REPORT NO. 3884

ASSESSMENT OF EFFECTS OF THE NELSON NORTH WASTEWATER TREATMENT PLANT DISCHARGE ON COASTAL ECOLOGY AND KAIMOANA

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Prepared for Nelson City Council

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EXECUTIVE SUMMARY

Nelson City Council (NCC) holds consent SAR-05-61-01-06 to discharge treated wastewater from the Nelson North Wastewater Treatment Plant (NWWTP) to Tasman Bay via a 350 m-long buried pipe and 18 m-long multiport diffuser. The consent began in 2004 and runs until December 2024. NCC has engaged the Cawthron Institute (Cawthron) to assist with the preparation of an assessment of environmental effects of the NWWTP discharge. In this report, we present the results of a review of the ecological values of the receiving environment and an assessment of effects of the present and future discharge upon them.

The treatment plant lies on the seaward, northwest corner of an area of low-lying land created in the early 1900s by infilling of the upper parts of Nelson Haven. Land to the east of the plant consists of pasture divided by drainage ditches, while just to the west lie the sandflats of the upper Nelson Haven. The NWWTP is protected from the sea by the Boulder Bank. Based on information from aerial photography and remotely operated vehicle and diver surveys, patchy boulder habitat extends offshore from the Boulder Bank to the south and east (inshore) of the outfall, with continuous boulder field inshore and individual or patches of boulders in predominantly sandy areas further offshore. The seabed immediately around the outfall is rippled, medium sand, becoming predominantly small boulders beyond 60 m north of the outfall.

The additional nutrient load provided by the outfall is not expected to result in local nuisance growths of phytoplankton or macroalgae. Given that the generation time for coastal phytoplankton is normally in the range of a few days rather than hours, the rate of dilution as nutrients travel downstream from the discharge would be expected to preclude any measurable effluent-related enhancement of phytoplankton (including nuisance species). In the case of macroalgae, monitoring around the outfall has shown that populations are sparse (apart from encrusting corallines) and there is no evidence of increased abundance closer to the outfall. The discharge may contribute about 4% of the annual load of total nitrogen from terrestrial sources into Tasman Bay / Te Tai-o-Aorere, which is a significant amount. However, the overwhelming driver of nutrient concentrations in the bay is the intrusion of offshore waters, which is estimated to contribute 90% of dissolved inorganic nitrogen. Consequently, although terrestrial sources of nitrogen are important for coastal primary productivity, there seems to be little potential for over-enrichment (eutrophication).

Based on monitoring data, it is very unlikely that toxic contaminants in the wastewater, such as trace metals and ammonia, will be present in the receiving environment at concentrations resulting in significant adverse effects on aquatic life. The present rectangular mixing zone for the discharge extends 250 m north and south of the outfall diffuser, parallel to the Bolder Bank, and 100 m shoreward and seaward. The effluent is estimated to be diluted at the boundary of the mixing zone by a factor of more than 350 times under existing discharge flow conditions and 280 times under predicted flow conditions for 2059 (lowest 1%iles of discharge flow rates). Consequently, adverse effects on visual clarity, concentrations of total

suspended solids and biochemical oxygen demand are expected to be negligible beyond the discharge mixing zone.

The fauna of the sandy areas around the outfall includes common species typical of similar habitats in the wider bay. The fish-like New Zealand lancelet was recorded in sandy sediment around the outfall in a survey in 1998. Extensive macroalgal beds are notably absent from the length of the Boulder Bank, including at the outfall location. The fauna of the offshore boulders and the outfall structure consists of species typical of the wider area, with high densities of the anemone *Actinothoe albocincta* also present on the outfall structure.

Surveys of sandy and boulder habitats around the outfall have not found obvious adverse effects of the discharge, even though the discharge has been in operation since 1968. This applied to both the habitats and the communities of organisms living in or on them. From this earlier study, and the low concentrations of trace metals in the sediments measured during the present assessment, it is unlikely that continued operation of the outfall will result in measurable adverse ecological effects on boulder and sand habitats or their biota.

The lack of any detectable effect on organisms living around the outfall suggests that there will be no consequent effects on the abundance or type of invertebrate or macroalgal kaimoana available. Abundances are naturally low and the area does not appear to be targeted for collection. The fish fauna of the Boulder Bank also appears to be of low diversity and abundance, and, if anything, abundances may be higher around the outfall. The risks associated with consumption of contaminated shellfish are addressed in a separate health-risk assessment report.

For each of the ecological features present in the receiving environment, the risk of significant adverse effects is low. This assessment derives primarily from monitoring data (benthic habitats) but also from estimates of dilution (local effects of nutrients, biochemical oxygen demand and toxicants) and relative loads (effects of nutrients on the wider coastal area). Because potential adverse effects were predicted to be less than minor, no additional mitigation is recommended.

This assessment of low risk was also assumed to apply to any Threatened and At Risk taxa that could possibly be present (but have not been recorded to date) in the coastal zone around the outfall. While it is possible that some of these taxa may be more sensitive than others to habitat disturbance or to altered nutrient concentrations or salinities, a lack of relevant information makes it impossible to predict effects with certainty. Nonetheless, the outfall has been operating since 1968 (and under the present wastewater treatment regime since 2010), so additional future effects on the wider receiving environment are unlikely.

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ABBREVIATIONS

As	Arsenic
BOD	Biochemical oxygen demand
cfu	Colony forming units
DIN	Dissolved inorganic nitrogen
DRP	Dissolved reactive phosphorus
ENT	Enterococci
FC	Faecal coliform
FIB	Faecal indicator bacteria
g/m^3	Grams per cubic metre (parts per million)
ha	Hectare
km	Kilometre
m	Metre or metres
m/s	Metres per second
m^3/s	Cubic metres per second
mg/kg	Milligrams per kilogram (parts per million)
mg/L	Milligrams per Litre (parts per million)
mg/m^3	Milligrams per cubic metre (parts per billion)
MPN	Most probable number
N	Nitrogen
NCC	Nelson City Council
NIWA	National Institute of Water and Atmospheric Research
NO_3	Nitrate
NO_x	Sum of nitrite and nitrate
NWWTP	Nelson North Wastewater Treatment Plant
NZCPS	New Zealand Coastal Policy Statement
NZTCS	New Zealand Threat Classification System
TDC	Tasman District Council
TN	Total nitrogen
TP	Total phosphorus
WWNP	Draft Whakamahere Whakatū Nelson Plan
WWTP	Wastewater treatment plant

1. INTRODUCTION

1.1. Background

Operation of the wastewater outfall at the northeastern end of the Boulder Bank began in 1968 with the discharge of raw sewage to Tasman Bay / Te Tai-o-Aorere (hereafter Tasman Bay) (Barter & Forrest 1998). In early 1979, the Nelson North oxidation pond was commissioned, and wastewater treated by the pond was discharged via the outfall. In 1996, the addition of a longitudinal baffle to the pond increased the efficiency of the pond and reduced the concentration of pathogens in the wastewater. Due to continuing problems with pond operation, the plant was further upgraded in 2008 to allow greater flexibility in pond management (Bailey & Conwell 2010). The existing pond was converted to a facultative pond and a new maturation pond was added. In addition, a trickling filter pre-treatment facility was added to allow greater control over pond loadings, and the capacity of the pond system was increased to cope with predicted future population growth. In 2010, a wetland area was added to the system to provide additional final treatment.

The outfall consists of a buried pipe c. 350 m long. It emerges from the seabed at its offshore end as an 18 m-long multiport diffuser, anchored to the seabed and aligned perpendicular to the Boulder Bank in a water depth of 11 m (Barter & Forrest 1998).

1.2. Purpose and scope of this report

The existing consent to discharge treated wastewater (SAR-05-61-01-06) began in 2004 and runs until December 2024. Nelson City Council (NCC) has engaged the Cawthron Institute (Cawthron) to assist with the preparation of an assessment of environmental effects of the Nelson North Wastewater Treatment Plant (NWWTP) discharge. In this report, we present the results of a review of the ecological values of the receiving environment and an assessment of effects of the present and future discharge upon them.

Effects of microbial contaminants derived from wastewater are primarily of concern for human health, rather than for ecological receptors, and are addressed in a separate study (Hudson & Wood 2023). Potential effects of the discharge on seabirds and shorebirds are also addressed in a separate study.

2. THE RECEIVING ENVIRONMENT OF TASMAN BAY

2.1. Physical environment

2.1.1. Coastal landforms and vegetation

The western and eastern sides of Tasman Bay consist of narrow coastal strips backed by the hills and mountains of Wharepapa / Arthur Range, Bryant Range and Richmond Range. The southern shore of the bay borders the low-lying Waimea Plains. The land surrounding Tasman Bay is dominated by native vegetation in the upper parts of the catchments, with pasture (low- and high-producing grassland) near the coast and along rivers in the mid-catchments (Figure 1). There are also large areas of exotic forestry and horticulture in the Waimea Plains and around Motueka.

The largest watercourses entering Tasman Bay are the Waimea and Motueka Rivers. The catchment of the Waimea River covers 726 h and that of the Motueka 2019 ha (Tuckey et al. 2006). Average annual flows in the Waimea and Motueka Rivers are 27.5 m^3/s and 82.1 m^3/s, respectively (Booker & Whitehead 2017).

The NWWTP lies on the seaward, northwest corner of an area of low-lying land created in the early 1900s by infilling of the upper parts of Nelson Haven between Glen Road and what is now Boulder Bank Drive. Land to the east of the NWWTP consists of pasture divided by drainage ditches, while just to the west lie the sandflats of the upper Nelson Haven. The WWTP is protected from the sea by the Boulder Bank, which stretches 13 km from Glenduan to The Cut. The bank consists of granodiorite boulders derived from Mackay Bluff that are transported south by longshore drift (Johnston 2001).

In contrast to the more exposed cobble and boulder habitats on the seaward side of the Boulder Bank (described in Section 2.3), the tidal flats of the upper Nelson Haven provide a sheltered environment for intertidal assemblages of saltmarsh and sandy sediments. These are not discussed further because they are not part of the receiving environment of the discharge (i.e. Tasman Bay).

Boffa Miskell (2015), on behalf of NCC, characterised the coastal marine and adjacent terrestrial areas of Horoirangi / Drumduan and the Boulder Bank in terms of natural character. The Boulder Bank was rated of ‘High’ natural character (the second-highest ranking). In the context of Policy 13 of the New Zealand Coastal Policy Statement (Department of Conservation 2010), this qualified this coastal area as being of ‘Outstanding Natural Character’. The adjacent Wakapuaka area was ranked as Moderate–Low because of its highly modified character. Note that although ecological values form part of the process of assessing natural character, other features such as landforms, intactness, integrity and lack of built structures are also relevant.

Consequently, assessing natural character, and the potential effects of the discharge upon it, is beyond the scope of the present report and we have not attempted to do so.

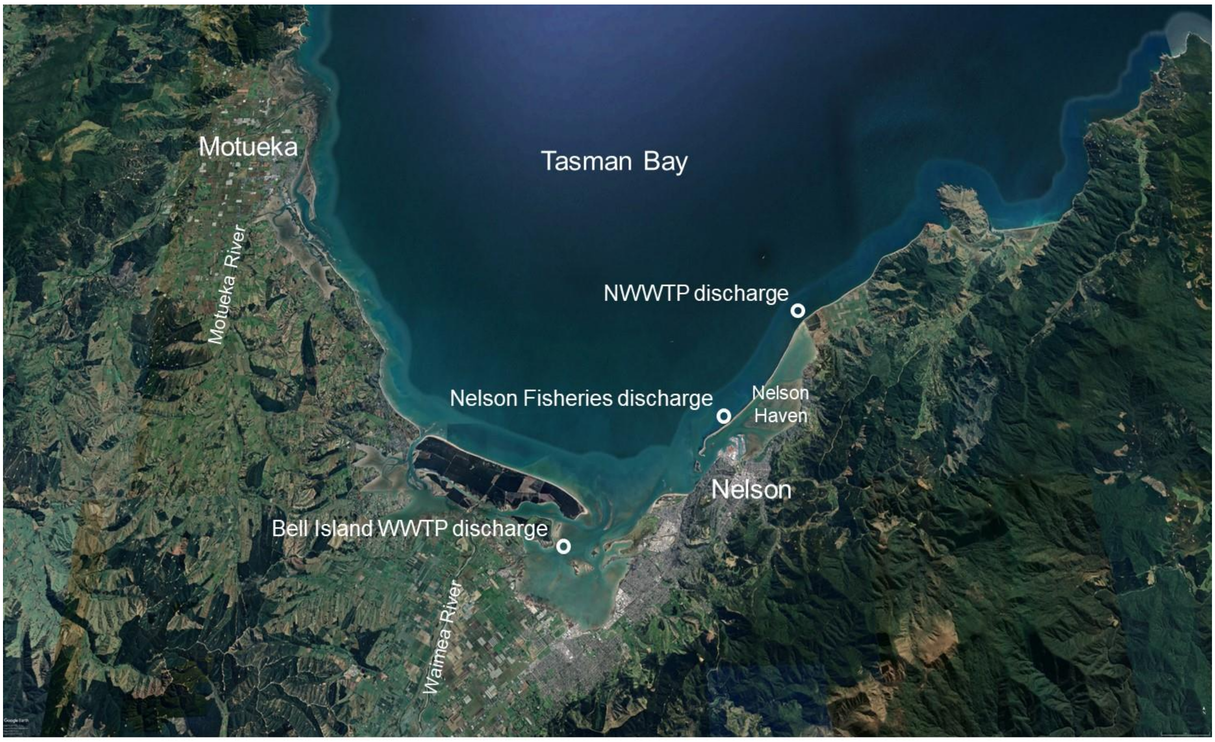


Figure 1. General location of the Nelson WWTP discharge and other wastewater discharges and main rivers in the catchments draining to Tasman Bay.

2.1.2. *Hydrodynamics*

Water movement in Tasman Bay is dominated by tidal flows, superimposed on weaker mean circulation (Heath 1976). Mean circulation is anticlockwise, with inflow around Farewell Spit, outflow on the eastern side and a return southwesterly flow on the coast near Nelson. In addition, surface flow is strongly, but variably, affected by wind. In contrast to Heath's description of circulation in Tasman Bay, Tuckey et al. (2006) identified a residual northward flow along the western side of the bay. Surface current speeds in Tasman Bay are typically 0.02–0.05 m/s (Heath 1976). The spring tidal range at Port Nelson is 3.77 m and the neap range 1.79 m.[1]

Description of the discharge mixing zone

The zone of reasonable mixing for the NWWTP discharge extends 100 m seawards and landward of the outfall diffuser and 250 m north and south of the diffuser, parallel to the Boulder Bank (Sneddon 2009). The mixing zone was proposed based on results

[1] See https://www.linz.govt.nz/sea/tides/tide-predictions/standard-port-tidal-levels (accessed 22 April 2020).

of a dye tracing and drogue[2] tracking study undertaken to determine the dispersion and dilution of the treated discharge in combination with analysis of outputs from CORMIX[3] modelling. In the dye study, qualitative (visual batch release) and quantitative (continuous low release) dye dispersion / dilution releases were conducted using Rhodamine WT dye. The batch release was used in conjunction with drogue releases to verify the path of the effluent plume, while the quantitative study was used to measure actual dilution rates of the discharge.

The results of the dye study indicated that, under an average discharge flow of 8,000 m^3/day, a minimum discharge dilution factor of approximately 100:1 is achieved directly over the boil and 500:1 is achieved within 250 m down-current of the outfall, with a dilution factor greater than 1,000:1 achieved offshore from the Schnappers Point surf break (approximately 1 km northeast of the outfall). These dilution factors were then used to evaluate receiving water quality and to derive receiving water and discharge quality consent conditions (Barter & Forrest 1998).

Further hydrodynamic modelling of the NWWTP discharge has been recently undertaken by MetOcean using a high-resolution 3D SCHISM model to inform the current consent application. The model simulations covered a range of climatic and discharge conditions to provide a more robust understanding of the dispersion of contaminants in the receiving environment (MetOcean Solutions Ltd 2022). To validate the model and assist with the characterisation of the hydrodynamic regime near the discharge point, Cawthron deployed a water quality monitoring buoy at a site approximately 250 m to the north of the outfall diffuser (at the boundary of the discharge mixing zone) to monitor a range of parameters (water currents, wave, salinity, temperature, oxygen, chlorophyll-*a*, turbidity). The dilution ratio predicted by the model under existing median discharge flow rates was 3,200:1 at the mixing-zone boundary[4]. This is a greater dilution than that earlier predicted by the CORMIX model (based on average flow). Under predicted future (2059) flow scenarios, a dilution of 2,300:1 is predicted for median flow rates and 280:1 for the lowest 1% of flow rates (1%ile).

2.1.3. Nutrients in the water column

Inputs of nutrients and other contaminants to Tasman Bay from land run-off are discussed in detail in an accompanying report (Campos 2023). They are briefly reviewed below.

[2] A drogue is a simple device that drifts with water currents and provides a measure of the direction and speed of water movement and hence the movement of the discharge plume.

[3] CORMIX (Cornell Mixing Zone Expert System) is a hydrodynamic mixing zone computer model that predicts the dispersion and dilution of a discharge plume in the near-field region under varying ambient current speeds. The model makes several assumptions and is not intended to be a substitute for *in situ* monitoring of the receiving environment but serves as a useful tool to help evaluate the potential effects of a discharge.

[4] Estimates of present and future dilutions at the mixing-zone boundary, based on MetOcean modelling, provided by Rob Lieffering, SLR, in an email to Ross Sneddon, 18 July 2023.

Nutrient inputs

The most important limiting nutrient (i.e. the nutrient that restricts plant growth) in Tasman Bay is nitrogen, in the form of dissolved inorganic nitrogen (DIN). Although inorganic phosphorus, iron and silica can also be limiting occasionally (MacKenzie 2004), these are unlikely to significantly constrain phytoplankton production in the bay.

Most land-derived nitrogen inputs to Tasman Bay are from rivers and streams (Campos 2023). The Motueka River alone contributes 60–70% of the input of 'new' (i.e. not generated by recycling within the bay) DIN from freshwater inputs to Tasman Bay (MacKenzie et al. 2003). Flood events in the catchment cause significant changes in the concentrations of DIN, phosphorus and silica in surface waters. MacKenzie et al. (2003) concluded that light is the main factor limiting phytoplankton production and nutrient assimilation in winter, and that seabed and water column remineralisation of organic matter plays an important role in generating annual nutrient concentration maxima within the water column of the bay.

In a study of inorganic nutrients in Tasman Bay in 1995–96, Mackenzie et al. (2003) showed seasonal variations in surface nitrate-nitrogen ranging from 3 mg/m^3 to 40 mg/m^3 in waters offshore from the Boulder Bank, with the highest concentrations recorded from late winter to spring. The spatial distributions of nutrients and their response to flood events in the bay are also strongly linked to riverine inputs.

Zeldis (2008) assessed the relative contributions of oceanic and freshwater nutrient sources in Tasman Bay and Golden Bay / Mohua (hereafter Golden Bay) using a theoretical nutrient budget approach. He calculated that around 90% of the DIN input is from the circulation of offshore waters into the bays. The model also suggested that water in Golden Bay is exchanged more frequently (approximately every 11 days) than in Tasman Bay (approximately every 41 days).

Quantification of nutrient discharges into Tasman Bay (from extensively modified catchments and point-source discharges) (Gillespie et al. 2011a) and investigation of the spatial and temporal distribution of nutrients in the bay (MacKenzie 2004) indicate the importance of freshwater sources of inorganic nutrients for coastal primary productivity. However, based on these findings and the estimated flushing rate, there seems to be little potential for problems associated with over-enrichment (i.e. eutrophication) to occur.

Nutrient sinks in Tasman Bay

Denitrifying activity in the sediments of Tasman Bay results in the loss to atmosphere of some of the nitrogen present. Denitrification occurs as a result of the conversion of nutrient forms of nitrogen (e.g. nitrates [NO_3], ammonium [NH_4]) to nitrogen gas (N_2) via microbially mediated processes. A further unknown but possibly significant proportion of inorganic nitrogen is lost from Tasman Bay and Golden Bay via denitrification in the water column (Zeldis 2008; Gillespie et al. 2011a). Although there is a lot of uncertainty in the bay-wide extrapolation of these estimates, losses of nitrogen from the ecosystem may at times constrain productivity. Perhaps more importantly, however, these losses may mitigate any adverse enrichment effects from increased anthropogenic nutrient inputs to the marine environment.

2.1.4. Contaminant inputs

Overview

Land use practices involved with agriculture, forestry and residential and industrial activities introduce natural and artificial fertilisers, herbicides, pesticides and numerous other nutrients, toxic contaminants and sediments to the coastal environment. The routes of introduction include rivers and streams, land run-off from impermeable surfaces, and direct industrial and other point-source discharges. The principal point-source discharges along the eastern side of Tasman Bay are from the Bell Island WWTP and NWWTP and the Sealord Group Ltd discharge of fish-processing effluent opposite Port Nelson (referred to here as the 'Nelson fisheries outfall').

Inputs from rivers and streams

In addition to nitrogen loading, another aspect of freshwater quality that is particularly important to the ecology of the marine receiving environment is sediment load. Faecal contamination is a focus of NCC monitoring because, although it is not an indicator of ecological health, it is important for human use of the environment. Faecal contamination in the marine environment is usually from land-based sources and is delivered via rivers and streams. Measurement of faecal contamination is important for assessment of water quality and shellfish health (discussed in Section 5.3).

As presented on the Land and Water Aotearoa (LAWA) website, streams in the Tasman Bay catchments display a wide range of water quality values for *E. coli* and physico-chemical parameters (Table 1).

Table 1. Water quality data from freshwater monitoring sites in Tasman and Nelson. The site closest to the coast in each catchment was selected from the Land and Water Aotearoa (LAWA) website (www.lawa.org.nz). On LAWA, each site is compared to the range of sites across the whole of Aotearoa New Zealand and placed in one of four quartiles. Red = worst 25%, Orange = worst 50% (but better than the worst 25%), yellow = best 50% (but not in the best 25%), green = best 25%. Each stream was compared with all streams in Aotearoa New Zealand. *E. coli* is the faecal indicator bacterium most commonly reported from fresh water. Source: Newcombe et al. (2015).

Site	Type	*E. coli*	Turbidity	Nitrogen	Ammoniacal-N
Riwaka River	rural	best 25	best 25	–	–
Motueka River	rural	best 25	best 25	best 50	best 25
Tasman Valley Stream	rural	worst 25	worst 50	–	worst 25
Seaton Valley	rural	best 50	worst 25	–	worst 25
Waimea	rural	best 25	best 50	best 50	best 25
Reservoir Creek	urban	worst 50	worst 50	–	worst 25
Saxton Creek	rural	worst 25	worst 50	–	worst 25
Orphanage Stream	rural	best 50	best 50	–	best 50
Poorman Stream	urban	best 50	best 50	–	best 25
Jenkins Creek	urban	worst 50	worst 50	–	worst 50
Maitai	urban	best 50[5]	best 25	–	best 25
Todds Valley Stream	rural	worst 50	best 50	–	best 50
Wakapuaka	rural	best 50[6]	best 25	–	best 25
Whangamoa	rural	best 25	best 25	–	best 25

Sediment deposition from land often increases substantially due to human-induced change. Increased sedimentation has been identified as potentially the most important land-based stressor in marine environments (Morrison et al. 2009). Sediments are transported into Tasman Bay and Golden Bay in marine currents from the West Coast (Michael et al. 2012), in the rivers that flow directly into the bays and directly from coastal erosion.

Mature forest cover is most effective at protecting land from eroding. Hence, higher levels of erosion are likely from rainfall onto pastoral land or onto harvested commercial forest land within c. 6–8 years of harvest and replanting (Jones 2008). There is a large extent of commercial forestry in the catchment of Tasman Bay, but substantial areas of horticultural land border rivers in the lower parts of the catchments (Figure 1). LAWA data (Table 1) show that large rivers flowing into Tasman Bay have relatively low turbidity. However, this is not necessarily a good indication of suspended sediment load and most riverine inputs are associated with floods (Gillespie et al. 2011b; Michael

[5] At the Collingwood Street Bridge frequent breaches of recreational bathing limits were recorded.
[6] At Paremata Flats frequent breaches of recreational bathing limits were recorded.

et al. 2012). Accordingly, total input is unlikely to be measured accurately by infrequent periodic monitoring.

Coastal erosion and inundation risks increase during periods of extreme tides, strong onshore winds and storm surges. Although the Tasman Bay / Golden Bay region is a relatively low-energy environment, more than 70% of the coastline is subject to some degree of long-term persistent erosion. Significant areas of erosion occur along the Te Mamaku / Ruby Bay to Māpua shoreline, exceeding losses of 1 m per year (TDC 2013). Currently, 28% of the Tasman Bay coast (from Waimea Inlet to Mārahau) has shoreline armouring (e.g. seawalls, causeways, stopbanks and reclamations) (Robertson & Stevens 2012).

While sediment input from rivers and streams is apparent in Tasman Bay, particularly after rainfall events, average annual sediment input into the Tasman Bay / Golden Bay coastal waters is relatively low by national standards (Hicks in Morrison et al. 2009). Moreover, sediment input over the two decades up to 2012 was relatively low (as calculated with a sediment-yield estimator, reported in Michael et al. 2012).

Sediment resuspension in Tasman Bay

Although sediment loadings from rivers during storms can be substantial, this is not necessarily the immediate driver of suspended sediment levels observed in coastal waters. Observations of a fluctuating and sometimes persistent near-bottom high-turbidity layer in river plume-affected regions of Tasman Bay (Gillespie & Rhodes 2006) suggest that ongoing sediment resuspension can affect benthic habitat characteristics for extended periods. Studies of the timing of high winds, rainfall and turbidity changes have shown that increases in turbidity are associated with wind-generated waves rather than river flow (Cornelisen et al. 2011). Where storm events include both high winds and rainfall, marine turbidity increases before the river discharge increases. It follows that wave action stirring up the seabed, rather than river input, is the immediate driver of storm-associated turbidity increases. The fine sediments associated with a frequently disturbed seabed are more readily resuspended, exacerbating the presence and persistence of near-bottom high turbidity (Gillespie & Rhodes 2006).

Resuspension of historically deposited sediment is arguably a more important driver of sediment impacts in Tasman Bay than the input of new sediments because current sediment input to Tasman Bay is relatively low. Although the dynamics of sediment input and resuspension are relatively well understood, the spatial extent and exact nature of environmental impacts are not. Suspended sediment is thought to impact primary productivity, scallop survival and re-establishment of biogenic habitat (structure created by animals or plants) in Taman Bay. However, the scale and degree of impact are not easily determined with available information, and nor is the nature of interactions with other factors (e.g. direct disturbance, nutrient availability).

2.1.5. Contaminants from point-source discharges to eastern Tasman Bay

Nelson fisheries outfall

During consent-related monitoring, receiving water quality was measured around the Nelson fisheries outfall (Figure 1) at 3-monthly intervals between 1995 and 2000 (Brown 2001; Sneddon et al. 2004). Dissolved oxygen concentrations around the outfall and at stations up to 500 m to the southwest and the northeast (along a transect parallel to shore) were above the applicable consent limit of 6 mg/L on all occasions. The range of pH across all stations was 8.0–8.5 and water clarity (Secchi depth) ranged from 1 m to 4.8 m. The average concentration of total nitrogen (TN) was highest at the station closest to the outfall (100 m inshore: 0.35 g/m^3) but differences among stations were less than variations over time at the same station.

The annual input of TN from the Nelson fisheries outfall is around 80 t (Campos 2023). This is approximately 4% of that discharged into Tasman Bay by streams and rivers. The fisheries outfall discharge of TN is of the same order of magnitude as the two WWTP discharges.

Because of mercury's ability to accumulate in fish (in the highly toxic form monomethyl mercury), concentrations of the metal were measured in the discharge monthly in 1993 and in the receiving waters on seven occasions between 1995 and 2003 (Sneddon et al. 2004). Concentrations in the discharge were generally in the range 1–6 mg/m^3, with two outlying values of 13 mg/m^3 and 50 mg/m^3. Concentrations in the receiving waters ranged from 0.001 mg/m^3 to 0.74 mg/m^3, with a maximum value of 0.09 mg/m^3 at stations 500 m from the outfall. The ANZG (2018) guideline for slightly to moderately disturbed systems is 0.1 mg/m^3.

Bell Island WWTP outfall

Monitoring of the receiving environment of the Bell Island WWTP outfall is carried out every 5 years and samples of the wastewater are analysed for nutrients (species of nitrogen and phosphorus) and faecal indicator bacteria (FIB). Based on these samples (one composite per time of sampling), the nutrient concentrations in the wastewater appear to have decreased by factors of two or three between 2001 and 2016 (the most recent sampling event: Table 2).

Table 2. Nutrient (g/m^3) and faecal indicator bacterial concentrations (cfu/100 ml for 2001, MPN/100 ml for all other times) in wastewater samples collected during the receiving water surveys for the Bell Island WWTP discharge. DIN = dissolved inorganic nitrogen, TN = total nitrogen, DRP = dissolved reactive phosphorus, TP = total phosphorus, FC = faecal coliforms, EC = *E. coli*, ENT = enterococci, NR = not recorded. Source: Morrisey & Berthelsen (2017).

Year	NO_2-N	NO_3-N	NH_4-N	DIN	TN	DRP	TP	FC	EC	ENT
2001[a]	NR	NR	NR	23.1	30.1	10.0	10.0	600	NR	< 200
2005[b]	< 0.10	0.16	26.0	26.2	33.0	8.3	8.9	2,400	NR	33
2006[c]	< 0.10	< 0.10	22.0	22.0	27.0	7.6	8.1	NR	NR	NR
2011[d]	0.53	0.13	11.0	11.7	20.0	3.1	4.2	790	NR	20
2016[e]	< 0.10	< 0.10	12.5	12.6	17.2	2.3	2.8	13	5	< 10

[a] Gillespie & Asher 2001 [b] Gillespie & Asher 2005 [c] Gillespie et al. 2006
[d] Gillespie et al. 2011c [e] Morrisey et al. 2016

Monitoring around the Bell Island WWTP outfall from 2001 to 2016 has indicated that other sources (possibly the Waimea River) may influence nitrate concentrations periodically. However, with the decrease in concentrations evident in the wastewater in the most recent surveys (2011 and 2016), there has been a corresponding decrease in concentrations in the immediate receiving environment. Increases in concentration relative to background have been confined to the mixing zone (Morrisey & Berthelsen 2017).

Biannual surveys of phytoplankton community composition in inner Tasman Bay (off the western and eastern entrances to Waimea Inlet), as part of the Bell Island WWTP monitoring, have not shown any indication of effects from the discharge in terms of increased abundances of bloom-forming species (Morrisey & Berthelsen 2017).

2.2. Phytoplankton population dynamics in Tasman Bay

Phytoplankton are the most important primary producers within the Tasman Bay and Golden Bay ecosystem. Seasonal and inter-annual variations in biomass and specific composition of the phytoplankton affect the productivity of benthic and pelagic food webs. Phytoplankton productivity and biomass in nearshore (< 30 m depth) regions of Tasman Bay and Golden Bay are strongly affected by river inflows, which supply essential inorganic nutrients (Section 2.1.3). These inflows also affect estuarine circulation processes, density stratification and light availability, all of which have implications for phytoplankton growth. Chlorophyll-*a* (a proxy for phytoplankton biomass) in Tasman Bay measured between 1998 and 2003 ranged from 0.5 mg/m^3

to 2.9 mg/m^3 (Newcombe et al. 2015), rating the bay as oligotrophic (low) to mesotrophic (moderate) in terms of phytoplankton productivity.[7]

Gillespie et al. (in press) estimated primary productivity across Tasman Bay and Golden Bay. The increasing depth of water offshore (and therefore greater volume for phytoplankton growth) contributed to a general pattern of greater depth-integrated productivity away from the coast (Figure 2). However, benthic productivity is relatively higher in shallow waters (i.e. < 20 m). Beyond approximately 40 m depth, light and nutrients become progressively more limiting (although this is not discernible in Figure 2).

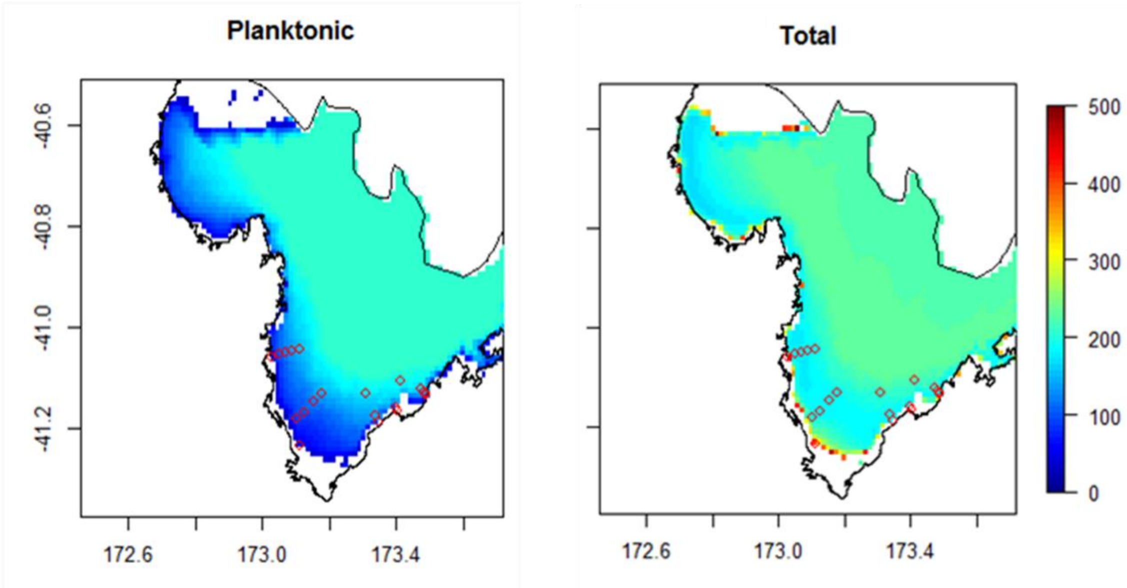


Figure 2. Spatial distribution of estimated planktonic and total primary production in Tasman Bay and Golden Bay (average using light intensities extracted for 2009–12 from the MODIS ocean-colour dataset). Values of primary production (mgC/m^2/day) are integrated over the water column and expressed as a rate per planar area. Estuarine areas are not included. Source: Gillespie et al. 2023 (forthcoming).

The modelled average productivity in Figure 2 shows a smoothed representation of productivity in the bays. In reality, phytoplankton biomass can vary widely with time, depth and location. Satellite imagery of chlorophyll-*a* in surface waters showed generally higher concentrations near the coast, but also high variability (Figure 3). As seen below, stratification dynamics can strongly influence the distribution of chlorophyll-*a* throughout the water column. The assessment of chlorophyll-*a* also becomes less reliable in turbid nearshore waters.

[7] The terms oligotrophic, mesotrophic and eutrophic correspond to systems receiving low, intermediate and high inputs of nutrients, respectively (Smith et al. 1999). These categories are based on international studies (Håkanson 1994), and ranges specific to New Zealand conditions have not been defined.

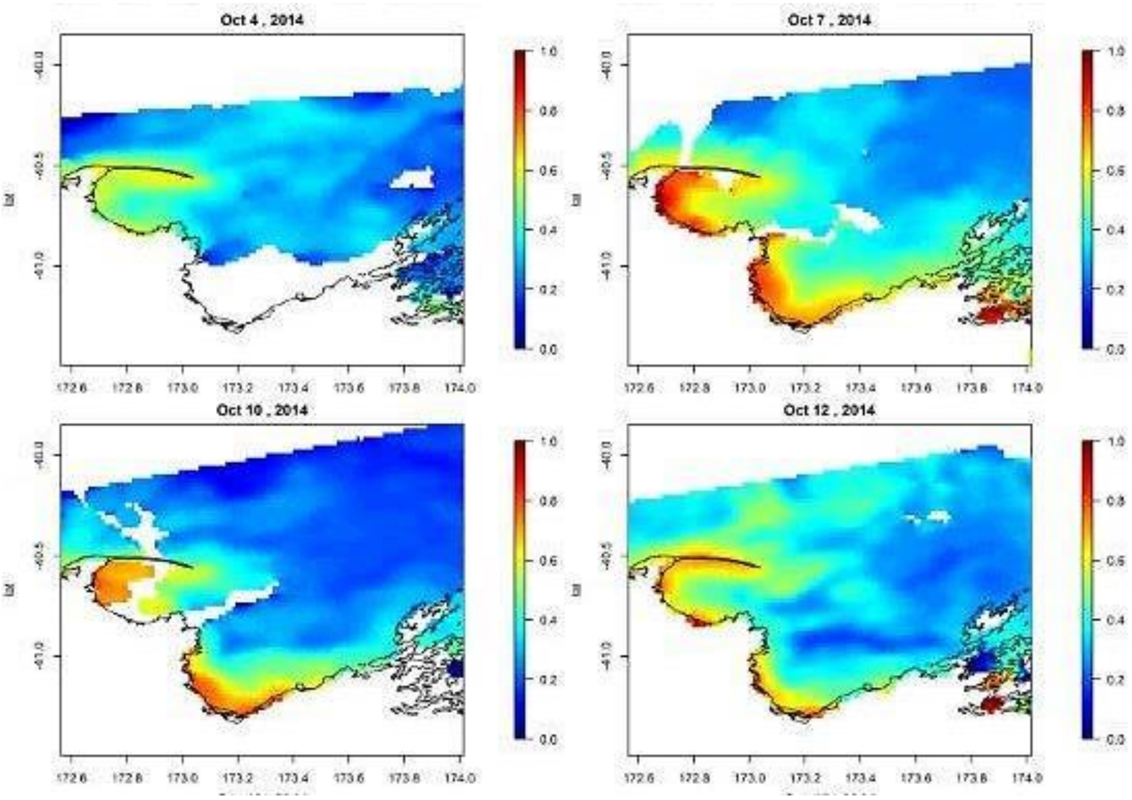


Figure 3. Chlorophyll-*a* concentrations in surface waters of Tasman Bay and Golden Bay visualised from ocean-colour data (MODIS Aqua level 2). Four days in October 2014. Source: Newcombe et al. (2015).

The variation observed in satellite images was also present in large multi-month surveys for Tasman Bay by MacKenzie & Adamson (2004). These surveys show that large gradients in phytoplankton (represented by chlorophyll-*a*) can exist throughout the bay and that these can change seasonally (Figure 4).

MacKenzie & Adamson (2004) also observed that temporal changes in the abundance and distribution of phytoplankton biomass in Tasman Bay are associated with changes in water column stratification from river and oceanic entrainment. In winter, a water column nitrate / nitrite concentration maximum develops due to advection of offshore waters into Tasman Bay, *in situ* remineralisation processes and light limitation of phytoplankton productivity at this time. Diatoms respond rapidly to water column mixing and high nitrate concentrations, and generally bloom in autumn and spring.

General seasonal patterns were observed, with the winter and spring period representing an annual productivity maximum. At these times, the conditions for shellfish nutrition are at their best. At most other times, flagellate-dominated phytoplankton communities within concentrated sub-surface layers are associated with a bay-wide mid-water column (10–15 m) pycnocline.[8] This is a common feature of the structure of the water column of Tasman Bay, coinciding with the depth range within which scallop and mussel growth and survival are highest.

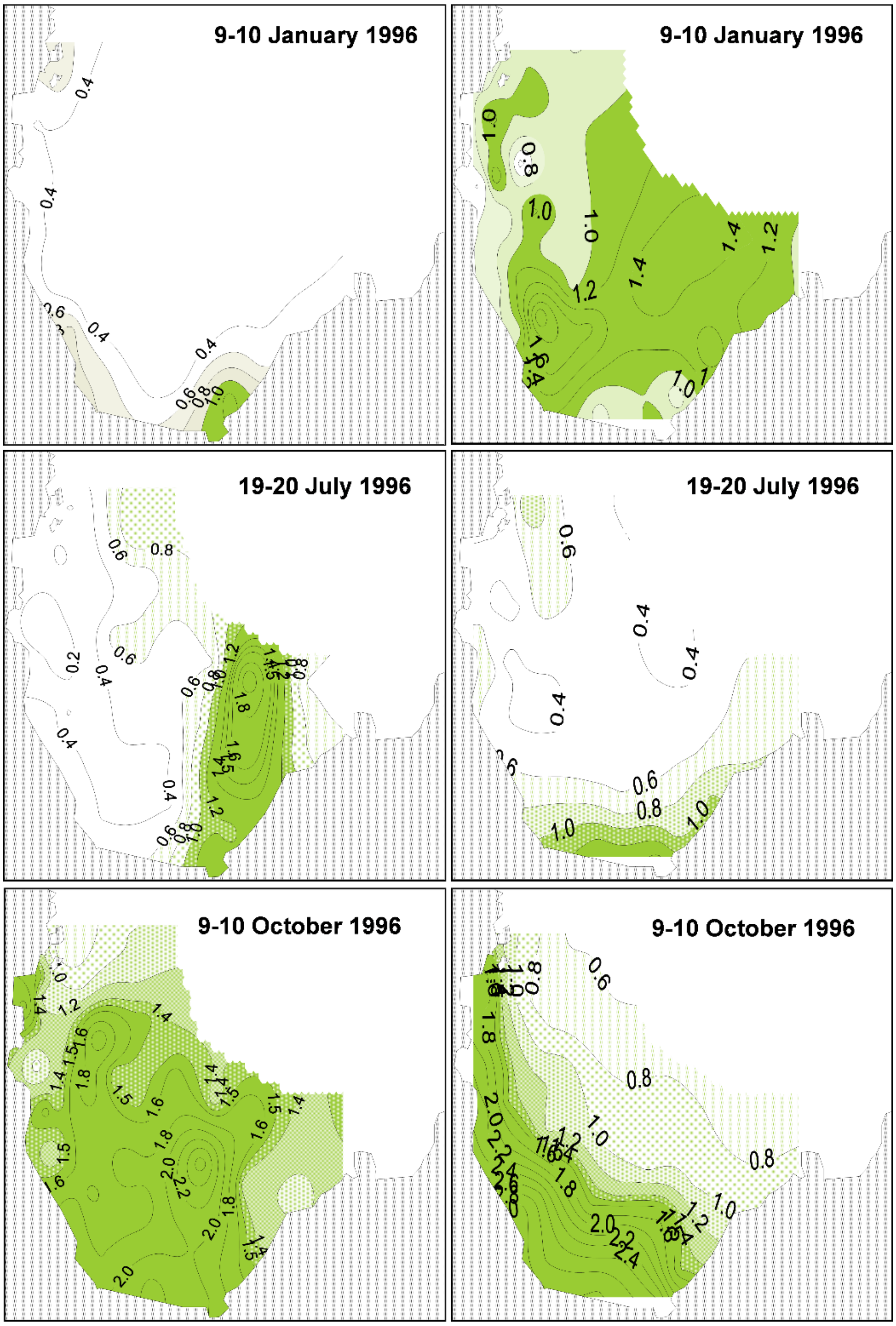


Figure 4. Examples of seasonal changes in phytoplankton biomass (mg/m^3 chlorophyll-*a*) concentrations in surface (left-hand column) and near-bottom (right-hand column) waters of Tasman Bay within the 30 m depth contour. Source: MacKenzie and Adamson (2004).

[8] A rapid change in density in a stratified water column.

The make-up of the phytoplankton community can influence the functioning of the ecosystem. The phytoplankton community structure and phenology in Tasman Bay is typical of a temperate coastal environment, although there is considerable year-to-year variation in biomass and taxonomic make-up, as well as the magnitude of photosynthetic productivity (MacKenzie & Adamson 2004). There has never been an attempt to achieve a complete taxonomic characterisation of the phytoplankton flora of Tasman Bay and Golden Bay (Newcombe et al. 2015).

Perturbations in the phytoplankton community (e.g. increases in primary productivity resulting from increased nutrient inputs) can lead to blooms of nuisance species. Although several toxic dinoflagellate species are known to occur in the region, no exceptional blooms of these species have been recorded. To date, the incidence of shellfish contamination with algal biotoxins has been low (MacKenzie 2004).

At about 20-year intervals, since at least the 1860s, there have been accounts of the accumulation of very large quantities of mucilage in the water column of Tasman Bay[9]. On a few occasions these events have been associated with harmful effects such as the mass mortalities of marine fauna and the impediment of fishing activities. The last major event that came to public attention was in 1981, although it is suspected that minor events are not uncommon.

In summary, there is a basic knowledge of the major species and their succession in Tasman Bay. There is a ‘typical’ pattern of winter diatom blooms followed by dinoflagellate dominance in summer.

2.3. Intertidal and shallow subtidal habitats, flora and fauna

The seabed over most of Tasman Bay consists of muddy sediment with varying amounts of sand and shell gravel (Figure 5). Organic content generally constitutes 4–7.5% of the weight of fine sediments (Grange 2007; Gillespie & Johnston 2012; Forrest 2014) but is lower in areas with coarser sediments (Sneddon & Clark 2011). The predominant organisms living in the sediments are polychaete worms, bivalve shellfish and crustaceans (e.g. McKnight 1969; Gillespie & Johnston 2012). The nature of the seabed around the NWWTP outfall is discussed in Section 3.2.

[9] It was earlier suggested that the colonial form of the haptophyte *Phaeocystis pouchetti* may be responsible for this phenomenon (Chang 1983). However, subsequent research has shown that the cause of these events is the planktonic dinoflagellate *Gonyaulax hyaline*, which produces polysaccharide mucilage (MacKenzie et al. 2002).

Figure 5. Map of benthic sediments in Tasman and Golden Bays. Source: Michael et al. (2012).

The shallow subtidal habitat of the Nelson Boulder Bank consists of an inshore fringe of boulders and cobbles sloping gently into deeper water. The boulder zone extends c. 200 m offshore, where patches of coarse, shelly sand are present among the boulders (Roan 1994; Brown 2001). Beyond this, patches of sediment become larger and finer-grained, until the seabed becomes sand and then mud. Sand is transported by water movement, and areas of boulders further from shore may be covered or uncovered over time, depending on patterns of water movement (Clark 2016). Monitoring of sediments among boulders around the Nelson fisheries outfall (in 7 m depth of water and 350 m offshore from the Boulder Bank, roughly opposite Port Nelson) has shown large spatial and temporal variability in sediment texture (Clark 2016).

As part of the consent monitoring for the fisheries outfall, Roan (1994) described three components of the biota of the Boulder Bank's subtidal fringe: organisms living within or on the sediment among the boulders, organisms living on the underside of boulders, and organisms living on top of boulders. The fauna of the sediments among the boulders was very limited but included large sabellid polychaetes and several species of bivalves (not named by Roan 1994).

Successive surveys of the Nelson fisheries outfall and surrounding area have reported very similar biota (Roan 1994; Brown 2001; Clark & Sneddon 2006; Sneddon 2009; Sneddon & Clark 2011; Clark 2016). The biota of the tops and sides of boulders included crustose coralline algae, sponges (*Tedania* sp., *Callyspongia* sp., *Chondropsis kirkii*), hydroids, barnacles (*Austrominius* (*Elminius*) *modestus*), tubeworms (*Galeolaria hystrix*, *Hydroides* sp.), sea squirts (including *Cnemidocarpa* sp.), the anemone *Actinothoe albocincta*, several species of sponge, and the bivalves *Modiolarca impacta* and *Anomia trigonopsis*. Apart from corallines, the only macroalga present was the encrusting brown *Ralfsia* sp. (Clark 2016).

Cushion stars (*Patiriella regularis*) were the most common mobile animals on shallow, hard substrata. Others included the eleven-armed starfish (*Coscinasterias muricata*), chitons (*Cryptoconchus porosus*), limpets (*Cellana stellifera*), snails (*Cominella virgata*, *Cookia sulcata*, *Lunella smaragdus* and *Coelotrochus tiaratus*), hermit crabs (*Pagurus* sp.), sea cucumbers (*Australostichopus mollis*) and occasional kina (*Evechinus chloroticus*). Green-lipped mussels (*Perna canaliculus*) are generally scarce on the Boulder Bank north of the lighthouse (Roan 1994; Sneddon et al. 2004).

The diffuser of the Nelson fisheries outfall was colonised by abundant *Actinothoe albocincta*, *Perna canaliculus* and bryozoans. Large brown seaweeds were represented by only a few scattered clumps of *Carpophyllum flexuosum*. The presence of these large numbers of suspension-feeding organisms on the diffuser is probably due to the supply of food in the form of suspended organic particles, and the relative stability of the substratum compared with the adjacent boulder habitat.

2.4. Fish and kaimoana

2.4.1. Shellfish

Historically, large, shallow beds of oysters (*Ostrea chilensis*) and green-lipped mussels occurred in the bays, but these have been removed through overexploitation, and commercial fishing for these species had virtually ceased by 2012 (Handley 2006; Handley & Brown 2012; Michael et al. 2012). Scallops (*Pecten novaezelandiae*) have been fished more recently, both commercially and recreationally, but have also declined and the fishery closed in July 2018 until further notice.[10] Decadal cycles have been identified in scallop abundance (based on abundance surveys and catch data), with highs occurring throughout the 1970s and in 1991–2002, the latter mainly in Golden Bay during a period of successful enhancement activity (Newcombe et al. 2015).

[10] https://www.mpi.govt.nz/travel-and-recreation/fishing/fishing-rules/challenger-region-fishery-management-area (accessed 28 May 2020).

2.4.2. Finfish and fisheries[11]

Tasman and Golden Bays are fished by trawling and seining for finfish, and have also been intensively dredged for scallops and oysters. Tuck et al. (2017) noted that together the bays comprise one of the most intensively fished and managed areas of photic-zone[12] seabed in Aotearoa New Zealand. The area is also used extensively for recreational fishing and boating.

The inshore commercial finfish fishery in Tasman Bay is primarily a trawl fishery. Commercial fishing is effectively excluded from Nelson Haven, the Waimea Estuary, and inside a line between the southern end of the Boulder Bank and the western tip of Moturoa / Rabbit Island, with prohibitions against trawl and surrounding nets, and set nets exceeding 1,000 m.[13] Other significant areas of exclusion in Tasman Bay include Tonga Island and Horoirangi Marine Reserves, the latter located approximately 4 km northeast of the NWWTP outfall.

Inshore commercial target species

The most important inshore species (by landed weight) trawled in Fisheries Statistical Area (FSA) 038 (comprising Tasman and Golden Bays, from Separation Point / Te Matau to Cape Stephens) are gurnard (*Chelidonichthys kumu*), snapper (*Pagrus auratus*), rig (*Mustelus lenticulatus*) and sand flounder (*Rhombosolea plebeia*) (Table 3). The 50 m depth contour is approximately 30 km offshore (northeast) from the NWWTP outfall. The 30 m contour is approximately 10 km offshore. Trawl survey catch rates reported by Stevenson and MacGibbon (2018) for depths between 20 m and 42 m in Tasman Bay indicated that the key species within this depth range include gurnard, snapper, barracouta (*Thyrsites atun*), sand flounder, John dory (*Zeus faber*), leatherjacket (*Meuschenia scaber*), jack mackerel (*Trachurus novaezelandiae*), tarakihi (*Nemadactylus macropterus*), rig and spiny dogfish (*Squalus acanthias*). In depths less than 30 m, flatfish, gurnard and snapper are likely to be the key commercial species, with rig an important bycatch.

The spatial distribution of catches, aggregated to either 'flatfish' (Figure 6) or 'finfish' (Figure 7), shows that the area offshore from the NWWTP outfall is relatively important for both categories and particularly for flatfish.

[11] Information, table and figures provided by Ross Sneddon (Cawthron). Note that although these data are five years old, they are still considered representative of post–Covid pandemic fishing effort given the low effort during the pandemic.
[12] The depth zone to which sufficient light penetrates to allow photosynthesis.
[13] Fisheries (Challenger Area Commercial Fishing) Regulations 1986: 6A (b) - CFR0136, 2B (1) (j) (2).

Table 3. Catch weight data for key species in the Tasman and Golden Bay Fisheries Statistical Area (FSA) 038 for the period October 2013 to October 2018. Data were obtained from the Ministry for Primary Industries Data Management Group under the Official Information Act 1982. Shaded cells designate those species included in the two aggregate classes for the fisheries data request (the data extract for the 'flatfish' aggregate also included catch information encoded to 'SOL' (sole), 'FLO' (flounder) and 'SDF' (sandfish), as well as the generic 'FLA').

Species code	Common name	Scientific name	FSA 038 catch wt[a]
Flatfish			
SFL	Sand flounder	*Rhombosolea plebeia*	99,427
FLA [b]	Flatfish	–	72,422
YBF	Yellow-belly flounder	*Rhombosolea leporina*	17,454
LSO	Lemon sole	*Pelotretis flavilatus*	10,326
GFL	Greenback flounder	*Rhombosolea tapirina*	4,747
ESO	New Zealand sole	*Peltorhamphus novaezeelandiae*	2,600
TUR	Turbot	*Colistium nudipinnis*	7.8
BRI	Brill	*Colistium guntheri*	0.8
	Total flatfish		**206,985**
Finfish			
GUR	Gurnard	*Chelidonichthys kumu*	222,674
SNA	Snapper	*Pagrus auratus*	119,852
SPO	Rig	*Mustelus lenticulatus*	108,372
BAR	Barracouta	*Thyrsites atun*	78,709
SCH	School shark	*Galeorhinus galeus*	56,582
JDO	John dory	*Zeus faber*	50,063
RCO	Red cod	*Pseudophycis bachus*	48,612
LEA	Leatherjacket	*Meuschenia scaber*	40,664
SPD	Spiny dogfish	*Squalus acanthias*	39,394
WAR	Common warehou	*Seriolella brama*	38,680
RSK	Rough skate	*Dipturus nasutus*	20,594
TAR	Tarakihi	*Nemadactylus macropterus*	7,674
STA	Giant stargazer	*Kathetostoma giganteum*	1,429
ELE	Elephant fish	*Callorhinchus milii*	332
	Total finfish		**792,967**

a. Average annual landed catch (kg) from all fishing methods for Fisheries Statistical Area 038 for the fishing years 2013/14 to 2017/18 (aggregated). From http://www.nabis.govt.nz

b. Catches filed under the non-specific 'FLA' code. It is likely that most were *Rhombosolea plebeia* (SFL).

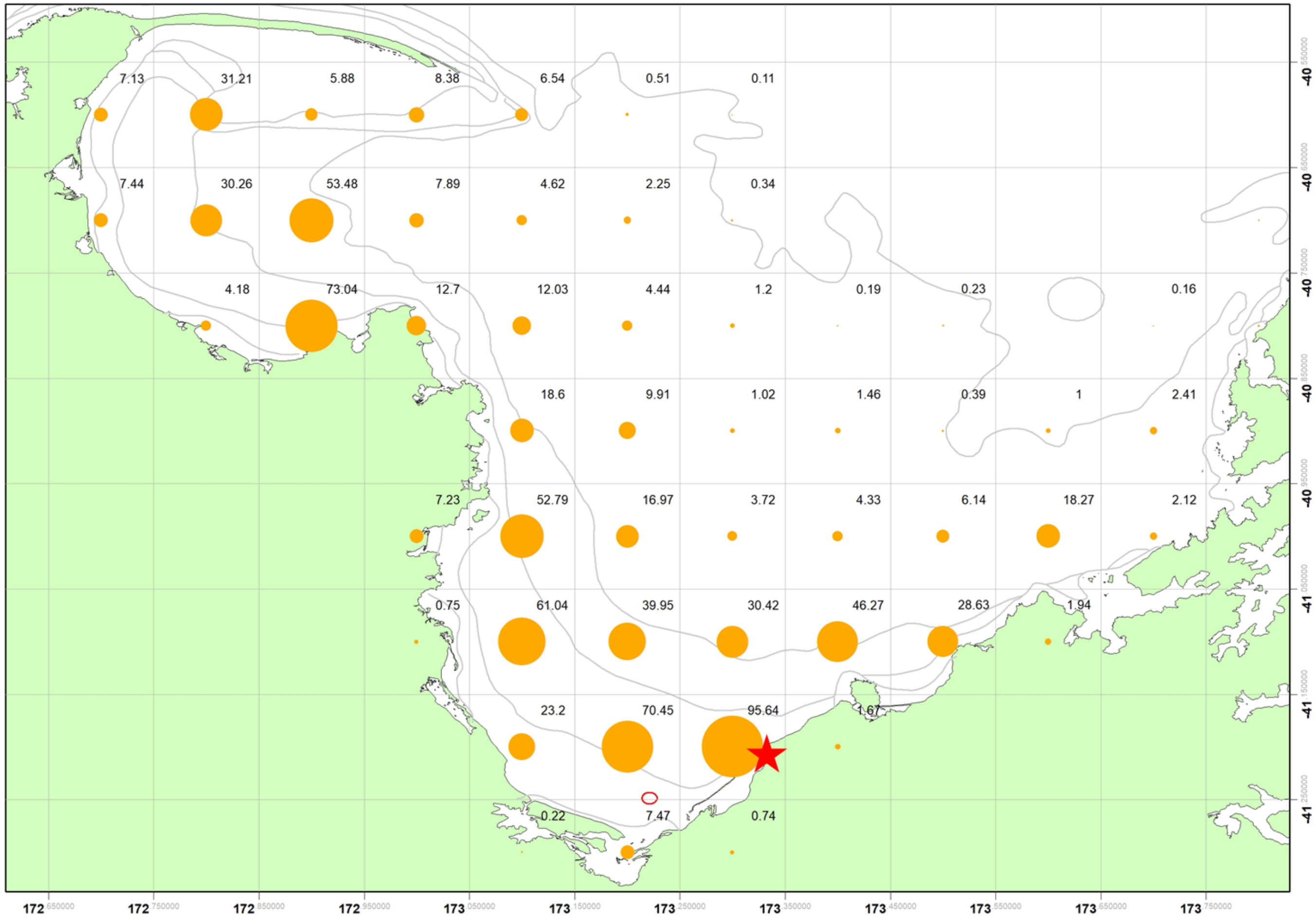


Figure 6. Recorded commercial catch weight (tonnes) for aggregated flatfish species in Tasman and Golden Bays (1 October 2013–1 October 2018) for 0.1-degree grid squares. The location of the Nelson North WWTP is indicated by the red star. Grey depth-contour lines are 5 m, 10 m, 20 m, 30 m and 50 m. Proportional symbols adjusted with Flannery compensation. Map prepared by Ross Sneddon (Cawthron Institute).

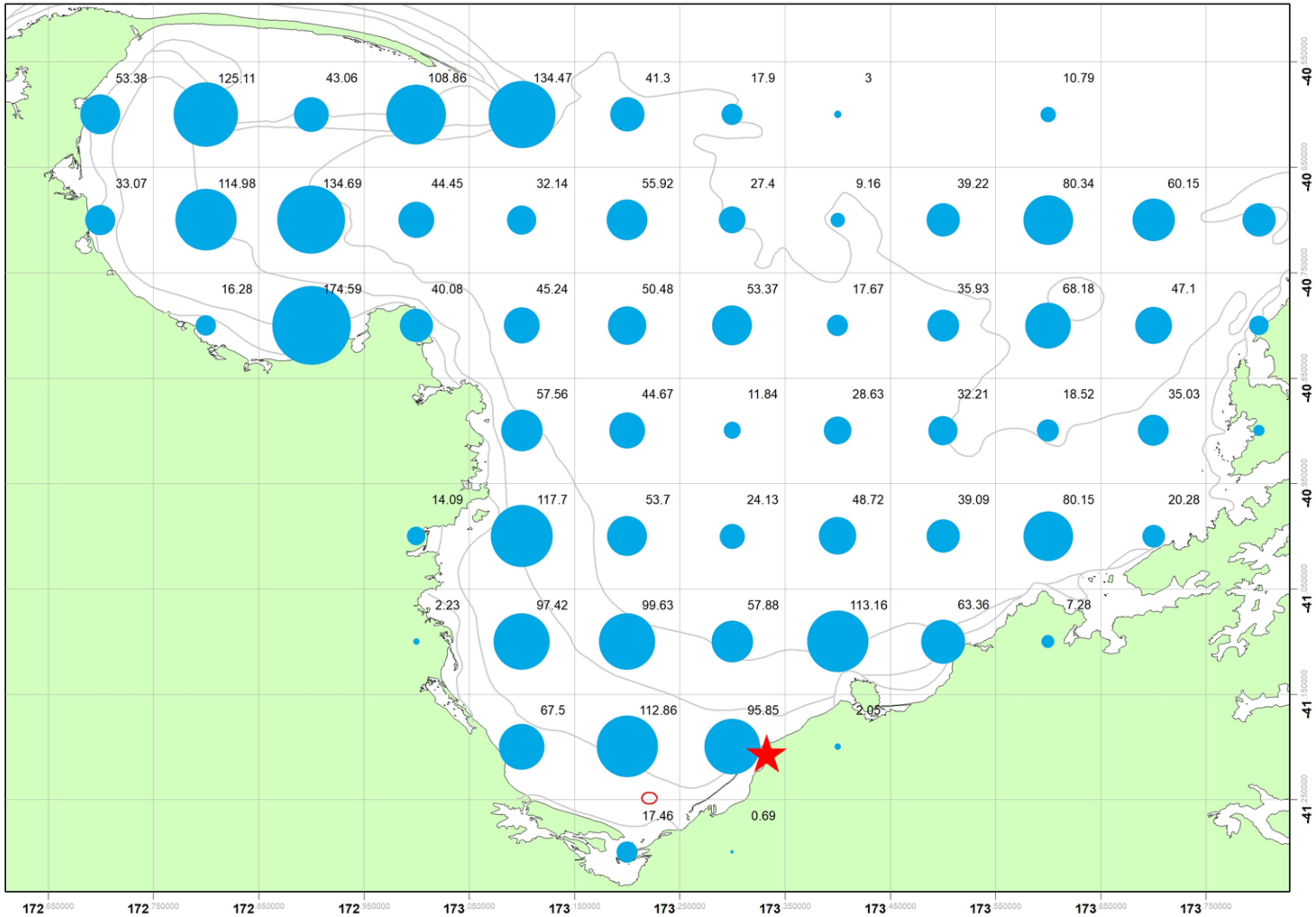


Figure 7. Recorded commercial catch weight (tonnes) for aggregated 'finfish' (excluding flatfish) species in Tasman and Golden Bays (1 October 2013–1 October 2018) for 0.1-degree grid squares. The location of the Nelson North WWTP is indicated by the red star. Grey depth-contour lines are 5 m, 10 m, 20 m, 30 m and 50 m. Proportional symbols adjusted with Flannery compensation. Map prepared by Ross Sneddon (Cawthron Institute).

Recreational fishing

Recreational fishing is widely practised in the inshore areas of Tasman Bay, with vessels launching from the boat ramp in Nelson marina to access areas in the southern and eastern parts of the bay. Target species include snapper, kahawai (*Arripis trutta*) and, in summer and autumn, kingfish (*Seriola lalandi lalandi*). Other species frequently landed include gurnard, rig and red cod (*Pseudophycis bachus*). Recreational dredging for scallops was also practised up until the scallop fishery was closed in 2018.

2.4.3. Kaimoana around the Nelson North WWTP

The coastal area around the NWWTP outfall does not appear to have any recognised value for food gathering (Barter & Forrest 1998). Roan (1994) and Barter and Forrest (1998) noted that blue and green-lipped mussels are rare along the shore in this part of the Boulder Bank. The nearest shellfish beds of recognised value are offshore oyster and scallop beds 7–8 km north of the outfall.

Roan (1994) and Brown (2001) reported that triplefins (*Forsterygion varium*) and spotties (*Notolabrus celidotus*) were common in shallow water along the Boulder Bank. Several other species were apparently attracted to the Nelson fisheries outfall, including carpet sharks (*Cephaloscyllium isabella*), kingfish, tarakihi, porae (*Nemadactylus douglasii*) and trevally (*Pseudocaranx dentex*).

3. PREVIOUS ASSESSMENTS AND CONSENT MONITORING FOR EFFECTS OF THE NELSON NORTH WWTP OUTFALL

3.1. Water quality

3.1.1. Trace metals and other chemical substances

Barter and Forrest (1998) reported the results of monitoring of wastewater quality from the NWWTP from November 1996 to September 1998. For most trace metals and non-metallic toxicants analysed, concentrations were very low and often below the limits of detection. Most were below their corresponding marine trigger values for the protection of 95% of aquatic species (ANZG 2018), even prior to dilution within receiving waters (the trigger values apply to concentrations in receiving waters). Copper exceeded its guideline values but only by a factor of eight or less, this being well below the dilution values derived by Barter and Forrest (1998) from dye studies and plume modelling (as well as by the more recent hydrodynamic modelling by MetOcean Solutions Ltd 2022). Consequently, adverse effects from these contaminants were considered unlikely.

More recent discharge monitoring undertaken during the period August 2020–December 2021 to support the consent application showed that the concentrations of trace metals and other chemical substances in the discharge were at or just above the limits of detection of the testing methods (Table 4). Where contaminants were detectable, concentrations were much lower than corresponding discharge consent limits and mostly below the ANZG (2018) trigger values for protection of 95% of marine species with little or no dilution. The exceptions were copper and zinc, the maximum recorded values of which exceeded their DGVs by factors of 4 and 2.5, respectively (note that the concentrations were measured in the effluent before any mixing with ambient waters, whereas DGV criteria apply to the receiving environment after mixing).

During the same 2020–21 monitoring period, concentrations of volatile and semi-volatile organic compounds and oil and grease in both the influent and effluent were at or just above the limits of detection of the testing methods. The one exception was a single relatively high result for oil and grease detected at the wetland outlet (159 g/m^3).

Table 4. Concentrations of trace metals and other toxicants in the discharge for the period August 2020 to December 2021, compared to effluent consent limits and the ANZG (2018) default guideline values (DGV) for protection of marine species in slightly to moderately disturbed systems.

Trace metal / substance	Minimum–Maximum (g/m^3)	Consent limit (g/m^3)	Below consent limit (Yes/No)	DGV [a] (g/m^3)
Cadmium	< 0.000053–0.00011	0.275	Yes	0.0007
Copper	0.0023–0.005	0.065	Yes	0.0013
Nickel	0.0044–0.0076	3.5	Yes	0.007
Zinc	0.0058–0.020	0.75	Yes	0.008
Chromium VI	0.00149–0.00380	1.37	Yes	0.0044
Lead	0.00036–0.00119	0.22	Yes	0.0044
Cyanide	Not detected	0.2	Yes	0.004
Phenol	< 0.02	20	Yes	0.4
Mercury	< 0.00008	0.02	Yes	0.0001

a. Level of protection: 95% of marine species in all cases apart from cadmium, nickel and mercury, which are listed at the (ANZG recommended) 99% level.

3.1.2. *Microbiological contaminants*

Microbiological water quality was monitored in the mixing zone of the outfall at 3-monthly intervals from November 2006 to March 2008, and then at 4-monthly intervals until April 2010. The duration of monitoring spanned the period from 2 years before to 2 years after the 2008 upgrade to the NWWTP (Bailey & Conwell 2010). Samples were taken within the discharge surface 'boil' and at 250 m, 500 m and 1,000 m northeast or southwest of the outfall, depending on the direction of tidal flow at the time. Samples were analysed for enterococci and faecal coliform bacteria.

Prior to commissioning of the upgrade in March 2008, concentrations of faecal coliforms and enterococci were elevated (up to 1,200 cfu/100 ml[14] for coliforms and 70 cfu/100 ml for enterococci) immediately around the boil on some sampling occasions and sometimes extended to the 250 m mixing-zone boundary. Bacterial concentrations were generally low outside the mixing zone (maxima of 70 cfu/100 ml and 5 cfu/100 ml for coliforms and enterococci, respectively). Concentrations were generally below analytical detection limits (5 cfu/100 ml) on other sampling occasions prior to the upgrade in 2008. Concentrations of coliforms were lower after the upgrade (maximum of 30 cfu/100 ml). Concentrations of enterococci were similar before and

[14] Colony forming units per 100 ml.

after. Concentrations of both variables were at or below limits of detection beyond the mixing zone.

The current coastal permit prescribes faecal coliform limits of 10,000 cfu/100 ml (calculated as a monthly median count over 12 months) and 80,000 cfu/100 ml (no more than one of 12 monthly samples to exceed this concentration). During the period of additional consent monitoring (August 2020–December 2021), concentrations of faecal coliforms in the effluent discharge ranged from 183 cfu/100 ml to 7,000,000 cfu/100 ml. Except for three samples yielding greater than 10,000 cfu/100 ml (two of which exceeded the upper consent limit; Figure 8), the monitoring results were at or below the lower consent limit. There were no significant increasing or decreasing trends in coliform concentrations during the monitoring period (Mann-Kendall test; 99% confidence).

The geometric mean concentrations in influent and final effluent samples were 3,970,000 cfu/100 ml and 2,140 cfu/100 ml, respectively. This corresponds to a 3.3$\log_{10}$ reduction on average, which is typical of those observed in other well-performing treatment plants in Aotearoa New Zealand.

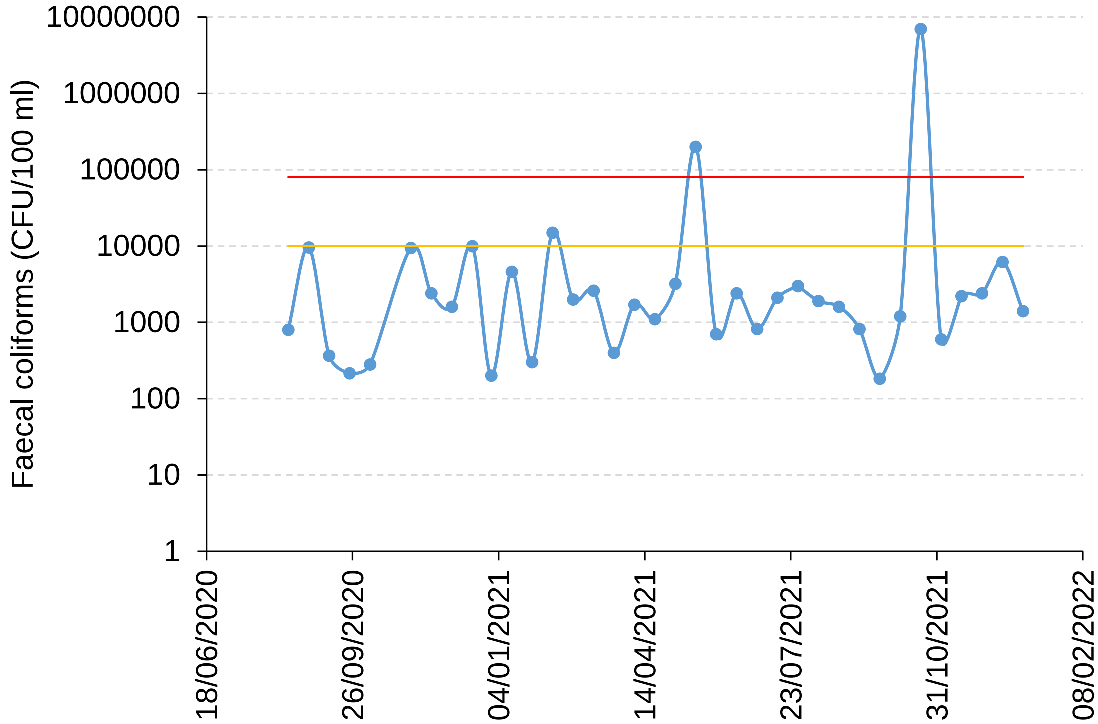


Figure 8. Time series of faecal coliforms in the final effluent discharge from Nelson WWTP, August 2020–December 2021. The orange reference line indicates lower standard for the monthly median monthly. The red reference line indicates the upper standard.

The geometric mean concentrations of enterococci in the discharge receiving environment during the same period were 13 cfu/100 ml at the site adjacent to the WWTP outfall (CM AW 01) and 21 cfu/100 ml at Schnappers Point (CM AW 02) (see Figure 10). This corresponds to respective reductions of 1.6 $\log_{10}$ and 1.4 $\log_{10}$ (in the marine environment due to dilution and decay) relative to the wetland outlet site (CR WW 03; 501 cfu/100 ml).

3.2. Seabed

Barter and Forrest (1998) mapped seabed habitats around the outfall using aerial photographs and diver surveys. They reported that boulder habitat extended tens to hundreds of metres seaward of the Boulder Bank. South and east (inshore) of the outfall, boulder habitat was relatively patchy, with continuous boulder field inshore and individual or patches of boulders in predominantly sandy areas further offshore (Figure 9). The seabed around the outfall was rippled medium sand, becoming predominantly small boulders beyond 60 m north of the outfall.

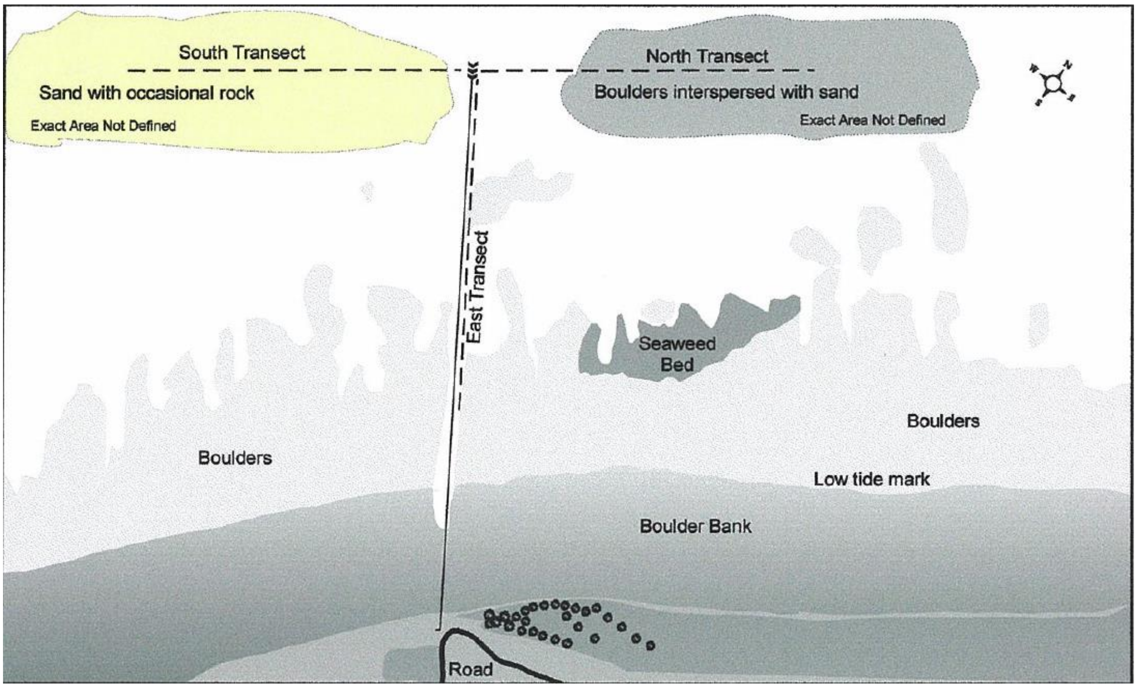


Figure 9. Map of seabed habitats around the Nelson North WWTP outfall. Source: Barter and Forrest (1998).

Sandy habitat

The sand around the outfall showed no sign of organic enrichment (such as black discoloration and smell of hydrogen sulphide), appearing brown and well oxygenated.

Concentrations of organic matter and nutrients (TN, ammonia, nitrate, nitrite and total phosphorus) were low at all stations (stations were located 50–1,000 m from the outfall in both directions parallel to shore). Concentrations (0.5–1.0% for organic matter and 50–70 mg/kg for TN) were within the range found at other sites along the Boulder Bank and elsewhere in Tasman Bay. Patches of microalgae were present on the surface of the sand but there was no indication that their growth was more prolific near the outfall than elsewhere (as might be expected if nutrients from the outfall were having a stimulating effect).

Prior to the present investigation, there had been no assessment of trace-metal contamination of sediments around the outfall. To address this information gap, sediment samples were collected on 26 November 2020 from four stations: 25 m and 500 m north of the outfall, and 25 m and 500 m south (locations are shown in Table 5). All sampling stations were in the same depth of water as the outfall.

There were no consistent differences between concentrations at locations closer to the outfall (25 m) and those 500 m away for any of the trace metals or organic matter (Table 5). Thus, there is no indication that the discharge has caused any enrichment of these contaminants in the sediments. This is not surprising given the low concentrations in the wastewater (see Section 5.1.3), and the sandy nature and low organic content of the sediments (metals tend to bind preferentially to fine, organic particles). All metals were present at concentrations well below guideline values at which adverse ecological effects may occur (ANZG (2018) DGV: Table 5).

Table 5. Grain-size distribution (by wet sieve) and concentrations of trace metals in sediments from sites around the Nelson North WWTP outfall, collected in November 2020. Coordinates are WGS84. All percentages are by dry weight. Metals concentrations as mg/kg dry weight. Also listed are ANZG (2018) default sediment-quality guideline values (DGV), at which adverse ecological effects may occur.

	Site				
	500 m north	25 m north	25 m south	500 m south	ANZG DGV
Latitude	41°11.851'	41°12.004'	41°12.026'	41°12.218'	
Longitude	173°19.500'	173°19.234'	173°19.212'	173°18.987'	
% > 2 mm	0.9	0.7	0.1	0.5	
% 1–2 mm	4.4	1.6	0.8	1	
% 0.5–1 mm	64.8	35.4	31.6	20.7	
% 250 µm–0.5 mm	29	61	59.9	72.5	
% 125–250 µm	< 0.1	0.3	0.5	3.8	
% 63–125 µm	< 0.1	< 0.1	< 0.1	< 0.1	
% < 63 µm	1	1.1	7	1.4	
% organic matter	0.94	0.86	0.84	0.86	
Arsenic	7.5	5.3	5.2	4.3	20
Cadmium	0.026	0.024	0.029	0.026	1.5
Chromium	5.9	4.5	5.0	4.9	80
Copper	13.8	13.5	14.0	13.7	65
Lead	5.7	5.6	5.9	5.9	50
Mercury	< 0.02	< 0.02	< 0.02	< 0.02	0.15
Nickel	3.1	2.5	2.6	2.6	21
Zinc	22.0	22.0	21.0	19.7	200

Barter and Forrest (1998) reported no indication of adverse ecological effects of the discharge on sediment-dwelling organisms around the outfall. The surface of the sand was largely barren at all sampling stations. The most conspicuous macrofauna were cushion stars (*Patiriella regularis*), and eleven-armed starfish (*Coscinasterias muricata*), hermit crabs (*Pagurus* spp.), sea cucumbers (*Australostichopus mollis*) and whelks (*Cominella virgata* and *Cominella adspersa*) were also widespread.

Animals living in the sediment (sampled by coring) were dominated numerically at all sampling stations by syllid polychaetes and amphipod crustaceans (Barter & Forrest 1998). The sediment fauna was similar between sites near the NWWTP outfall and the station furthest from it (1,000 m to the south), and there was no dominance by taxa known to be characteristic of organically enriched sediments.

The New Zealand lancelet (*Epigonichthys hectori*), a fish-like member of a group of primitive chordates (and, therefore, of scientific interest), was unexpectedly abundant in sandy sediments around the outfall. This species is endemic and occurs around

Aotearoa New Zealand, but the top of the South Island is towards the southern limit of its distribution (Paulin 1977). They are apparently uncommon in shallow (i.e. diveable) waters but have been reported from other surveys in the Nelson–Marlborough area and, according to Paulin (1977), are 'known from several bays within Tasman Bay and Croiselles Harbour'. Lancelets require clean, well-oxygenated sand with low mud content (Paulin 1977). Their presence at sites around the outfall supports the conclusion that the discharge has not adversely affected the surrounding sediment or the organisms living within it. Subsequent surveys have not sampled the sediment around the outfall, so it is not known whether the lancelet populations are still present.

Hard-substratum habitats

Boulder habitat around the outfall was sampled by Barter and Forrest (1998), who found that it supports a community of encrusting and mobile taxa comparable to other parts of the Boulder Bank. Extensive macroalgal beds are notably absent from the length of the Boulder Bank, including the NWWTP outfall location. There was no sign of excessive sedimentation on rocky habitats near the outfall, nor any patterns in the distribution of organisms that might have suggested an effect from the discharge.

Prominent taxa reported by Barter and Forrest (1998) included cushion stars, eleven-armed starfish, kina, the snail *Cookia sulcata*, the ascidian *Cnemidocarpa* sp., nesting mussels (*Modiolarca impacta*) and window oysters (*Anomia zelandica*). The encrusting biota included coralline algae and various sponges (e.g. *Callyspongia ramosa*, *Raspailia topsenti*, *Ecionemia* (*Ancorina*) *alata*, *Aaptos aaptos* and *Tethya* spp.), and compound ascidians (including *Botryllus schlosseri* and *Didemnum candidum*). Few fish species were seen, the most common being variable triplefins (*Forsterygion varium*), spotties (*Notolabrus celidotus*) and blue cod (*Parapercis colias*).

In addition to those recorded on nearby boulder habitats, the organisms living on the outfall structure included high densities of the anemone *Actinothoe albocincta* (Barter & Forrest 1998). Densities of *A. albocincta* decreased to moderate levels on the boulders immediately surrounding the outfall and declined further with increasing distance. This matches the distribution of this species on the Nelson fisheries outfall (Section 2.3), presumably due to the supply of food in the form of particulate organic material.

Under the conditions of the present discharge permit, a qualitative ecological survey of the seabed around the NWWTP outfall diffuser is carried out every 5 years. The pre-commissioning baseline survey was undertaken in 2006 (Bailey & Conwell 2010), and post-commissioning surveys have been documented by Barter (2013), Sneddon (2018) and Morrisey (2021). The most recent was brought forward by two years to inform the current consent application.

The surveys provide a description of the ecological communities colonising the surface of the diffuser and the seabed in its immediate vicinity. Scuba divers record a video transect along both sides of the diffuser for approximately 20–30 m from the distal end.

In addition, 0.10 m^2 photo-quadrat images are taken at eight pre-established locations on the diffuser surface. The water depth at the point of discharge is c. 11 m at low tide.

The fauna on the diffuser has included both mobile and encrusting taxa and was similar to that described for the Nelson fisheries outfall (Section 2.3). Among the mobile taxa were several species of starfish: the cushion star, the reef starfish *Stichaster australis* and the eleven-armed starfish (in descending order of abundance: Sneddon 2018). Sea cucumbers (*Australostichopus mollis*) and kina (*Evechinus chloroticus*) also occurred at the base of the diffuser where it met the sediment of the seabed.

The encrusting biota was dominated in all surveys by the anemone *Actinothoe albocincta* and crustose coralline algae. Other taxa included finger sponges (e.g. *Callyspongia ramosa*), solitary ascidians (*Cnemidocarpa* sp.), compound ascidians (e.g. *Aplidium* sp., *Botryllus schlosseri* and *Didemnum candidum*) and green-lipped mussels.

Results of the 2021 survey were generally consistent with those of the previous surveys in terms of the composition of the fauna on the diffuser (Barter 2013; Sneddon 2018; Morrisey 2021).

3.3. Kaimoana

As noted in Section 2.4.1, blue and green-lipped mussels are rare along the shore in this part of the Boulder Bank and the area around the NWWTP outfall does not appear to have any recognised value for food gathering (Barter & Forrest 1998). The nearest shellfish beds of recognised value are offshore oyster and scallop beds 7–8 km north of the outfall (the latter fishery is currently closed). Green-lipped mussels are present in low densities on the outfall structure (Sneddon 2018). Barter and Forrest (1998) assessed the human health risk from consuming shellfish contaminated with microbial pathogens against guidelines for concentrations of faecal indicator bacteria in shellfish-gathering areas. They concluded that, during peak flows, concentrations could exceed guidelines for up to 1,000 m downstream of the NWWTP diffuser. However, because the plume tended to remain offshore, and because there are few harvestable shellfish along the Boulder Bank, Barter and Forrest considered the associated health risk to be minimal.

Commercially or recreationally targeted species of fish recorded during monitoring surveys around the outfall diffuser include blue cod and snapper (*Pagrus auratus*) (Bailey & Conwell 2010; Sneddon 2018). However, there is no evidence that the area is targeted for fishing, probably because of the presence of the outfall. As far as we are aware, the lancelets reported by Barter and Forrest (1998) (see Section 3.2) are not targeted as kaimoana.

3.4. Conclusions from monitoring to date

At the time of Barter and Forrest's (1998) survey, the NWWTP outfall had been operating for c. 30 years, and during the first 10 years of its operation had been discharging untreated sewage into Tasman Bay. Despite this history (and the nearest sampling stations being just 25 m from the outfall), there was no clear evidence of any adverse effect on the surrounding benthic sediments or communities.

In a review of the results from the 2006, 2013 and 2018 surveys, Sneddon (2018) concluded that there was no visual indication of the accumulation of discharge constituents or discharge-related effects on hard substrata within the vicinity of the diffuser. Benthic faunal communities associated with both the diffuser structure and the surrounding boulders showed no obvious patterns in the distribution of species to suggest an adverse effect from the discharged wastewater. Re-examination of fixed quadrats on the diffuser structure showed that many sessile organisms did not persist from one survey to the next. This turnover of encrusting biota was attributed partly to their limited lifespan, but also to episodic disturbance by storms, followed by resettlement.

4. POTENTIAL FUTURE CHANGES TO THE WWTP DISCHARGE

Although flows can be buffered within the WWTP and otherwise managed, it is likely that future discharge rates will be similar, overall, to the flows into the plant (shown in Table 6). Concentrations and loads of contaminants discharged to Tasman Bay, however, are highly dependent on the effectiveness of treatment within the WWTP and will not necessarily correlate closely with those in the influent. Predictions of future effluent quality were not available to us at the time of writing.

Table 6. Current and predicted (2059) influent wastewater flow, loads and composition. Data provided by Stantec.

Variable	Unit	2022	2059
Average dry weather flow	m^3/day	7,400	10,100
Average daily flow	m^3/day	8,200	11,100
Peak wet weather flow	m^3/day	36,800	49,800
Chemical oxygen demand (COD) load	kg/day	3,300	4,500
Biochemical oxygen demand (BOD)	kg/day	1,800	2,400
Total suspended solids (TSS)	kg/day	1,700	2,200

5. ASSESSMENT OF CONTINUING AND FUTURE ECOLOGICAL EFFECTS

5.1. Water quality

5.1.1. Nutrients and phytoplankton

The most important limiting nutrient (i.e. that restricts plant growth) in Tasman Bay is nitrogen, in the form of dissolved inorganic nitrogen (DIN). Although inorganic phosphorus, iron and silica can also be limiting occasionally (MacKenzie 2004), these nutrients are not thought to significantly constrain phytoplankton production in the bay (although they may influence the types of phytoplankton present).

There are no national guideline values for nutrient concentrations in Aotearoa New Zealand marine waters. ANZECC (2000) recommended that, in the absence of locally derived criteria, the guideline values for southeast Australia be used as a set of default criteria for Aotearoa New Zealand. But these have consistently proved too conservative, with even relatively pristine waters from several Aotearoa New Zealand regions regularly exceeding the southeast Australian values. While several regional authorities have made efforts to develop a set of area-specific guideline values for coastal waters, there has been no formal process undertaken for Tasman Bay. Like ANZECC (2000), the ANZG (2018) guidelines that supersede them place an emphasis on developing such area-specific values using a combination of reference-site and laboratory- or field-effects data.

Surface water quality (including nutrients) was sampled monthly over the period August 2020 to December 2021 at two locations just offshore from the Boulder Bank: the first opposite the western end of the treatment pond (‘Adjacent to NWWTP’, c. 20 m offshore) and the other opposite the eastern end (‘Schnappers Pt’, c. 40 m offshore) (Figure 10). The median concentration of nitrate / nitrite over this period was 0.014 g/m^3 at both sites (Table 7). This value is similar to those for nitrate measured in Tasman Bay in 1995–96 (0.003–0.040 g/m^3: Mackenzie et al. 2003). The highest values occurred from June to August, consistent with the pattern of highest concentrations in late winter to spring reported in the 1995–96 study (Figure 11). Concentrations of dissolved reactive phosphorus were also highest during winter and early spring (Figure 11).

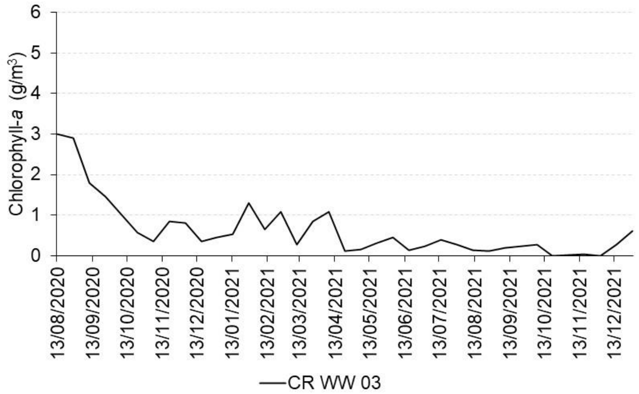
Figure 10. Location of offshore water quality monitoring stations in the Nelson North WWTP and Tasman Bay sampled monthly from August 2020 to December 2021. Offshore stations are shown by green pins; other stations are wastewater (yellow) and groundwater (red) sampling locations.

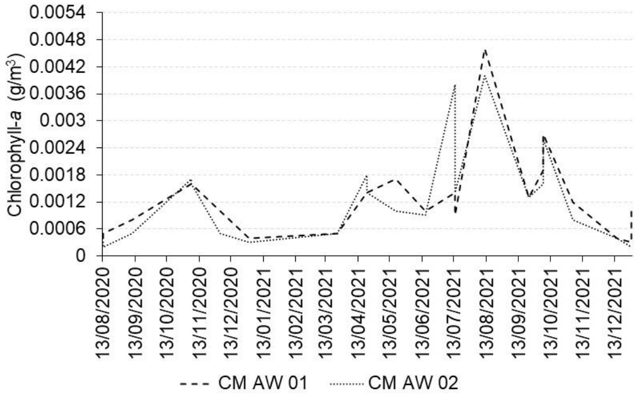
Table 7. Concentrations of nutrients in the discharge and in seawater samples collected from August 2020 to December 2021 at two coastal sites near the Nelson North WWTP. All concentrations are in g/m^3. Chl-*a* = chlorophyll-*a*, NO_x = nitrate / nitrite, DRP = dissolved reactive phosphorus, TP = total phosphorus, TAN = total ammoniacal nitrogen; *n* = number of sampling dates. Concentrations for total Kjeldahl nitrogen and total nitrogen were all or mostly below the limit of detection.

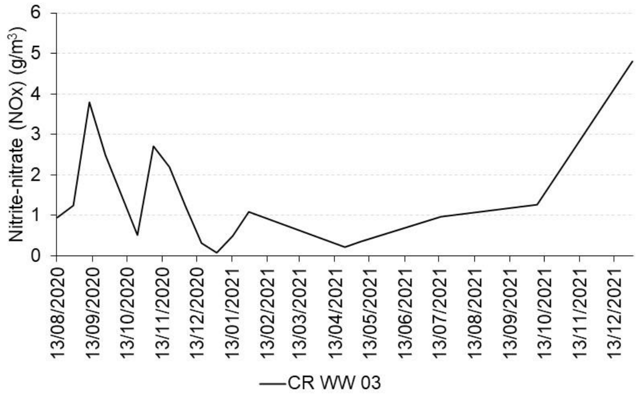
Sampling site		**Chl-*a***	**NO_x**	**DRP**	**TP**	**TAN**
NWWTP discharge	Median	0.355	1.1	3.2	4.3	12.2
	95th percentile	2.075	4.0	6.1	6.6	27.8
Adjacent to NWWTP	Median	0.0011	0.0138	0.0135	0.0200	–
	95th percentile	0.0027	0.0516	0.0343	0.0380	–
Schnappers Point	Median	0.0010	0.0144	0.0125	0.0160	–
	95th percentile	0.0037	0.0220	0.0410	0.0490	–
	n	22	22	22	21	36

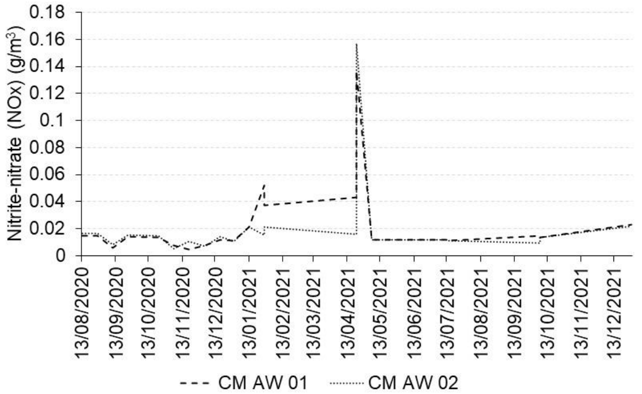
NWWTP discharge

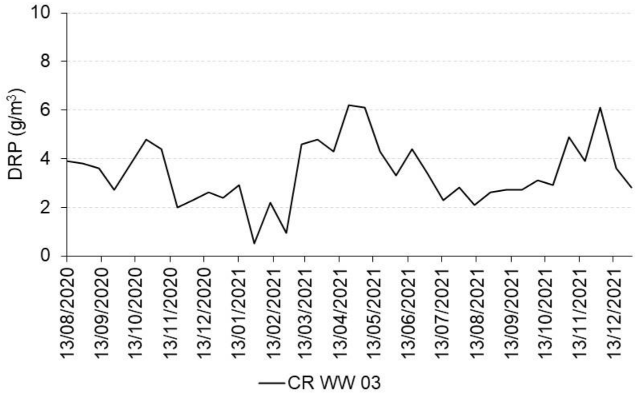
Tasman Bay sites

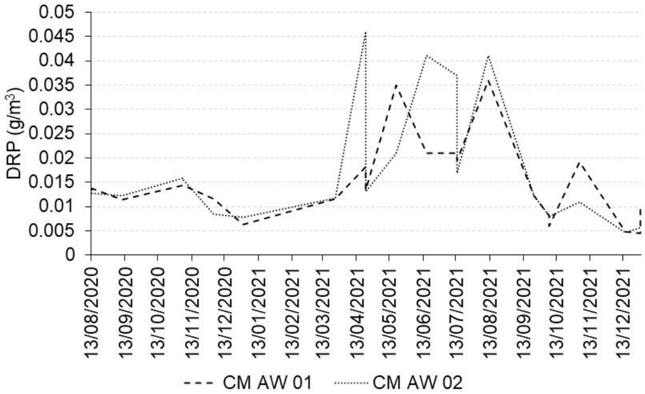


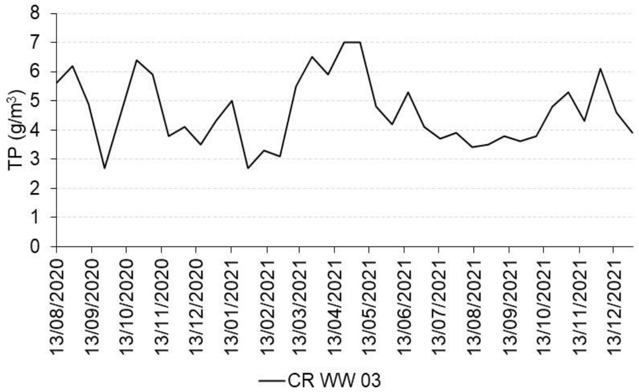


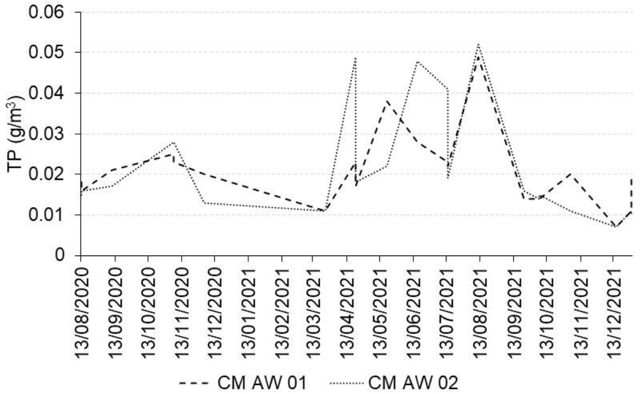












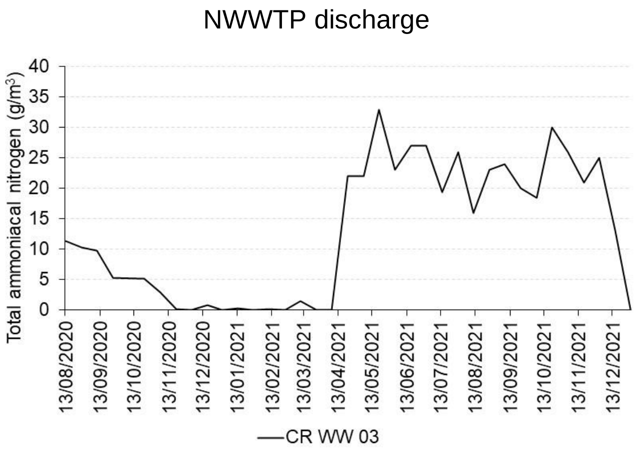


Figure 11. Concentrations of chlorophyll-*a* and nutrients in the NNWWTP discharge (sampled fortnightly) and at two offshore water quality monitoring stations in Tasman Bay (sampled monthly) during the period August 2020 to December 2021. DRP = dissolved reactive phosphorus; TP = total phosphorus. The dotted line is the station 'Adjacent to NWWTP' and the solid line is the 'Schnappers Pt' station (see Figure 10).

The additional nutrient load, particularly that of DIN, provided by the outfall is not expected to result in local nuisance blooms of phytoplankton or macroalgae. In the case of the former, given that the generation time for coastal phytoplankton is normally in the range of a few days rather than hours (Weiler & Chisholm 1976), the rate of dilution as nutrients travel down-current from the discharge would be expected to preclude any measurable discharge-related enhancement of phytoplankton (including nuisance species).

Monitoring of water quality off the Boulder Bank between August 2020 and December 2021 shows highest concentrations of chlorophyll-*a* in late winter and spring (Figure 11). The range of concentrations recorded during the 2020–21 survey was 0.0002–0.0046 g/m^3. These values are consistent with those recorded in Tasman Bay between 1998 and 2003 (0.0005–0.0029 g/m^3: Newcombe et al. 2015), and the characterisation of the bay as oligotrophic (low) to mesotrophic (moderate) in terms of phytoplankton productivity.

Monitoring around the NWWTP outfall has shown that populations of macroalgae are sparse (apart from encrusting corallines), and there is no evidence of increased abundance closer to the outfall.

At wider spatial and temporal scales, cumulative effects of nutrient inputs may occur if the combined sources of nutrients enrich the waters of Tasman Bay to a sufficient degree. The discharge may contribute about 4% of the annual load of total nitrogen from terrestrial sources, which is a significant amount. However, the overwhelming driver of nutrient concentrations in the bay is the intrusion of offshore waters (Zeldis 2008), which is estimated to contribute 90% of DIN. Consequently, although terrestrial sources of nitrogen are important for coastal primary productivity, there seems to be little potential for over-enrichment (eutrophication). This is likely to be particularly true for open areas of coast such as that around the NWWTP outfall.

As part of the consent application process for the Bell Island WWTP discharge, Gillespie and Berthelsen (2017) estimated the capacity of the waters of inner Tasman Bay to assimilate additional nitrogen loading. Comparisons of inner Tasman Bay nutrient concentrations and areal loading estimates with reported Aotearoa New Zealand and overseas guidelines / standards suggested that there was capacity for assimilation of additional nutrients without expression of adverse enrichment effects. Based on the average inner bay TN and DIN concentrations summarised in their report, these nutrients could possibly be increased by approximately 24% and 38%, respectively, without causing a shift to beyond a ‘very low’ eutrophication status.

5.1.2. Ammonia toxicity

Nitrogen in seawater is mainly present as nitrate and, in some inshore coastal areas, as ammoniacal nitrogen (consisting of ammonium plus ammonia). In general, these concentrations are low in summer and greater in winter, and are low and variable at the sea surface, and increase with depth. Low surface levels in summer are generally caused by phytoplankton growth.

Ammoniacal nitrogen is an important nutrient but is also toxic to organisms at high concentrations. The concentration ratio of the more toxic unionised form (ammonia, NH_3) to the ionised form (ammonium, NH_4^+) in solution depends on several factors, most notably temperature, pH, concentration of dissolved oxygen and salinity. Ammoniacal nitrogen toxicity increases with temperature and pH and decreases with concentration of dissolved oxygen and salinity. For example, the ANZG default guideline for marine waters at pH 8.0 is 0.91 g/m^3 total ammoniacal nitrogen but decreases to 0.42 g/m^3 at pH 8.4. The marine guidelines are based on toxicity data for three species of fish, 15 species of crustaceans, two species of molluscs and a rotifer (ANZG 2018).

Concentrations of total ammoniacal nitrogen in the NWWTP discharge between August 2020 and December 2021 ranged 0.01-33 g/m^3. On at least some occasions, therefore, the undiluted effluent is likely to be ecotoxic at the point of discharge. To estimate the potential for toxicity following initial mixing and dilution in the receiving environment, concentrations in the discharge were compared with the ANZG (2018) trigger values in Table 8 after allowing for a 220-fold dilution. This is the minimum dilution predicted by modelling at the edge of the near field[15] based on a range of discharge flow rates and ambient current speeds (MetOcean Solutions Ltd 2022). This dilution occurred 155 m

[15] The near field is defined by MetOcean Solutions Ltd (2022) as ‘the zone of strong initial mixing where the so-called near-field processes occur (i.e. the initial jet characteristic of momentum flux, buoyancy flux and outfall geometry influence the jet trajectory and mixing of a wastewater discharge)‘, often referred to as the ‘boil’.

down-current from the point of discharge (see MetOcean Solutions Ltd 2022, table 3-1).

A toxicity safety factor, defined as the ratio of the ANZG (2018) guideline to the discharge concentration at 220× dilution, was calculated for the minimum, median and maximum concentrations of ammoniacal nitrogen measured in the effluent between August 2020 and December 2021 (Table 8, n = 36). A value larger than 1 indicates that the receiving water concentration was below the water quality guideline and no toxicity to fish or other marine organisms would be anticipated.

The highest toxicity potential arises from a combination of the highest ammoniacal nitrogen concentration measured in the effluent (33 g/m^3; Table 8) and the highest pH measured in the receiving environment (8.4 at Schnappers Point; Table 8). This worst-case combination gives a toxicity factor of 2.8 (Table 8), indicating that the potential for ammonia toxicity outside the near field of the discharge is very low. The potential will be lower still at the boundary of the mixing zone, where the estimated minimum dilution is 355:1 under the current discharge conditions and 280:1 under predicted future (2059) conditions.[16] Other mitigating factors include the fact that ammoniacal-N concentrations in the discharge are typically higher in winter, when seawater temperatures, and consequently ammonia toxicity, are lower. Ammonia is also a non-conservative contaminant and will be metabolised to other forms of nitrogen by microbial activity in the water column after discharge.

[16] Estimates of present and future dilutions at the mixing-zone boundary, based on MetOcean modelling, provided by Rob Lieffering, SLR, in an email to Ross Sneddon, 18 July 2023.

Table 8. Concentrations of total ammoniacal nitrogen measured in the Nelson North WWTP discharge and the pH of seawater at the two monitoring stations off the Boulder Bank during the period August 2020 to December 2021. Concentrations are g/m^3. ANZG (2018) guidelines for the protection of aquatic life are shown adjusted for ambient pH and are those for slightly to moderately disturbed systems (equivalent to protection of 95% of species). Also shown are toxicity safety factors (ratio of ANZG guideline to the concentration of ammoniacal nitrogen following initial dilution). Source of ammoniacal nitrogen and pH data: Nelson City Council.

	Median (range)	n	ANZG (2018) marine guideline value (g/m^3 as total ammoniacal-N)
pH (CM AW 01 – Adjacent to NWWTP)	8.1 (7.9–8.2)	21	0.75 (1.1–0.62)
pH (CM AW 02 – Schnappers Point	8.0 (7.1–8.4)	38	0.91 (3.56–0.42)
Total ammoniacal-N monitored in the NWWTP discharge	12.2 (0.01–33)	36	
Toxicity safety factor adjacent to NWWTP	13.5 (4.1–24,200)		
Toxicity safety factor at Schnappers Point	16.4 (2.8–78,320)		

5.1.3. *Non-nutrient contaminants*

From the results of the studies by Barter and Forrest (1998) summarised in Section 3.1, it is very unlikely that toxic contaminants in the wastewater (such as trace metals and ammonia) will be present in the receiving environment at concentrations that would have any significant adverse effect on aquatic life.

More recent measurements of the concentrations of trace metals and other toxicants in the wastewater (Table 9) are consistent with the 1996–98 data, in that all measured values were well below the consent limits. The consent limits were presumably derived by multiplying the guidelines for the protection of aquatic life that were applicable at the time the limits were chosen (ANZECC 1992; shown in Table 9) by the expected dilution at the boundary of the mixing zone. The ANZECC guidelines apply to the receiving waters rather than the discharge. Because concentrations in the discharge were below the consent limit, both the historical and recent data suggest that significant adverse effects on aquatic life are unlikely.

The ANZECC (1992) guidelines have since been updated (in 2000 and 2018). Apart from mercury and phenols, the guideline concentrations for the toxicants listed in Table 9 have all decreased from those in ANZECC (1992) to the concentrations given in the most recent updates (ANZG 2018: see Table 9). This raises the possibility that the consent limits, being based on the less stringent 1992 guidelines, may not adequately protect aquatic life.

Among the 2016–19 data, however, some of the toxicants (lead, cyanide [apart from in 2019] and phenols) met the ANZG (2018) guidelines without consideration of dilution effects. The analytical detection limits for the remaining toxicants were above their respective ANZG guidelines, so no true comparison can be drawn. Nevertheless, if the measured concentrations were at the detection limits, the 2018 guidelines would be met with dilution factors ranging from 1.4 (cadmium, nickel and zinc) to 21 (mercury). These dilution factors are well within those expected at the boundary of the mixing zone (355 in 2021, 280 in 2059; Rob Lieffering, SLR, email to Ross Sneddon, 18 July 2023). Among the 2020–21 data, only copper exceeded the ANZG guideline (cyanide was not measured during this period). Copper met the guideline with 2.6 times dilution. Method detection limits improved during the monitoring period, and concentrations were below respective guideline values for all contaminants that were not present at concentrations above their respective detection limits, except in the case of phenols.

Table 9. Concentrations of toxicants measured annually in the Nelson North WWTP discharge as part of consent monitoring for the period July 2016–July 2019 and c. monthly from August 2020–December 2021. Note: 2020–21 values are medians (and 95th percentiles), *n* = 15. All concentrations are g/m^3. Mercury is the inorganic form. Consent limits, ANZECC (1992) and ANZG (2018) guidelines for the protection of aquatic life are also shown. ANZG (2018) values are those for slightly to moderately disturbed systems (equivalent to protection of 95% of species, except for cadmium, mercury and nickel, which are 99% of species). Source: Nelson City Council.

Toxicant	**Jul 2016**	**Jul 2017**	**Jul 2018**	**Jul 2019**	**Aug 2020 – Dec 2021**	**Consent**	**ANZECC (1992)**	**ANZG (2018)**
Cadmium	<0.0011	<0.0011	<0.0011	<0.0011	<0.000053	0.275	0.002	0.0007
Chromium VI [a]	<0.011	<0.011	<0.010	<0.011	0.0022 (0.0037)	1.37	0.05	0.0044
Copper	<0.011	<0.011	<0.011	<0.011	0.0034 (0.0048)	0.065	0.005	0.0013
Lead	<0.0021	<0.0021	<0.0021	<0.0021	0.0005 (0.0012)	0.22	0.005	0.0044
Mercury	<0.0021	<0.0021	<0.0021	<0.0021	<0.00008	0.02	0.0001	0.0001
Nickel	<0.011	<0.011	<0.011	<0.011	0.0046 (0.0075)	3.5	0.015	0.007
Zinc	<0.021	<0.021	<0.021	<0.021	0.0104 (0.0153)	0.75	0.05	0.008
Cyanide	<0.0010	<0.0010	<0.0010	0.04	Not recorded	0.2	0.005	0.004
Total phenols	0.02	0.02	0.02	<0.02	<0.02	20	0.05	0.4 / 0.011[b]

[a] The chromium guideline used is that for chromium VI (the most toxic form of the metal).
[b] The guidelines shown are for phenol / pentachlorophenol (the 99% level of protection for the latter, as recommended by ANZG).

Concentrations of toxic contaminants (trace metals and total phenols) were measured on a single occasion (24 March 2021) at two sites just offshore from the Boulder Bank ('Adjacent to NWWTP' and 'Schnappers Pt': see Figure 10). The concentration of copper exceeded the ANZG (2018) guideline for 'slightly to moderately disturbed' systems (see Table 9) at both sites, but by a factor of 25 times at Schnappers Point. The concentration of zinc (0.064 g/m^3) also exceeded the ANZG trigger at Schnappers Point, but not at the site adjacent to the NWWTP.

Wastewater treatment plant discharges are one of the main anthropogenic sources of phenolic compounds in coastal waters (Roark 2020). Concentrations of total phenols (rather than the concentrations of individual phenolic compounds) were measured in the receiving environment samples. The concentrations of total phenols (< 0.02g/m^3 adjacent to the NWWTP, 0.09 g/m^3 at Schnappers Point) were below the ANZG (2018) guideline for phenol (the parent compound: 0.4 g/m^3) at both sites. The concentration at Schnappers Point was above the guideline for pentachlorophenol (0.022 g/m^3), the only other phenolic compound for which a guideline has been derived. It is therefore possible that, if pentachlorophenol represented more than c. 25% of the total phenols, the guideline could have been exceeded.

The high concentrations of copper, zinc and total phenols recorded at Schnappers Point were much higher than the 95%ile values of the 15 samples of effluent tested between August 2020 and December 2021. This suggests that the discharge is unlikely to be the source or that these were anomalously high concentrations and not characteristic of the effluent.

The present mixing zone for the discharge extends 250 m from the outfall in each direction parallel to the shore. The discharge is estimated to be diluted at least 220-fold at 150 m from the outfall (the 'near field': MetOcean Solutions Ltd 2022) and at least 280-fold (depending on flow) at the boundary of the current mixing zone (see above). Consequently, adverse effects on visual clarity, suspended-solid concentration and dissolved oxygen are expected to be negligible.

5.1.4. Compliance with section 107 of the Resource Management Act 1991

A discharge shall not be granted a permit if, after reasonable mixing, it is likely to give rise to all or any of the following effects:

1. the production of any conspicuous oil or grease films, scums or foams, or floatable or suspended materials
2. any conspicuous change in the colour or visual clarity
3. any emission of objectionable odour
4. any significant adverse effects on aquatic life.

Barter and Forrest (1998) noted that in calm weather the discharge plume was discernible mainly as a change in surface tension where the buoyant low-salinity plume reaches the water surface. During 20 visits to the outfall over the course of a year, they did not see any conspicuous visual effects from the outfall, nor any floating or suspended material. The plume was not visible and vertical profiles of the water column using a transmissivity sensor did not indicate any change in water clarity. Overall, Barter and Forrest's (1998) observations of the discharge area and measurements of water clarity found no evidence of films, scums, foams, floatable or suspended materials or conspicuous changes of colour or clarity.

Qualitative examination of the sequence of satellite images of the area around the outfall in Google Earth™ revealed three images[17] in which a plume appeared to be visible (from a total of 18 images of sufficient quality that a plume was likely to be visible if present). One possible cause for these visible plumes is the discharge of microalgae derived from the treatment ponds. Whatever the cause, the satellite images do not suggest that it is a regular occurrence and, when it does occur, the visibility of the plume will be dependent on weather conditions, with windy conditions likely to make it less conspicuous.

Emission of odour is outside the scope of this assessment and potential effects on aquatic life are addressed in Sections 5.2–5.5.

5.2. Seabed

Based on their survey of sandy and boulder habitats around the outfall, Barter and Forrest (1998) were unable to detect any obvious adverse effects of the discharge, even though it had been in operation for c. 30 years and had been discharging untreated sewage for the first 10 years. This applied to both the physical habitats and the communities of organisms living in or on them.

From this earlier study, and the present low concentrations of trace metals and organic matter in the sediments (see Section 3.2), we do not expect that continued operation of the outfall will result in future adverse ecological effects on boulder or sand habitats or their respective communities. Nor is there any reason to expect that improvement in wastewater quality would have any discernible beneficial ecological effect, given the similarity between stations surveyed near and remote from the outfall (Barter & Forrest 1998).

5.3. Kaimoana

The lack of any detectable effects on organisms living around the outfall (Section 5.2) suggests that there will be no consequent effects on the abundance or type of

[17] The dates of these images were January 2013, August 2018 and June 2019.

invertebrate or macroalgal kaimoana available in the vicinity. In any case, abundances are naturally low and the area does not appear to be targeted for collection. The fish fauna of the Boulder Bank also appears to be of low diversity and abundance, and if anything, fish abundance may be higher around the outfall. Consequently, adverse effects on kaimoana availability from the continued operation of the discharge are expected to be less than minor.

Barter and Forrest (1998) concluded that 'the water quality effects of the sewage discharge are minor and localised, and we consider the actual and potential impacts on recreational uses and values in the area to also be minor'. They attributed this to the level of wastewater treatment, which produces a 'relatively low strength effluent', coupled with an outfall structure and location that facilitates dispersion and dilution. This assessment suggests that any broader-scale effects on finfish are likely to be negligible.

There will, however, be general restrictions on food gathering in Nelson coastal waters related to microbial quality, particularly of filter-feeding shellfish such as mussels and oysters. NCC is currently trying to address the problem of microbial contamination of stormwater.[18] Until this problem is resolved, NCC advises that shellfish should not be taken from within the enclosed waters of the Waimea Inlet and the Nelson Haven as they may exceed maximum microbiological guideline levels due to stormwater contamination. There are also notices advising against collection of shellfish at the shore end of the NWWTP outfall. The risks associated with consumption of contaminated shellfish are addressed in detail in the human health-risk assessment report (Hudson & Wood 2023).

5.4. Valued habitats and Threatened and At Risk species

5.4.1. Background

This section provides an assessment of the marine flora and fauna of the outfall site in the context of Policy 11 of the New Zealand Coastal Policy Statement (NZCPS) and Appendix 13 of the Draft Whakamahere Whakatū Nelson Plan (WWNP).

The purpose of Policy 11 (a) of the NZCPS is to protect indigenous biological diversity in the coastal environment by avoiding adverse effects on:

1. indigenous taxa that are listed as Threatened or At Risk in the New Zealand Threat Classification System (NZTCS) lists
2. taxa that are listed by the International Union for Conservation of Nature and Natural Resources as Threatened

[18] http://www.nelson.govt.nz/environment/water-3/river-stream-water-flows/gathering-shellfish-in-nelson (accessed 27 February 2023).

3. indigenous ecosystems and vegetation types that are threatened in the coastal environment, or are naturally rare
4. habitats of indigenous species where the species are at the limit of their natural range, or are naturally rare
5. areas containing nationally significant examples of indigenous community types and
6. areas set aside for full or partial protection of indigenous biological diversity under other legislation.

Appendix 13 of the WWNP is based on Policy 11 of the NZCPS and has two key parts. Part A addresses habitats and Part B addresses vulnerable species.

Because both policies provide strong direction to avoid adverse effects on their listed values, and therefore may be given particular weight in NCC's decision on discharge consent, we provide supporting evidence that:

- it is reasonable to conclude that the listed values are not present
- even if listed values are present, there are unlikely to be adverse effects from the discharge.

The assessment adopted the following approach:

1. identify which of the taxa covered in the policies do or do not occur in the habitats present near the existing outfall
2. confirm that there are no records (if that is the case) of the listed values being found in the proposal area, and (if possible) set out why it is reasonable to conclude that, in all likelihood, the listed values are not present
3. assess whether, if the values were present (but unrecorded), there would or would not be adverse effects on them from the proposed discharge (i.e. continuation of the existing discharge).

In relation to point 3, it is noted that the proposal for which resource consent will be sought does not involve physical works above (or below) the high-tide level. Therefore, no land-based activities need to be taken into consideration. For this reason, it is not necessary to consider effects on terrestrial plants or birds.

5.4.2. Valued habitats

Table 1 of Appendix 13 of the WWNP lists key habitats, of which the following are relevant in the present context:

- soft-sediment subtidal (non-estuarine)
    - shellfish reefs (biogenic habitat created by shellfish, including mussels and oysters, and horse mussel beds)

    - loose sediment (invertebrate dominated, often mobile disturbance-tolerant taxa).
- rocky reefs (hard-substrate-dominated)
    - reef and seaweed communities.
- sea surface and water column
    - sea surface and water column communities (including planktonic productivity, benthic and pelagic fish).

Of the listed habitats, soft sediment dominates the vicinity of the mixing zone and likely extends subtidally for much of the length of the Boulder Bank in the nearshore environment (Sections 2.3 and 3.2). This sandy habitat does not appear to be associated with significant shellfish beds or other extensive biogenic features. Closer to shore, mobile sands transition to the hard substrate of the Boulder Bank. Table 1 of Appendix 13 of the WWNP notes that movement of boulders in intertidal and shallow subtidal areas limits potential for communities to become established. Compiled survey observations indicated a diverse but relatively limited epibiotic community associated with the nearshore boulder substrates. Observations from 5-yearly diver monitoring has indicated little to no discernible effect on these habitats attributable to the NWWTP discharge (Sneddon 2018; Morrisey 2021).

5.4.3. Threatened and At Risk species

Because of the difficulty of demonstrating the absence of small, rare and cryptic plants and invertebrates, we took an indirect approach to assessing the likelihood of any Threatened or At Risk species occurring at the discharge location. We collected information on the distribution and habitat preferences of Threatened and At Risk species (where available) and used this to identify which species could potentially occur at the discharge location.

Fourteen species of macroalgae in the Threatened or At Risk categories of the NZTCS are listed as having been recorded in the northern South Island (not necessarily Tasman Bay) in a review by Nelson et al. (1992). That review was based on a limited number of surveys and collections, so the list is by no means exhaustive, but it does identify some Nationally Endangered species that *could* occur at the discharge location. There have also been major taxonomic revisions of some macroalgal groups since Nelson et al.'s list was published. The NZTCS risk assessment for macroalgae lists more than 600 taxa for which there are insufficient data to assess threat status. In addition, the Threatened and At Risk lists include taxonomically indeterminate taxa – i.e. types that cannot be assigned to a known species (two in the Threatened list and six in the At Risk list). This uncertainty emphasises two important points: the conservation status of species present in Tasman Bay may not be well understood, and it may not be possible to assign specimens from the bay to taxonomically indeterminate Threatened or At Risk forms.

There are also several invertebrates listed as Threatened or At Risk that *could potentially* occur at the discharge location. The stalked barnacle *Idioibla idiotica* (Threatened: Nationally Critical) has been recorded from intertidal to deep subtidal locations around Aotearoa New Zealand (Buckeridge & Newman 2006). This species was apparently once relatively common in the low intertidal zone in Aotearoa New Zealand but had not been collected in this habitat for at least a decade at the time of Buckeridge and Newman's paper. A single specimen was collected from 50 m water depth in Piwhane / Spirits Bay (hereafter Spirits Bay) in Northland in 1998. The likelihood of it occurring at the discharge location is, therefore, unknown.

The lampshell (brachiopod) *Pumilus antiquatus* (Threatened: Nationally Critical) has been recorded on rocks and boulders below the low-tide mark from three locations in the South Island: Lyttelton Harbour / Whakaraupō, near Karitane and Otago Harbour (Bowen 1968). Given that its distribution is poorly known, it is possible, if unlikely, that it could occur at the discharge location.

The polychaete worm *Spio aequalis* (Threatened: Nationally Endangered) has been recorded from the Chatham Islands, Moeraki, Banks Peninsula, Wellington and Northland (Aupōuri Peninsula). It is one of the largest species of spionid (5–15 cm) and occurs on exposed coasts, possibly burrowing in sand under stones.[19] This species could potentially occur at the discharge location, given the presence of suitable habitat, but given its conspicuous size, the fact that it has not been recorded from Tasman Bay makes this unlikely.

The golden limpet, *Cellana flava* (At Risk: Declining), is described by Willan et al. (2010) as frequent to common in mid- to low intertidal areas from Dunedin to East Cape and the Chatham Islands. This species could potentially occur at the discharge site.

The volute gastropod *Alcithoe davegibbsi* (At Risk: Declining) has been collected off Spirits Bay, but we were unable to find information on habitat type (probably soft sediments, similar to other species in the genus) or depth range. It is conservatively assumed that this species could occur at the discharge location, given the presence of potentially suitable habitat.

The benthic *Octopus kaharoa* (At Risk: Declining) has been recorded in the depth range 73–540 m on soft substrata.[20] It has been collected from Northland, Taranaki, the Bay of Plenty, East Cape, Hawke's Bay, the coast of Wairarapa and the west and east coasts of the South Island as far south as the Canterbury Bight.[21] Given that it is

[19] See www.inaturalist.org/posts/6784-rediscovery-of-spio-aequalis-after-missing-for-over-50-years (accessed 17 December 2019).
[20] See https://www.iucnredlist.org/species/163340/1000039 (accessed 17 December 2019).
[21] See https://www.gbif.org/species/4357191 (accessed 17 December 2019).

rare, and records are sparse, it may also occur in shallower water than previously recorded and could potentially occur at the discharge location.

Table 2 of Appendix 13 of the WWNP lists key indigenous species, none of which are directly relevant in the present context. Marine mammals, wading birds and seabirds are included but are beyond the scope of the present report. The migratory fish listed all depend on access to rivers, of which there are none in the project area.

We emphasise that none of the macroalgae or invertebrates identified above have been recorded in Tasman Bay, and that their presence at the project site is only possible rather than probable. There are no features of the outfall location that suggest these species are more likely to occur there than at other locations along the adjacent coast.

As noted in Section 3.2, the New Zealand lancelet (*Epigonichthys hectori*) was found to be abundant in sandy sediments around the outfall by Barter and Forrest (1998). Although patchily distributed nationally and naturally uncommon, lancelets are not listed in the Threatened or At Risk categories of the NZTCS. However, while documented also from Croisilles Harbour and the Marlborough Sounds, this species may be near the southern limit of its natural range. The presence of a healthy population close to the established outfall and the species' observed preference for low-silt sandy habitats suggested that the discharge was having little effect on the benthic environment.

5.5. Risk assessment summary

The approach to risk assessment was based on modifications of those proposed by EIANZ (2018) and Burgman (2005). The levels of risk were derived from sequential consideration of the following factors (the categories of each factor are shown in Table 11):

- the ecological value of the organisms or habitats affected
- the spatial scale and duration of the effect
- the magnitude, or consequences, of any effect that occurs
- the likelihood of the effect occurring.

The level of ecological risk is derived from a combination of the value of the ecological feature and the magnitude of the effect (Table 10). In the absence of any features of special ecological or conservation interest in the coastal area around the outfall, the habitats and biota are considered to be of moderate value.

Table 10. Level of risk of an adverse effect.

		Ecological or conservation value			
		Very high	**High**	**Moderate**	**Low**
Magnitude	**High / severe**	Very high	Very high	Moderate / high	Low / moderate
	Moderate / medium	High	Moderate / high	Low	Very low
	Low / minor	Moderate	Low / moderate	Low	Very low
	Negligible	Low	Very low	Very low	Very low

The level of risk was assessed for the following potential hazards:

- Eutrophication from inputs of nutrients to Tasman Bay
- Enrichment of sediments by nutrients and organic matter, resulting in altered physical habitat and reduced dissolved oxygen concentrations
- Toxic effects from inputs of trace metals
- Reduced water quality from inputs of biochemical oxygen demand (BOD) and suspended solids, resulting in reduced dissolved oxygen concentrations and smothering of organisms, respectively.

For each of the ecological features present in the receiving environment, the risk of significant adverse effects was assessed as **Low** (Table 11). As described in Sections 5.1–5.3, this assessment derives primarily from monitoring data (benthic habitats), but also from estimates of dilution (local effects of nutrients, BOD and toxicants) and relative loads (effects of nutrients on the wider coastal area).

The ‘Low’ risk status was assumed to apply equally to Threatened and At Risk taxa and, consequently, adverse effects will be avoided. We note that the assumption of a low level of risk to general habitats and biota at the discharge location applying also to Threatened and At Risk invertebrate taxa incorporates a level of unavoidable uncertainty. It is possible that some of these taxa are more sensitive than others to habitat disturbance or to altered nutrient concentrations or salinities. However, the lack of comprehensive information on these taxa makes it impossible to predict effects with certainty. Conversely, we are also assuming that these taxa could be present, but in most cases this is unlikely. It is also relevant that the outfall has been operating since 1968 (and under the present wastewater treatment regime since 2010). Hence, additional future effects on the wider receiving environment (rather than that immediately around the outfall) are unlikely.

Table 11. Summary of potential ecological effects of the Nelson North WWTP discharge on the coastal receiving environment.

Potential environmental effect	Ecological feature	Value	Spatial scale of effect	Duration of effect	Magnitude of effect	Likelihood of effect	Level of risk
Eutrophication from inputs of nutrients to Tasman Bay	Benthic micro- and macroalgae around outfall	Moderate	Small	Persistent (duration of activity)	Low / minor (based on monitoring)	High	Low
	Phytoplankton and macroalgae in Tasman Bay	Moderate	Large	Persistent (duration of activity)	Low / minor (based on relative load)	High	Low
Enrichment of sediment by nutrients and organic matter, resulting in altered physical habitat and reduced dissolved oxygen concentrations	Biota of sediments adjacent to outfall, including kaimoana species	Moderate	Small	Persistent (duration of activity)	Low / minor (based on monitoring)	Moderate	Low
Toxic effects from inputs of trace metals and other toxicants	Biota of sediments and hard substrata adjacent to outfall, including kaimoana species	Moderate	Small	Persistent (beyond duration of activity)	Low / minor (based on expected dilution)	High	Low
Reduced water quality from inputs of BOD and suspended solids, resulting in reduced oxygen concentrations and smothering of organisms	Biota of sediments and hard substrata adjacent to outfall, including kaimoana species	Moderate	Small	Persistent (duration of activity)	Low / minor (based on expected dilution)	High	Low

Definition of terms used in table:

Spatial scale of effect: Small (tens of metres), Medium (hundreds of metres), Large (> 1 km)

Duration of effect: Short (days to weeks), Moderate (weeks to months), Persistent (years or more)

Magnitude of effect: Negligible (no or very slight change from existing conditions), Low / minor (minor change from existing conditions, minor effect on population or range of the feature), Moderate / medium (loss or alteration to key element(s) of existing conditions, moderate effect on population or range of the feature), High / severe (major or total loss of key element(s) of existing conditions, large effect on population or range of the feature)

Likelihood of effect: Low (< 25%), Moderate (25–75%), High (> 75%)

Level of risk: Very low (effect too small to be discernible or of concern), Low (discernible effect but too small to affect others), Low / moderate (noticeable but will not cause any significant adverse effects), Moderate (noticeable effect that may cause adverse impact but could be mitigated), Significant (noticeable effect and will cause serious adverse impact but could be mitigated)

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