above;
- (c) where practicable, restoring water quality to at least a state that can support such activities and ecosystems and natural habitats;
- (d) requiring that stock are excluded from the coastal marine area, adjoining intertidal areas and other water bodies and riparian margins in the coastal environment, within a prescribed time frame; and
- (e) engaging with tangata whenua to identify areas of coastal waters where they have particular interest, for example in cultural sites, wāhi tapu, other taonga, and values such as mauri, and remedying, or, where remediation is not practicable, mitigating adverse effects on these areas and values.
- Policy 22: Sedimentation
 - (1) Assess and monitor sedimentation levels and impacts on the coastal environment.
 - (2) Require that subdivision, use, or development will not result in a significant increase in sedimentation in the coastal marine area, or other coastal water.
 - (3) Control the impacts of vegetation removal on sedimentation including the impacts of harvesting plantation forestry.
 - (4) Reduce sediment loadings in runoff and in stormwater systems through controls on land use activities.

National Policy Statement for Freshwater Management 2014

The National Policy Statement for Freshwater Management 2014 (NPS-FM, amended 2017 (New Zealand Government 2017)) sets out the objectives and policies for freshwater management under the Resource Management Act 1991. Its relevance to the review of ECan's estuarine monitoring is centred around Objective C1:

"To improve integrated management of fresh water and the use and development of land in whole catchments, including the interactions between fresh water, land, associated ecosystems and the coastal environment."

And Policy C1:

"By every regional council: a) recognising the interactions, ki uta ki tai (from the mountains to the sea) between fresh water, land, associated ecosystems and the coastal environment; and b) managing fresh water and land use and development in catchments in an integrated and sustainable way to avoid, remedy or mitigate adverse effects, including cumulative effects."

Regional policy

Coastal resource management issues and rules for the Canterbury region are set out in the Regional Coastal Environment Plan (RCEP) (Environment Canterbury 2012). Both the policies of the NZCPS and the NPS-FM (as well as other national regulations) are required to be given effect to by the RCEP. In terms of coastal quality, Volume 1 of the RCEP addresses coastal marine area (CMA) activities and water management. This includes Chapter 7 – which describes policies specifically relating to coastal water quality, water quality classes as set out in Schedule 4, and the areas where those classes are applicable as defined in Schedule 5. Areas where those classes are applicable are also marked on the Planning Maps in Volume 2 of the RCEP.

Environmental context

Water quality of New Zealand's estuaries is strongly affected by land use in upstream catchments; as urban land cover upstream from New Zealand's estuaries increases, concentrations of nutrients, faecal indicator bacteria and chlorophyll-*a* increase (Dudley et al. 2020). Agricultural intensification also strongly affects water quality of the freshwaters that feed New Zealand's estuaries (Snelder et al. 2018, Snelder et al. 2020). Future increases in Christchurch's population, expansion of its urban area and intensification of agriculture have the potential to result in poorer water quality flowing into Ihutai/Avon-Heathcote Estuary. Providing quantitative assessments of current and future impacts of land use on Ihutai/Avon-Heathcote Estuary requires data on pollutant loads from land, and the ocean with which these freshwaters mix in estuaries (Plew et al. 2020). Assessment of land use-change effects on water quality are, however, conducted against a background of medium- to longer-term climate variation (e.g., ENSO, Interdecadal Pacific Oscillation, climate change). As discussed above, New Zealand has experienced unprecedented marine heat waves in four out of five summers since 2016 (Patston 2019), with some of the highest temperature anomalies focused on the eastern South Island which may be affecting algal growth.

Recent national initiatives

Five key recent national initiatives of relevance to ECan's monitoring of Ihutai/Avon-Heathcote Estuary water quality and ecosystem health are outlined below.

Estuarine Trophic Index

The New Zealand Estuarine Trophic Index (ETI) project initiated by the regional sector's Coastal Special Interest Group (Coastal SIG) was completed in early 2017. The project produced three tools to assist regional councils in determining the susceptibility of an estuary to eutrophication, assess its current trophic state, and assess how changes to nutrient load limits may alter its current state. The tools determine estuary eco-morphological type, where an estuary sits along the ecological gradient from minimal to high eutrophication, and provide stressor-response tools (e.g., empirical relationships, nutrient models) that link the ecological expressions of eutrophication (measured using appropriate trophic state indicators) with nutrient loads (Robertson et al. 2016a, Robertson et al. 2016b, Zeldis et al. 2017a, Zeldis et al. 2017b, Zeldis et al. 2017c). The ETI tools are designed to meet the requirements of councils under the NPS-FM. With regard to Ihutai/Avon-Heathcote Estuary, these tools can make use of freshwater and marine water quality data collected under existing programmes to set nutrient load thresholds for estuary health. An assessment of this type has been conducted on Ihutai/Avon-Heathcote Estuary by Plew et al. (2017) and was updated in section 5 of this report based on the most recent calculated nutrient loads. Key information requirements for using the ETI tools are outlined in Section 0.

National Coastal Water Quality Assessment

In 2016, the Ministry for the Environment (MfE) commissioned NIWA to collate, review and analyse existing coastal water quality data gathered by the 16 regional and unitary authorities. The resulting report (Zeldis et al. 2017b) includes state and trend analyses of water quality variables most commonly used by councils for monitoring eutrophication, sedimentation and climate related long-term change. The report also provided recommendations for future analysis and reporting, including water quality thresholds, communication of trends, data quality, and uncertainty in water quality measurements. In addition, recommendations were made for improving monitoring networks at both regional and national levels. Methods developed in that report were used extensively in the present report on Ihutai/Avon Heathcote Estuary.

National Environmental Monitoring Standards (NEMS) for Water Quality

A NEMS addressing sampling and measuring of estuarine and coastal waters was released in February 2020. This document, together with additional documents for groundwater, rivers and lakes, establishes best practice for field measurements, water sample collection and laboratory testing.. NEMS is primarily focussed on long-term (e.g., State of the Environment - SoE) monitoring, making its contents highly relevant to the Healthy Estuary and Rivers of the City monitoring programme. The NEMS includes a process to assign a quality code to individual water quality measurements and aspects of sample collection, measurement and laboratory testing that have the potential to influence data quality. Thus, its practices should inform monitoring for Ihutai/Avon Heathcote Estuary.

Managing Upstream: Estuaries

"Managing Upstream: Estuaries State and Values" was an MfE commissioned project with the aim to better account for impacts on estuarine values when setting management objectives and freshwater limits under the NPS-FM. The NIWA and Cawthron-led project also sought to increase knowledge of the state of different estuary types in New Zealand. Stage 1 of the project included a recommended suite of state variables (e.g., for SoE monitoring), with a smaller subset of these (e.g., rate of sediment deposition) identified for potential estuarine attribute development¹⁴ (Zaiko et al. 2018). The recommended variables address the values of ecosystem health, human health for recreation and mahinga kai, and include a range of water quality, sediment quality and biological measures. The recommended water quality variables include nutrient concentrations (nitrogen, phosphorus), chlorophyll-*a*, dissolved oxygen, water clarity (e.g., Secchi disc), total suspended sediments, and faecal indicator bacteria.

National Microbiological Water Quality Guidelines

The Microbiological Water Quality Guidelines for Marine and Freshwater Recreational Areas (Ministry for the Environment/Ministry of Health 2003) form a pivotal reference for water quality management in New Zealand. A recent MBIE Envirolink Tool project initiated by the regional sector's

¹⁴ In the same way that the NPS-FM 2014 contains variables as attributes (e.g., chlorophyll-*a* and total nitrogen as measures of ecosystem health in lakes).

Coastal SIG in 2015/16 saw a review of four aspects of the marine component of the guidelines (McBride et al. 2019). Of particular relevance to this report are components 2 and 4; shellfish water quality guidance and the appropriate indicator(s) to use in brackish water bodies for SoE reporting and public health risk management, respectively. At present, the guidelines do not define a shellfish gathering season or how many samples should be collected, and do not specify whether *E. coli*, enterococci or faecal coliform indicator bacteria should be tested in brackish (e.g., estuarine) waters such as Ihutai/Avon-Heathcote Estuary, that are used for recreational purposes (McBride et al. 2019).