Appendix M Assessment of seabed effects

Ngāi Tahu Seafood Resources Limited

Hananui Aquaculture Project

Seabed Assessment of Effects

Evidence of Holly Bennett regarding *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* and Proposed Conditions

Holly Bennett
11-10-2025

Introduction

My name is Holly Bennett.

My role in relation to the Hananui Aquaculture Project ("**HAP**") has been to provide expert evidence in relation to seabed effects. I was the lead author of the *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* which is provided within **Appendix M** of the application.

This evidence has been prepared to accompany the application by Ngāi Tahu Seafood Resources Limited ("**NTS**") for approvals required for the HAP under the Fast-track Approvals Act 2024 ("**FTAA**"). It has been prepared on the understanding that the process for determining applications under the FTAA does not require a hearing to be held, and accordingly the purpose of this evidence is to confirm that, relative to my area of expertise, the *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* provides an appropriate description of the relevant environment, the proposed activities comprising the effects of the HAP on that environment, and the way those effects are proposed to be managed.

My findings are set out in full in the *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* included within **Appendix M** of the application.

While this application is not being considered by the Environment Court, I confirm that I have read the Code of Conduct for expert witnesses contained in the Environment Court of New Zealand Practice Note 2023 and that I have complied with it when preparing this evidence. Other than when I state I am relying on the advice of another person, this evidence is within my area of expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions that I express.

Qualifications and Experience

I am a Marine Ecologist.

I graduated from Victoria University of Wellington with a PhD in Marine Science.

I have over 10 years of research experience examining the impacts of human activities on the marine environment, with a particular focus on aquaculture. At the Cawthron Institute, my work centres on understanding the interactions between aquaculture operations and the surrounding ecosystem, with an emphasis on the benthic effects of salmon farming. I have led and contributed to numerous commercial and applied research projects, including site selection for prospective aquaculture developments, routine monitoring of marine farms, comprehensive environmental impact assessments and research assessing the ecological effects of open ocean aquaculture on seabed communities.

I have authored or co-authored 12 papers that have been published in high profile peer-reviewed international scientific journals, as well as >40 other technical reports including co-authoring the most recent update to the Best management practice guidelines for salmon farms in the

Marlborough Sounds Part 1: Benthic environmental quality standards and monitoring protocol (Version 1.2, MPI 2022).

In providing this evidence in relation to seabed effects, I have considered the following matters as relevant to that topic:

- The project description provided by NTS as set out in section 6 of the application;
- The description of the existing environment, the effects of the HAP on that environment and their significance, and the proposed management and mitigation measures to manage those effects all as set out in the assessment of environmental effects accompanying the application;
- The technical assessments of
    - *Hananui Aquaculture Project: marine biosecurity assessment of effects*, Forrest et, al, 2025
    - *Water column assessment: Hananui Aquaculture Project*, Wilson 2025; and
- A range of references describing the existing seabed environment, pre-dredging conditions, sediment and habitat types, and elements of the oyster and blue cod fisheries
- MPI site scoping investigations for the area
- Ngāi Tahu Seafood Biosecurity Management Plan

Confirmation of Contents of Report and Proposed Conditions

I confirm that in my opinion the *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* contain an accurate and appropriate description of the environment, the actual and potential effects of the HAP, and the recommended actions to manage those effects within my area of expertise.

I confirm that in my opinion the contents of the *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* may be relied on in making a decision on the approvals sought for the HAP and confirm that provided effects within my area of expertise are managed as proposed in the application those effects will not be unacceptable and will be managed to a standard that I consider meets good practice.

I confirm that I have reviewed the conditions that NTS proposes for the approvals being sought as they relate to my area of expertise. I confirm that in my opinion, those proposed conditions are appropriate.

[redacted]

Holly Bennett

11.11.25

Ngāi Tahu Seafood Resources Limited

Hananui Aquaculture Project

Seabed Assessment of Effects

Evidence of Malcolm David Smeaton regarding *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura, Cawthron Report 4188*

Malcolm David Smeaton
11-11-2025

Introduction

My name Malcolm David Smeaton.

My role in relation to the Hananui Aquaculture Project ("**HAP**") has been to provide expert evidence in relation to the benthic effects assessment. I was co-author of *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* which is provided within **Appendix M** of the application. I undertook the depositional modelling for this report and wrote sections of the report that relate to the depositional modelling component of the assessment.

This evidence has been prepared to accompany the application by Ngāi Tahu Seafood Resources Limited ("**NTS**") for approvals required for the HAP under the Fast-track Approvals Act 2024 ("**FTAA**"). It has been prepared on the understanding that the process for determining applications under the FTAA does not require a hearing to be held, and accordingly the purpose of this evidence is to confirm that, relative to my area of expertise, *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* provides an appropriate description of the relevant environment, the proposed activities comprising the effects of the HAP on that environment, and the way those effects are proposed to be managed.

My findings are set out in full in the Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura included within **Appendix M** of the application.

While this application is not being considered by the Environment Court, I confirm that I have read the Code of Conduct for expert witnesses contained in the Environment Court of New Zealand Practice Note 2023 and that I have complied with it when preparing this evidence. Other than when I state I am relying on the advice of another person, this evidence is within my area of expertise. I have not omitted to consider material facts known to me that might alter or detract from the opinions that I express.

Qualifications and Experience

I am an Oceanographer at the Cawthron Institute and have held this position since October 2019. Prior to this, I worked for the Cawthron Institute on a casual basis from January 2017 to October 2019 while completing my PhD at University of Otago. Since starting work at Cawthron I have authored/co-authored 32 client reports, 5 conference papers, 8 journal articles, and 4 advice letters. I have been an expert witness for two resource consent hearings and one Covid-19 fast track application.

I graduated from University of Otago in 2020 with a PhD in Oceanography. I was also awarded a BAppSc (Hons I) in Energy Management from this institute in 2012.

I specialise in physical oceanography and mathematical modelling. Depositional modelling, and Lagrangian particle tracking in general, is one of my specific areas of expertise. I completed depositional modelling for the Blue Endeavour salmon farm application in 2021, the original

Hananui application in 2022, and have undertaken depositional models of existing farms in the Marlborough Sounds. Recently, I was contracted by Ministry for Primary Industries to undertake depositional modelling at several potential aquaculture settlement areas for feasibility purposes. Currently, I am working on a depositional model for a novel finfish pen design as part of the MBIE-funded Re-imagining Aquaculture project.

Outside of aquaculture, I have experience using Lagrangian particle tracking models for marine biosecurity purposes as well as mussel bed restoration efforts. I am currently leading a project that uses particle tracking models to inform a surveillance strategy for the control and eradication of sea spurge (*Euphorbia paralias*) in New Zealand and recently completed a similar project to assess the risk of ballast water exchange as a vector for invasive species.

In providing this evidence in relation to depositional modelling component of the benthic effects assessment, I have considered the following matters as relevant to that topic:

- The project description provided by NTS as set out in section 6 of the application;
- The description of the existing environment, the effects of the HAP on that environment and their significance, and the proposed management and mitigation measures to manage those effects all as set out in the assessment of environmental effects accompanying the application;
- Relevant scientific literature concerning recent advances and applications of depositional modelling methods;
- The hydrodynamic model report of Oceanum (2025) concerning the hydrodynamic model used as part of the depositional modelling:
    - Oceanum. 2025. Oceanographic modelling for the proposed Hananui Aquaculture Project in Te Ara a Kiwa. Report 1 – hydrodynamic model. Raglan: Oceanum. Prepared for SLR Consulting Ltd.

Confirmation of Contents of Report and Proposed Conditions

I confirm that in my opinion *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* contains an accurate and appropriate description of the environment, the actual and potential effects of the HAP, and the recommended actions to manage those effects within my area of expertise.

I confirm that in my opinion the contents of *Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura* may be relied on in making a decision on the approvals sought for the HAP and confirm that provided effects within my area of expertise are managed as proposed in the application those effects will not be unacceptable and will be managed to a standard that I consider meets good practice.

I confirm that I have reviewed the conditions that NTS proposes for the approvals being sought as they relate to my area of expertise. I confirm that in my opinion, those proposed conditions are appropriate.

[redacted]

Malcolm David Smeaton

Oceanographer

CAWTHRON

Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura

Cawthron Report 4188

World-class science
for a better future

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Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura

Holly Bennett, Malcolm Smeaton, Lisa Floerl, Paula Casanovas

Prepared for Ngāi Tahu Seafood Resources

Glossary

Term	Definition
DML	Discovery Marine Limited
LINZ	Land Information New Zealand
MBES	Multibeam echo sounder
OLE	Outer limit of effect
ROV	Remotely operated vehicle
ZME	Zone of maximum effect

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Executive summary

Ngāi Tahu Seafood Resources (Ngāi Tahu Seafood) proposes developing an area of ~ 1,285 ha for offshore salmon farming 2–6 km off northern Stewart Island / Rakiura, ~ 13 km northwest of Oban (referred to as the Hananui proposal area). The full development will comprise four farms, each with up to two blocks of 10 pens. A staged development is proposed, beginning with Stage 1 (one block of 10 pens at each of the four farm sites; up to 15,000 tonnes per annum [$t{\cdot}yr^{-1}$] feed discharge), before progressing to Stage 2 (two blocks of 10 pens at each of the four farm sites; up to 25,000 $t{\cdot}yr^{-1}$ feed discharge). This report characterises the seabed environment both within the Hananui proposal area and across a wider (~ 12,500 ha) 'survey area'. It then assesses the potential effects of farming on seabed habitats and communities.

Existing environment

The Hananui proposal area lies in 20–40 m of water with strong mean current velocities (0.56 $m{\cdot}s^{-1}$ near the surface and 0.34 $m{\cdot}s^{-1}$ near the seabed) flowing along a northwest–southeast axis. Sediments are predominantly coarse sands with varying amounts of shell hash, gravel and a small proportion of mud. They are well oxygenated, contain low organic content and support macrofaunal communities of naturally low abundance but moderate diversity.

Six main habitat types were mapped across the total survey area:

- sand (39% of total survey area)
- sandy shell hash (27%)
- coarse gravel with sand and shell (22%)
- bryozoan-sponge reefs (7%)
- bushy-bryozoan thickets (4%)
- patchy low-relief bryozoan-sponge habitat (1%).

The Hananui proposal area itself comprised:

- sand (66%)
- sandy shell hash (16%)
- coarse gravel with shell and sand (18%).

Habitat modelling predicted trace amounts of biogenic habitat within the proposal area (< 0.1%); however, these were assessed as modelling artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance – essentially artefactual flecks within a sand-dominated substrate and not representative of true biogenic habitat. Verified biogenic habitats (bryozoan-sponge reefs, bushy-bryozoan thickets, patchy low-relief bryozoan-sponge habitat) were absent from the proposal area but occur nearby. These habitats support high biodiversity, including potentially sensitive taxa such as frame-building bryozoans, sponges, tube worms, brachiopods and large bivalves.

Assessment of effects

Deposition of organic material from feed and faeces is the primary ecological concern. In dispersive, coarse-sediment environments, organic matter is expected to be readily resuspended and widely dispersed, producing a broader but less intense footprint than at sheltered inshore sites.

Two depositional metrics were used: one to assess deposition without resuspension (solids flux, kg or g solids $m^{-2}{\cdot}yr^{-1}$), and one incorporating resuspension (residual solids, $g{\cdot}m^{-2}$). Residual solids provide a more realistic metric for high-flow sites such as Hananui. Solids flux is useful for determining a zone of maximum effect (ZME).

Model outputs were averaged over the three months of peak feed discharge at each farm block. Under the proposed single year-class regime, peak inputs will be followed by periods of low to no discharge, with sites fallowed for ≥ 3 months between cycles. Thresholds developed for Marlborough Sounds farms were used to relate model outputs to levels of ecological effect, expressed as depositional zones:

- *Zone of maximum effect* (> 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ flux): up to 19 ha of sandy seabed habitats (1.5% of the proposal area) may receive high enrichment beneath pens during periods of peak production, with 145 ha of moderate enrichment expected in the rest of the ZME.
- *Primary footprint* (≥ 7 $g \cdot m^{-2}$ residual solids): moderate enrichment is predicted across 473–533 ha (37–42% of the proposal area) of sandy seabed habitats during periods of peak production. The primary deposition footprints are largely predicted to remain within the Hananui proposal area.

Outside of the primary footprints, low levels of farm-related organic waste may disperse more than 17 km from the proposal area boundary. Within this broader dispersal zone, a conservative outer limit of effects (OLE) footprint (> 0.7 to < 7 $g \cdot m^{-2}$ residual solids, an order of magnitude less than the primary footprint) indicates the maximum spatial extent at which farm-related effects might be detectable. Within this footprint, minor enrichment is possible.

During periods of peak production, the OLE encompasses up to a further 1,262 ha of seabed per farm block at each stage. The OLE footprint is mostly constrained to sandy habitats with varying amounts of shell hash and gravel. At Stage 1, the outer edge of this footprint (where residual solids are ~ 0.7 $g \cdot m^{-2}$) overlaps slightly with up to 8.4 ha (~ 1.4% of the total mapped biogenic habitat). At Stage 2, parts of the lower end of this footprint (i.e. where residual solids are ~ 2 $g \cdot m^{-2}$ or less) is predicted to overlap with up to 43 ha of the mapped bryozoan-sponge reef, 25 ha of the patchy low-relief bryozoan-sponge habitat, and 16 ha of the bushy-bryozoan thickets (up to 5%, 17% and 3% of the total mapped area per habitat type, respectively).

Outside of the OLE, farm-derived waste may be detectable but adverse ecological effects are unlikely.

Beneath pens, sediments are expected to show elevated organic content, reduced redox potential and dominance by opportunistic macrofauna. Under the single-year-class farming regime proposed by Ngāi Tahu Seafood, these periods of very high deposition will be followed by periods of low to no deposition, during which a degree of seabed recovery would be expected.

Across the primary footprint, moderate enrichment may reduce diversity but increase abundance, while beyond this, effects are expected to be minor or negligible.

Epibiota densities are low within predicted ZME and primary footprints, but small areas of biogenic habitat at the outer extent of the OLE may experience slight enrichment that could enhance food supply for suspension feeders. While the exact nature of effects from low levels of deposition on these ecologically valuable and sensitive seabed habitats is uncertain, the level of residual solids predicted to reach these habitats is significantly lower (at least an order of magnitude lower at Stage 1) than the 9 $g \cdot m^{-2}$ threshold at which no effects have been observed on rocky reefs in Tory Channel / Kura Te Au. This suggests that biogenic habitats outside of the proposal area are unlikely to be exposed to deposition levels that would cause stress. Nevertheless, monitoring of these communities is crucial to ensure adverse effects do not occur.

Other potential effects include seabed disturbance from mooring installation (habitat displacement, sediment resuspension) and effects associated with farm structures (biofouling drop-off, shading). These can be minimised through best-practice management, with residual effects expected to be minor or less.

Management approach

Ngāi Tahu Seafood proposes a staged development, beginning with Stage 1 to enable monitoring of effects, before progressing to Stage 2. Management will follow an effects-based framework, with clear environmental thresholds and adaptive management to allow responsive operational adjustments (e.g. reducing farming intensity where required), particularly if monitoring indicates potential far-field effects on sensitive biogenic habitats. Mitigation measures include fallowing (3 months per production cycle), avoidance of high-value habitats through site selection and buffer zones.

Conclusions

Our assessment indicates that seabed effects are expected to be localised and manageable, with no overlap between primary depositional footprints and mapped biogenic habitats. While biogenic habitats at the OLE edge may experience minor enrichment, predicted deposition is at least an order of magnitude below levels predicted on rocky reefs in the Marlborough Sounds where no effects have been detected, making adverse impacts unlikely. Monitoring will ensure that any unexpected changes are detected and addressed.

Residual uncertainty remains regarding tolerance thresholds for sensitive taxa at low deposition levels and for coarse sandy habitats in dispersive environments, where empirical data are limited. These uncertainties have been accounted for through conservative modelling and assumptions, combined with a relatively precautionary development pathway involving staged expansion, staggered stocking and fallowing. A robust monitoring framework with predefined triggers and adaptive management will provide further assurance.

Overall, the proposed development is not expected to result in adverse seabed effects, and seabed ecological integrity is likely to be maintained.

1. Introduction

The Ngāi Tahu Seafood Resources (Ngāi Tahu Seafood) Hananui Aquaculture Project is a two-stage salmon farming development proposed for an exposed coastal marine area of ~ 1,285 ha (the Hananui proposal area; Figure 1), located 2–6 km off the northern coast of Stewart Island / Rakiura (hereafter Rakiura).

This report describes the seabed environment within and surrounding the Hananui proposal area and the wider (~ 12,500 ha) 'survey area' (see Figure 1), assesses potential environmental effects on benthic habitats from the proposed operation, and outlines recommended mitigation measures. It also summarises updates since the 2023 COVID-19 Fast-track submission, including refinements to the seabed assessment (e.g. updated depositional modelling, revised habitat mapping) and changes to the farm design and operation (e.g. improved siting) that are expected to reduce potential seabed effects.

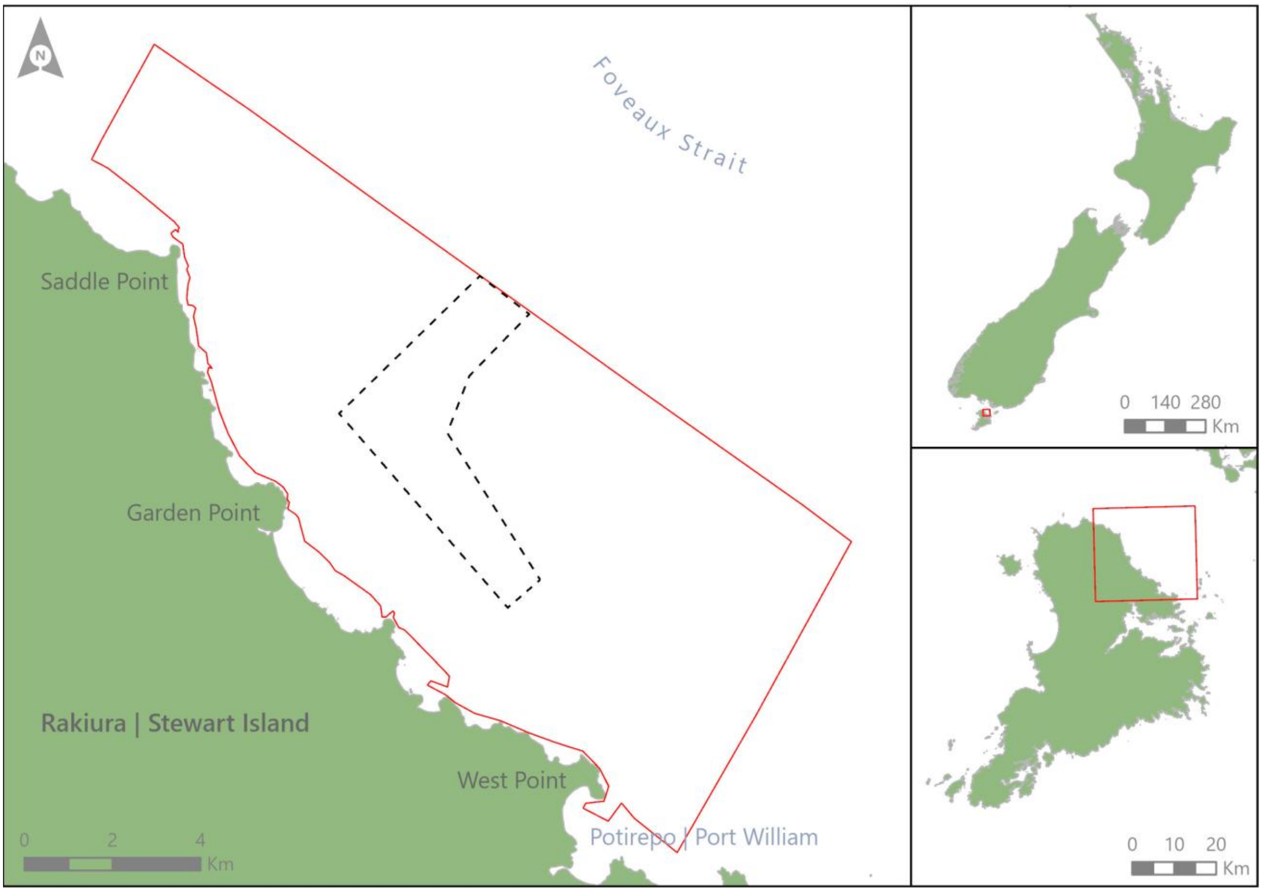

Figure 1. The Ngāi Tahu Seafood Hananui 'proposal area' (black dashed line) and the broader 'survey area' (red polygon, left panel), which includes data collected under Ngāi Tahu Seafood contracts (2018–25), as well as information from preliminary scoping investigations commissioned by the Ministry for Primary Industries (MPI) in 2018 and a seabed mapping exercise survey conducted as part of a wider Land Information New Zealand (LINZ) survey in early 2022.

2. Summary of existing data on seabed characteristics and benthic communities around northern Rakiura

The seabed off northern Rakiura is strongly influenced by the energetic tidal currents of Foveaux Strait. Water enters the strait from the west and southeast, flowing northeast past Ruapuke Island (Cranfield et al. 2003). These strong tidal flows, combined with frequent Southern Ocean swells, create a dynamic seafloor environment characterised by active sediment transport and high seabed disturbance (Cranfield et al. 2003). The interaction of these forces drives sediment movement and plays a major role in shaping seabed structure and benthic habitats (Anderson et al. 2019).

Beneath the seabed lies relict pebble gravel deposited during glacial low sea levels when the strait was exposed (Cullen et al. 1967). This gravel is commonly overlain by patches of coarse sand and shell debris, particularly in areas of lower tidal energy. Large fan-shaped subtidal dunes, some exceeding 6 m in height and extending several kilometres, are present northeast of Garden Point. These dunes have formed where converging currents have facilitated the accumulation of substantial volumes of biogenic sediment (Cranfield et al. 2003).

Historically, this region supported extensive biogenic reefs, locally known as 'mullock', primarily composed of the bryozoan *Cinctipora elegans* and associated invertebrates such as ascidians, sponges, polychaetes and large bivalves, including oysters and mussels (Cranfield et al. 1999). These reefs, aligned with tidal currents, extended from hundreds of metres to kilometres. They provide essential habitat for ecologically and commercially important species, including dredge oysters / tio (*Ostrea chilensis*) and blue cod / rāwaru (*Parapercis colias*) (Cranfield et al. 2001; Beentjes et al. 2019).

Oyster dredging began in 1867 and historically targeted these complex reef habitats (Cranfield et al. 2003). Over more than a century, commercial dredging removed reef frameworks and released large quantities of biogenic sediment. This sediment was then transported by currents and deposited downstream. The extensive subtidal dune field northeast of Garden Point is thought to have formed directly through this process, with sediments composed almost entirely of fragmented bryozoans and molluscs.

Today, bryozoan reefs in Foveaux Strait persist but are fragmented and substantially reduced in extent relative to their historical coverage. Remaining patches tend to occur in areas with lower physical disturbance, such as sheltered embayments or deeper zones less affected by trawling and dredging (Anderson et al. 2019). These surviving reefs appear as dense, encrusting bryozoan mats and low-profile reef structures, often interspersed with associated fauna such as sponges and ascidians. Despite their reduced size and degraded condition, these remnants continue to provide important habitat complexity and refuge for benthic species.

The removal of reef structures and associated sediment redistribution has had lasting ecological consequences, including widespread biogenic habitat loss and declines in associated benthic species.

Decades of oyster dredging have also contributed to long-term declines in dredge oyster populations, driven by sustained fishing pressure and exacerbated by periodic mass mortality events caused by the parasite *Bonamia exitiosa* (Michael et al. 2015). Meanwhile, the degradation of reef habitat has reduced structural complexity, affecting other reef-associated species. For example, blue cod, which are strongly associated with mullock reefs, tend to occur at lower densities in areas where these habitats have been lost or degraded (Cranfield et al. 2001; Carbines et al. 2004; Beentjes et al. 2019).

In addition to bryozoan reefs, Foveaux Strait and northern Rakiura support a variety of other biogenic habitats, including patches of sponges, large bivalve beds, tube worm aggregations and sea tulips (Jones et al. 2016). Shallow rocky reefs around Rakiura maintain dense macroalgal communities dominated by kelp and fucoid species, which function as important nursery and feeding grounds for fish and invertebrates. Rock-wall habitats in Foveaux Strait exhibit a high diversity of encrusting invertebrates, including sponges, bryozoans, ascidians and hydroids, which colonise hard substrates and significantly enhance habitat complexity (Kettles et al. 2017).

Collectively, these features illustrate a spatially heterogeneous seabed shaped by both natural physical processes and historical human disturbance.

3. Characterisation of the proposal area

3.1 Seabed characterisation methods

Seabed characterisation for the proposal area drew on a combination of acoustic mapping, video surveys and sediment sampling conducted between 2018 and 2025. Initial surveys took place from October 2018 to July 2019, and were supplemented with earlier investigations in the wider survey area (Bennett et al. 2018; Taylor and Jary 2018). These efforts included side-scan sonar, drop-camera video surveys and sediment sampling using a van Veen grab sampler (sampling locations provided in Figure 2).

In early 2022, Discovery Marine Limited (DML) undertook a multibeam echosounder (MBES) survey covering ~ 12,500 ha in the area as part of a wider Land Information New Zealand (LINZ) programme. The survey provided high-resolution bathymetry and acoustic backscatter data, which supersede earlier side-scan sonar efforts (see Bennett et al. 2022) by offering more detailed and spatially accurate seabed information. The side-scan results are therefore not presented here, although drop-camera video footage collected during those earlier surveys has been used to support ground-truthing of the MBES data. These video records, together with additional imagery collected during subsequent drop-camera and remotely operated vehicle (ROV) surveys in 2022 and 2025, are included in this report.

Sediment sampling and macrofaunal analysis (October 2018 and January 2019[1])

Sediment samples were collected from 31 sites throughout the proposal area using a van Veen grab sampler (Figure 2). Cores were assessed for physical properties (colour, odour, texture) and redox potential (Eh_{NHE}). Sub-samples from the upper 3 cm were analysed for organic content (percentage ash-free dry weight, %AFDW) and particle size distribution using a seven-fraction grain size analysis (brief method descriptions for the physical and chemical analyses are provided in Appendix 1).

Macrofaunal communities were characterised using separate 10 cm-deep sediment cores, which were sieved at 0.5 mm, preserved, and identified by taxonomic experts. Community metrics, such as total abundance, taxa richness and biotic indices, were calculated (Appendix 2). Multivariate analyses (nMDS, cluster analysis and SIMPER) were conducted using PRIMER v7 to identify spatial patterns and key taxa contributing to community structure.

Multibeam echosounder (MBES) survey (January and February 2022)

The MBES survey, conducted by DML between January and February 2022, generated high-resolution bathymetry, backscatter imagery, sediment data and water column profiles for the wider survey area

[1] More recent sediment samples were collected during the 2022 MBES survey (see below). Together with MBES and video data, these samples confirm that broad sediment patterns remain consistent across the proposal area, providing sufficient confidence that additional sediment sampling was not required for this application.

(~ 12,500 ha). These data were supplied to Cawthron Institute (Cawthron) for habitat classification and mapping.

Ground-truthing using seabed imagery from the video surveys (see below) was integrated with the MBES data to develop a predictive habitat model (see Appendix 3 for details). This model supported the creation of a detailed, spatially extensive seabed habitat map.

Video surveys and biological ground-truthing (March 2018 to April 2025)

Drop-camera survey methods

High-definition drop-camera video surveys were carried out across six surveys between 2018 and 2019 (March, June, October and December 2018; January and July 2019) at sites selected based on early sonar imagery. Additional surveys targeting sites selected based on the MBES data and areas of uncertainty within the MBES-based habitat model were conducted in March 2023 and April 2025. At each site, a camera with integrated lights and surface-fed video was lowered until the seabed was in view. At least 1 minute of seabed footage was recorded as the vessel drifted with the current.

Imagery from the 2018–19 surveys was geolocated via the boat's GPS, while surveys in 2023 and 2025 used an ultra-short baseline (USBL) positioning system, allowing for more precise spatial tracking of the camera independent of vessel movement. All footage was analysed for habitat classification and identification of visible epifauna.

Remotely operated vehicle (ROV) survey methods

A series of ROV surveys were conducted in October 2021, July and September 2022, and April 2023. This enabled targeted exploration of transitional habitats and offered greater control and coverage. The ROVs were flown along the seabed, either along pre-defined transects or in exploratory patterns, depending on the survey objective. As with the drop-camera surveys, a USBL system was used to georeference footage independently of the vessel.

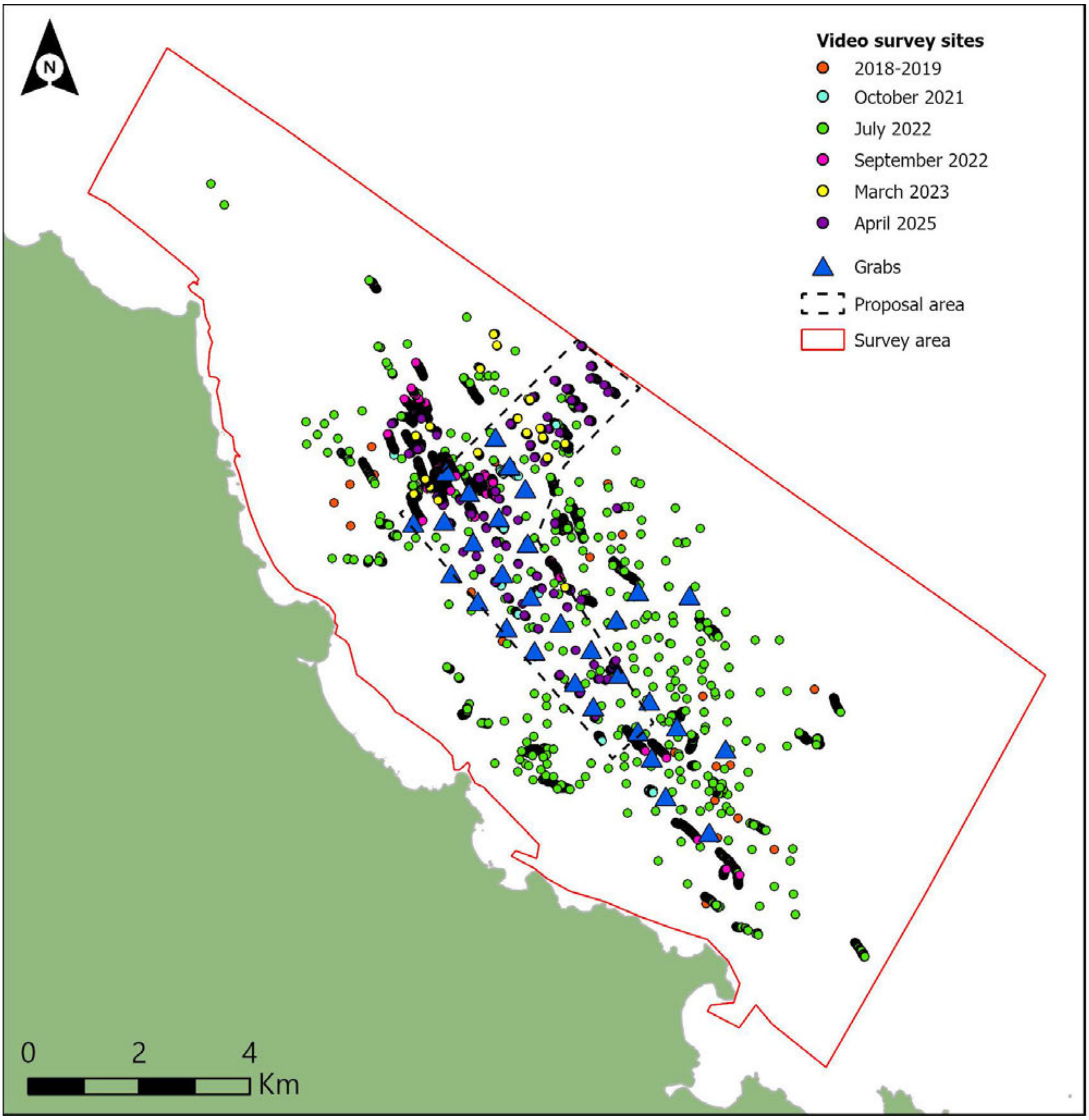

Figure 2. Locations of sediment grab sampling stations and video survey sites used to characterise seabed habitats and biological communities within and around the Hananui proposal area (dashed black outline). The wider multibeam echosounder (MBES) survey area is shown in red. Blue triangles = sediment grab locations; orange dots = video sampling locations. Note that some video surveys were carried out along a transect, and as such, sites overlap and appear as a continuous black line.

3.2 Seabed characterisation results

Bathymetry

High-resolution MBES data provide detailed bathymetric information across both the proposal area and wider survey area (Figure 3). This dataset underpins all subsequent mapping and seabed characterisation presented in this report.

Water depths across the wider survey area range from ~ 5–10 m nearshore to ~ 40 m offshore. The seabed generally slopes gradually away from the coast, with depth contours running broadly parallel to the shoreline. A prominent geomorphic feature of the area is the extensive system of sandbanks and sand wave fields extending northeast from between Port William / Potirepo (hereafter Port William) and Garden Point to the outer survey boundary. Additional notable features include Newton Rock, as well as several smaller isolated rocky outcrops located up to 2 km offshore between Port William and Garden Point (Figure 3), and a broad depression northeast of Saddle Point reaching a maximum depth of 57 m.

Within the 1,285 ha proposal area (delineated by the dashed black line in Figure 3), depths range from 20–25 m inshore to ~ 40 m at the offshore boundary. The seabed here forms part of the broader sandbank system and is characterised by mobile sandy habitats with elongate ridges, sand wave fields and shallow channel-like depressions. Rocky outcrops are largely absent from the proposal area and the seabed appears to be primarily shaped by sediment mobility and tidal currents.

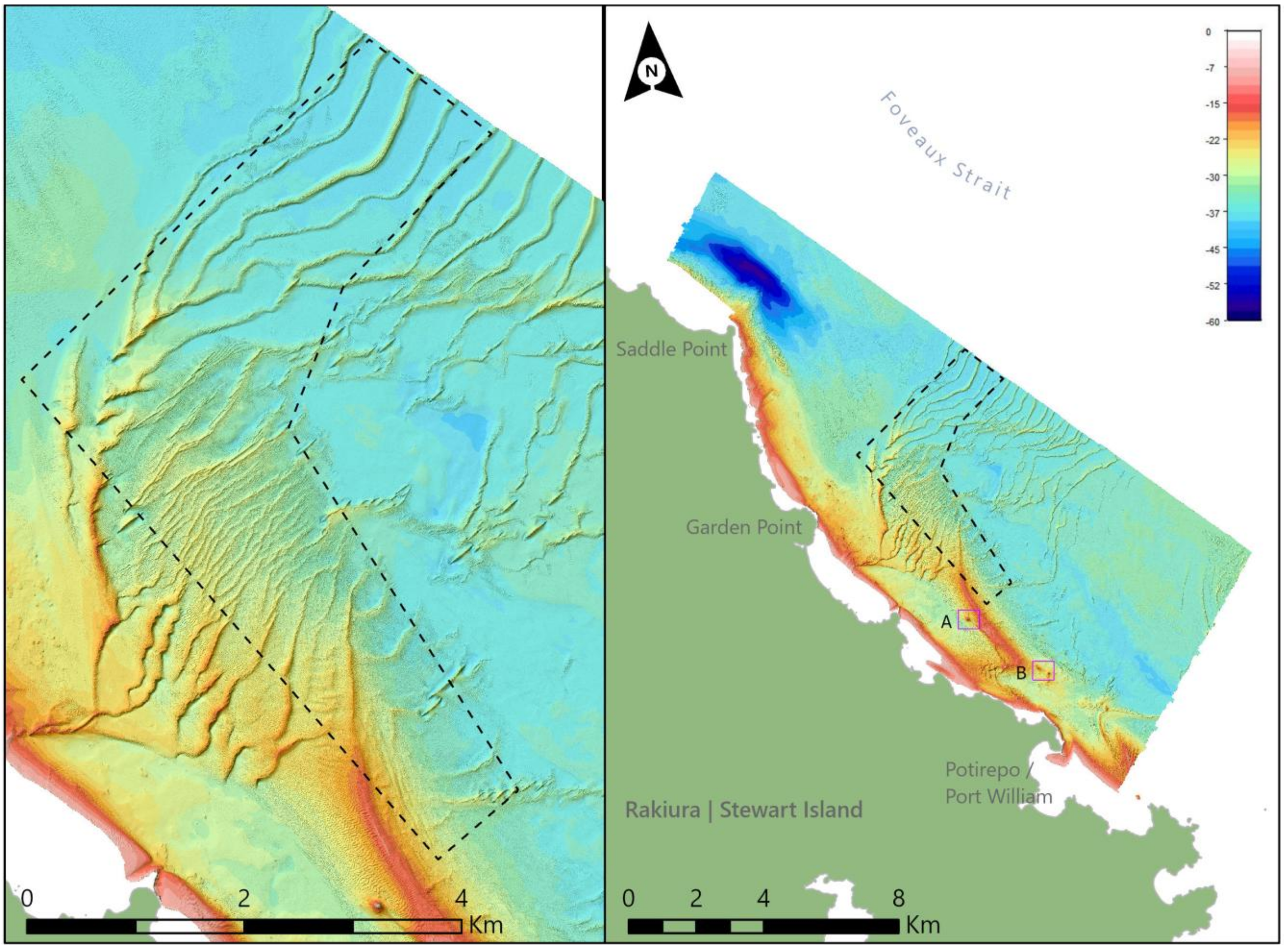

Figure 3. Multibeam echosounder (MBES) bathymetry of the Hananui proposal area (dashed black outline) and surrounding seafloor. Depths range from ~ 5–10 m nearshore to ~ 40 m offshore. The seabed is dominated by mobile sandy habitats with elongate ridges, sand wave fields and shallow channels. Prominent features include sandbanks, isolated rock outcrops up to 2 km offshore, and a broad depression northeast of Saddle Point (maximum depth 57 m). In the map extent on the right, detail A shows Newton Rock and detail B shows the smaller rock outcrops surveyed by video.

Sediment physical and chemical properties

Multibeam echo sounder (MBES) backscatter data provide insight into the physical characteristics of the seafloor across the proposal area and wider survey area. Variations in backscatter intensity reflect differences in substrate hardness and surface texture, allowing broadscale interpretation of seabed types. In the backscatter mosaic (Figure 4), darker areas indicate lower acoustic reflectivity, typically associated with softer substrates such as fine sand, silt, mud or areas with higher organic content (DML 2022). Conversely, brighter areas correspond to stronger acoustic returns, generally indicative of harder seabed types such as compacted sand, shell hash, gravel or exposed rock.

To support interpretation of the acoustic data, qualitative sediment observations were made at selected stations during the MBES survey (marked as green dots in Figure 4). Sediment type was visually assessed from material collected with a grab sampler. In contrast, the sediment composition pie charts in Figure 4 are based on a separate set of samples from the earlier seabed characterisation surveys. These underwent quantitative particle size analysis, providing detailed sediment composition data. Results from this earlier sampling confirm that sand is the dominant sediment type, ranging from 25% to 97% of total composition. Gravel content was highly variable, from less than 0.1% to 72%, while mud generally represented a minor fraction, between 3% and 9%. Generally, the seabed is dominated by well-sorted medium to fine sands, with variable amounts of shell hash and occasional gravel. This sediment pattern is consistent with a dynamic shelf environment shaped by wave and current action, which tends to remove finer particles and leave behind coarser material. Although grab samples were not collected from the northeastern part of the proposal area, bathymetry, backscatter and video imagery indicate that this area shares similar depth and substrate characteristics (e.g. coarse gravel, sand and shell hash) with nearby sampled sites such as grab stations 16, 29 and 20 (Figure 4).[2]

Chemical analysis of the earlier sediment samples showed low to moderate levels of organic material (0.8–3.1% AFDW). Redox potential measurements (average Eh_{NHE} ~ 382 mV) indicate well-oxygenated conditions, consistent with a well-flushed, non-enriched benthic environment.

Full results of sediment grain size fraction and sediment analyses are provided in Appendix 4.

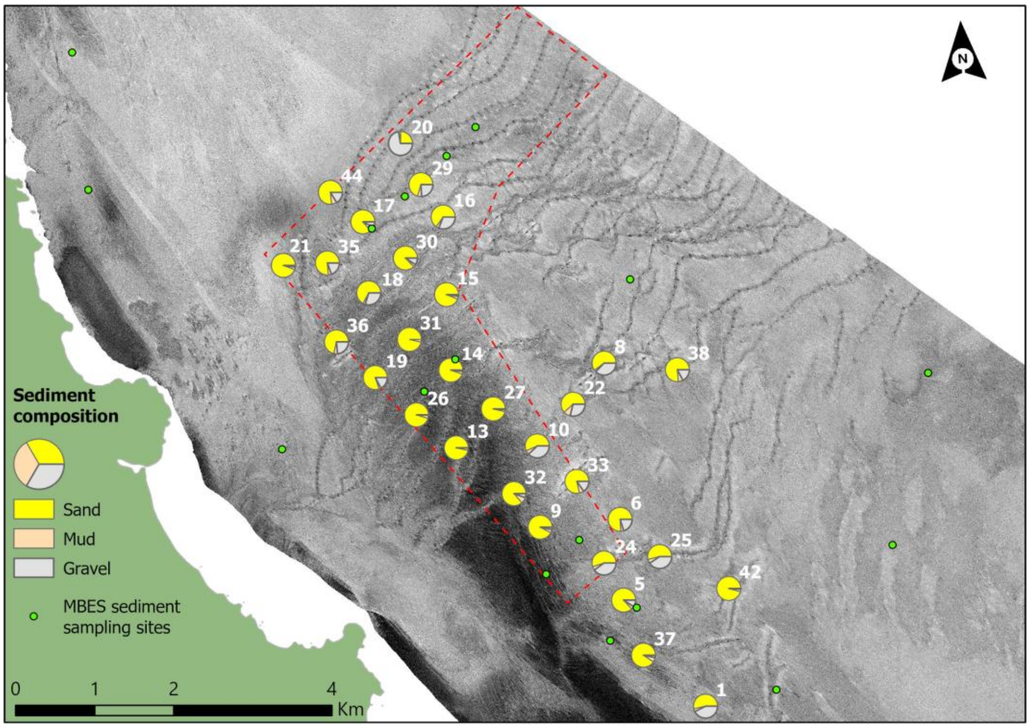

Figure 4. Sediment composition at 31 grab sampling sites within and surrounding the Hananui proposal area (outlined in red), overlaid on the multibeam echosounder (MBES) backscatter mosaic. Locations of seabed sediment samples collected during the MBES survey for ground-truthing acoustic data are also shown (green dots). Numbers relate to grab sample IDs in Appendix 4.

[2] Targeted sediment sampling for this area is recommended as part of the baseline monitoring programme.

Seabed habitats and communities

Six primary seabed habitat types were identified across the wider survey area based on MBES data and ground-truthing from drop-camera video and ROV surveys: sand; sandy shell hash; coarse gravel with shell and sand; bushy-bryozoan thickets; bryozoan-sponge reefs; and patchy low-relief bryozoan-sponge habitat.[3] These classifications were used in conjunction with the predictive habitat model to estimate the spatial extent of each habitat type and to produce a habitat map of the survey area (Figure 5). Representative images of seabed habitat types are provided in Figure 6 and a full summary of conspicuous biota observed within each habitat type is provided in Table 1.

Within the proposal area, three primary seabed habitat types were recorded:

- **Sand** – Covering ~ 66% of the proposal area, this habitat is characterised by sand ripples, waves and large sandbanks. Some shell hash was present, mainly in the troughs of sand ripples or waves. Epifaunal assemblages were very sparse, with brittle stars observed occasionally and large bivalves very rarely. Tufts of bushy bryozoans or small patches of encrusting sponges and bryozoans were very occasionally observed on isolated shell debris.
- **Sandy shell hash** – Comprising around 16% of the proposal area, this habitat consists of sandy terrain with moderate to high amounts of shell hash. The surface typically exhibits fewer sand ripples and wave forms than the primarily sand habitat, suggesting a more consolidated substrate. Notably, this sediment type forms the peaks of elongated subtidal ridges in the northeast portion of the proposal area. Tufts of bushy bryozoans and small patches of encrusting sponges and bryozoans were occasionally observed on shell debris. Epifaunal abundance and diversity was marginally higher than in pure sand habitats, likely reflecting the increased structural complexity provided by the shell hash.
- **Coarse gravel with shell and sand** – Present across ~ 18% of the proposal area, this habitat is more consolidated than the primarily sand habitats and features a mix of coarse sediments, including gravel, shell debris and cobbles, with very occasional isolated patches of biogenic structure such as bushy and erect bryozoans and sponges. It forms the troughs of large, elongated subtidal ridges in the northeast section of the proposal area.

The habitat model predicted very small proportions of biogenic habitat types within the proposal area; specifically, bryozoan-sponge reef (0.094%), bushy-bryozoan thickets (0.004%) and patchy low-relief bryozoan-sponge habitat (0.014%). These are seen as small flecks within predominantly sand habitat (Figure 5) and are considered modelling artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance. Multibeam backscatter and bathymetry show sand ripple formations in these areas, and previous investigations confirmed that similar features were sand-dominated and did not represent true biogenic habitat. Based on this validation and the consistency of surrounding substrate characteristics, we assume the remaining flecks are of the same nature and confirm that no verified biogenic habitat occurs within the proposal area. Across the wider survey area, the six primary seabed habitat types were:

- **Sand** – 39% of the total survey area (as described above).
- **Sandy shell hash** – 27% of the total survey area (as above).

[3] The habitat map has been updated since the previous assessment (Bennett et al. 2022) with additional video observation points and includes a new category of habitat, patchy low-relief bryozoan-sponge habitat.

- **Coarse gravel with shell and sand** – 22% of the total survey area (as above).
- **Bushy-bryozoan thickets** – 4% of the total survey area, these thickets, primarily composed of *Orthoscuticella innominata* and associated calcareous tube worms, support moderate to high epifaunal diversity, including sponges, brachiopods, large bivalves, and fish species such as blue cod and tarakihi (*Nemadactylus macropterus*).
- **Bryozoan-sponge reefs** – 7% of the total survey area, these structurally complex habitats were formed by reef-building bryozoans (e.g. *Celleporaria agglutinans*, *Cinctipora elegans*) and sponges (e.g. *Dactylia varia*) and host a wide range of associated fauna.
- **Patchy low-relief bryozoan-sponge habitat** – 1% of the total survey area. This habitat consists of sparse, low-profile assemblages of sponges and small frame-building bryozoans, interspersed with areas of coarse gravel with shell and sand, and supports low densities of visible benthic fauna. Compared to more structurally complex bryozoan-sponge reefs, these low-relief habitats exhibit markedly lower biodiversity and habitat complexity. Cranfield et al. (2004) proposed a model of five successional stages for benthic habitat regeneration in Foveaux Strait following oyster dredging, ranging from early colonisation by opportunistic species to the development of mature, biodiverse reef structures. The patchy low-relief bryozoan-sponge habitat observed here likely represents an early successional stage, whereas the bryozoan-sponge reefs described above correspond to later successional stages (stages 4–5 in Cranfield et al. [2004]), characterised by high structural complexity and diverse assemblages.

Rock outcrops

The MBES and ROV surveys also identified several small rock outcrops outside of the boundaries of the proposal area (Figure 3). The outcrops range from low-relief rock with some sponges, kina (*Evechinus chloroticus*), sea cucumber (*Australostichopus mollis*) and brittle stars (mostly *Ophiopsammus maculatus*), to more biodiverse outcrops, 1–2 m wide and heavily encrusted with epifauna, including sponges, bryozoans and ascidians.

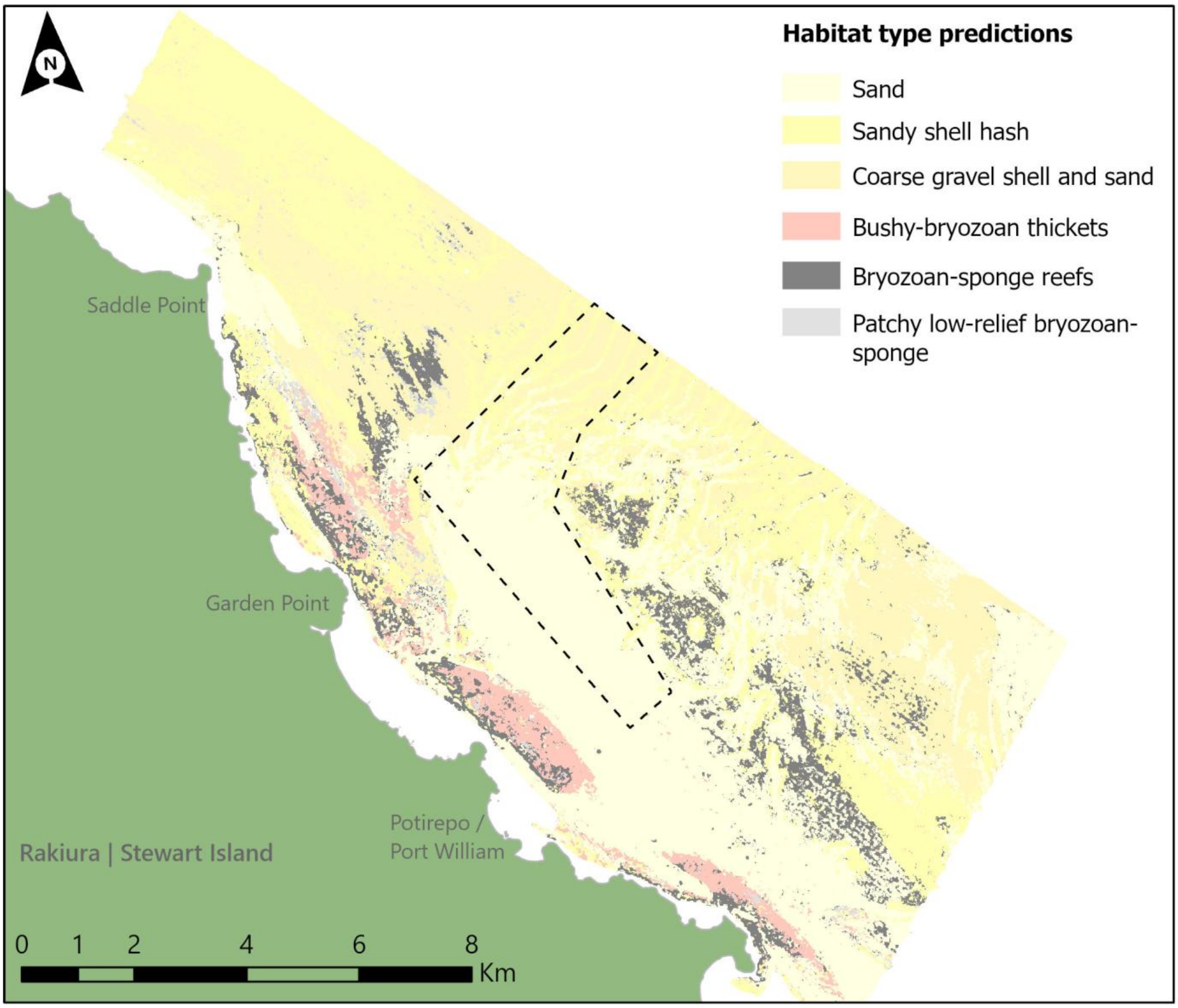

Figure 5. Habitat map showing the predicted extent of habitat based on the multibeam echosounder (MBES), sediment sampling and video validation surveys. Sediment sampling was also conducted to ground-truth the MBES data, although it was not directly used in the habitat prediction model. Black outline = the Hananui proposal area.

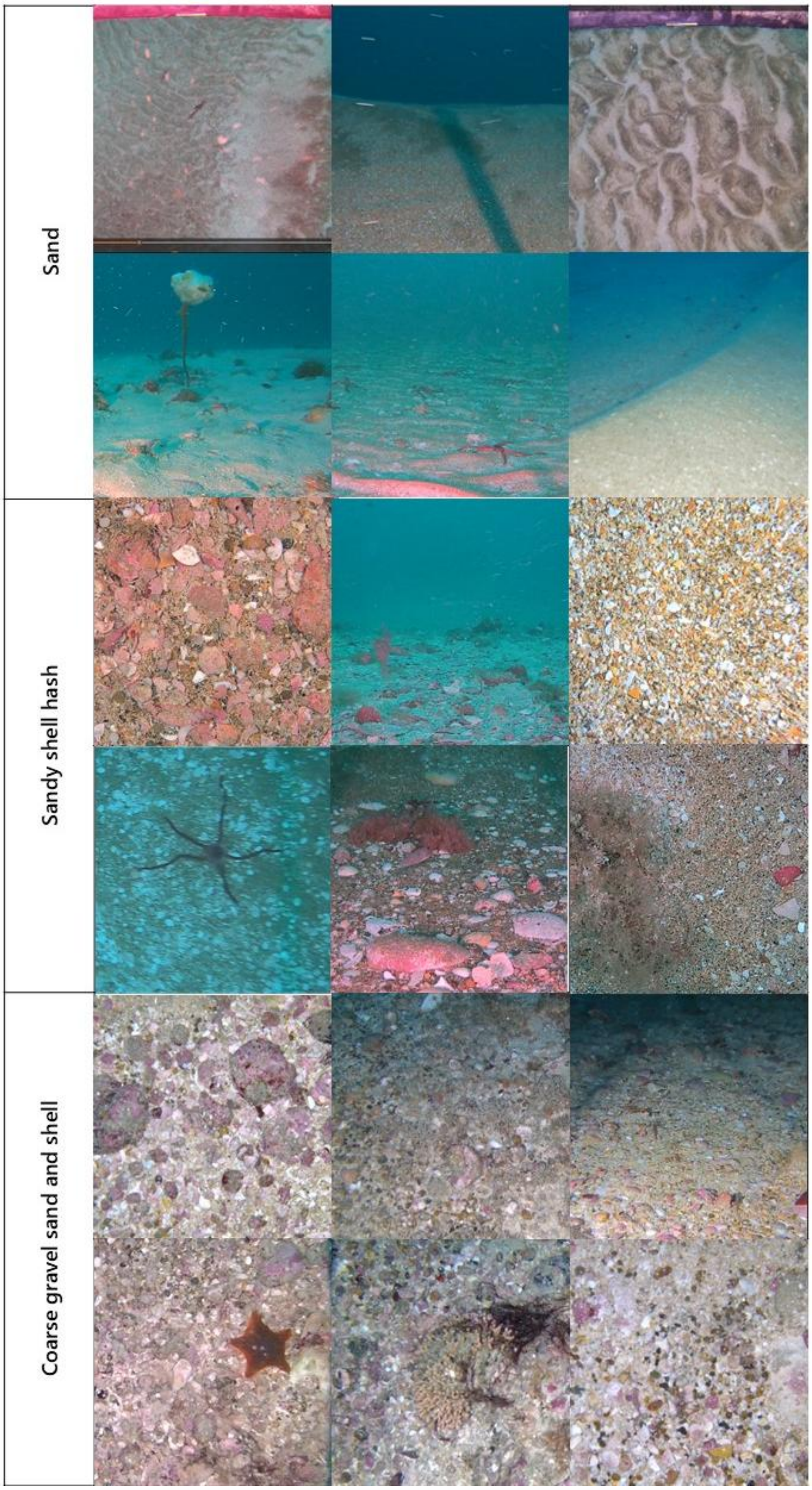

Figure 6. Representative images of seabed habitats observed within the Hananui proposal area and wider survey area, illustrating the range of substrate types and biogenic communities, as well as differences in structural complexity and biodiversity across habitat types.

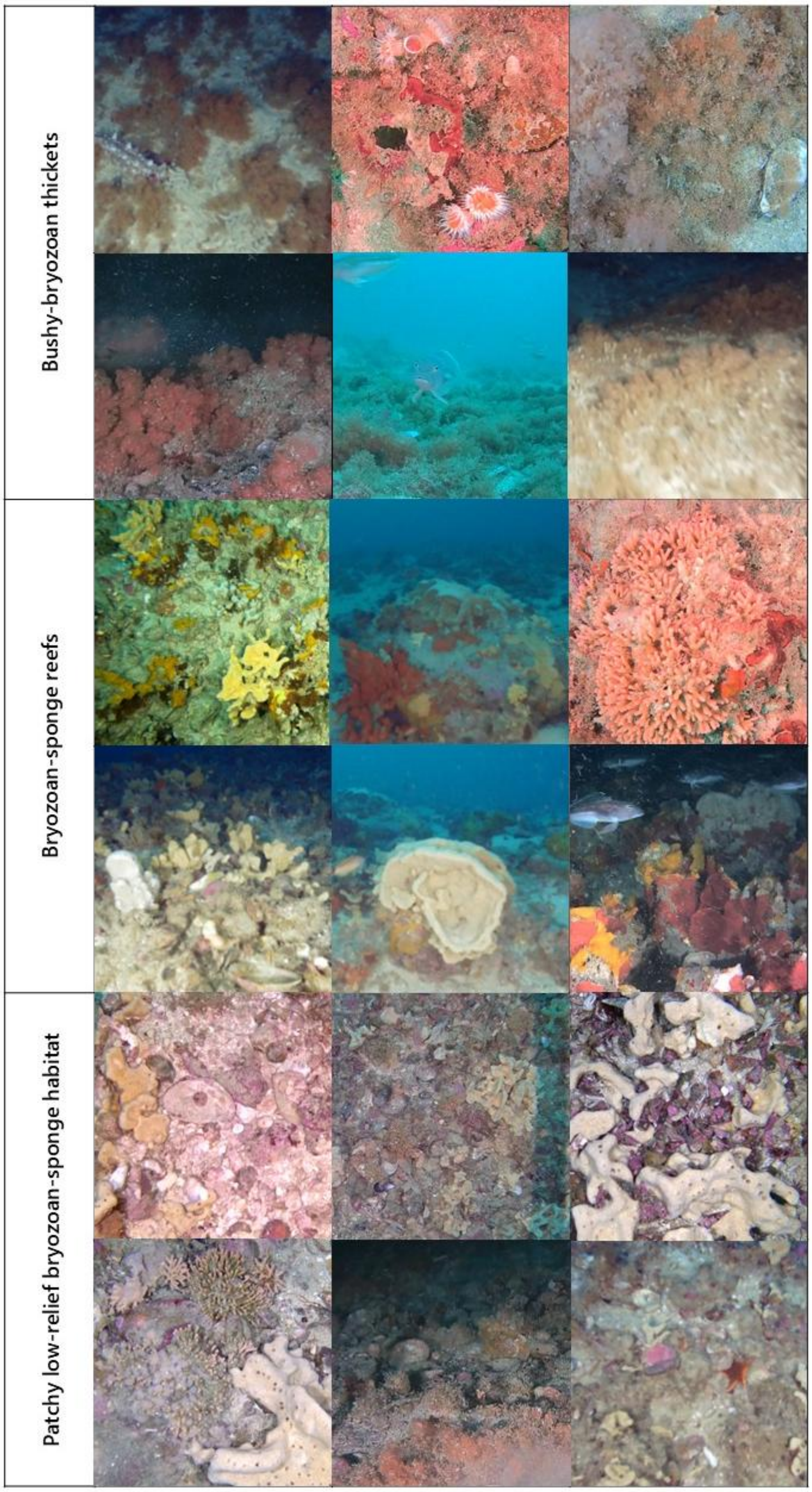

Figure 6 (cont.). Representative images of seabed habitats observed within the Hananui proposal area and wider survey area, illustrating the range of substrate types and biogenic communities, as well as differences in structural complexity and biodiversity across habitat types.

Table 1. Conspicuous biota observed within each habitat type across the Hananui proposal area and wider survey area.

Habitat type	Conspicuous benthic biota	Fish species
Sand	Brittle stars (mostly *Ophiopsammus maculatus*). Some red and brown drift algae. Occasional tufts of bushy bryozoans and feather hydroids. Very occasional isolated encrusting bryozoans, sponges, ascidians (e.g. sea tulip, *Pyura pachydermatina*), sea cucumbers (*Australostichopus mollis*), 11-armed sea stars (*Coscinasterias muricata*) and cushion stars (*Patiriella regularis*). Very occasional gastropods, including saw shells (*Astraea heliotropium*) and turret shells (*Maoricolpus roseus*); brachiopods (likely *Neothyris lenticularis* or *Magasella sanguinea*); and large bivalve species including the dredge oyster (*Ostrea chilensis*), scallops (*Pecten novaezelandiae*), dog cockles (likely *Tucetona laticostata*), bearded horse mussel (*Modiolus areolatus*), tuatua (*Paphies subtriangulata*) and *Oxyperas elongatum*.	Blue cod (*Parapercis colias*), leather jacket (*Meuschenia scaber*), triplefin
Sandy shell hash	As for sand, but with a slightly higher abundance of brittle stars observed.	Blue cod, leather jacket, triplefin
Coarse gravel with shell and sand	As for sandy shell hash, but with very occasional small erect bryozoans (mainly *Cinctipora elegans*) and sponges.	Blue cod, leather jacket, triplefin
Bushy-bryozoan thickets	Areas of abundant bushy bryozoans (*Orthoscuticella innominata*). Calcareous tube worms (likely *Galeolaria hystrix*), often in mounds. In places, a moderate abundance of small erect bryozoans (mainly *C. elegans*) and sponges. A moderate number of feather hydroids. Sparse foliose red algae. Some red and brown drift algae. Occasional individual or small patches of large bivalves, including dredge oysters, scallops, bearded horse mussels and dog cockles. Occasional patches of brachiopods. Occasional kina (*Evechinus chloroticus*), sea tulips, sea cucumbers. Gastropods, including saw shells. Occasional brittle stars, cushion stars and 11-armed sea stars.	Blue cod, leather jacket, tarakihi (*Nemadactylus macropterus*)

Habitat type	Conspicuous benthic biota	Fish species
Bryozoan-sponge reefs	Biogenic clumps formed by the massive encrusting bryozoan *Celleporaria agglutinans* and erect bryozoan *C. elegans*. Several other erect and encrusting species of bryozoan, including fragile lacy forms such as *Hornera foliacea* and possibly *Hornera robusta*. Abundant encrusting and erect sponges (including mainly likely *Dactylia varia* and *Crella incrustans* and possibly *Iophon minor* and *Chondropsis* cf. *kirkii*). Colonial ascidians (including *Botrylloides* sp., *Botryllus* sp. and *Eudistoma circumvallatum*). Calcareous tube worms (as above, likely *G. hystrix*). Moderate abundance of bushy-bryozoan and feather hydroids. Sparse foliose and encrusting red algae. Occasional individual or small patches of large bivalves, including dredge oysters (sparse to patchy), scallops (sparse), bearded horse mussels (sparse) and dog cockles (sparse to abundant). Occasional patches of brachiopods. Patches of sea anemones (*Anthothoe albocincta*) in some areas. Occasional kina, sea tulips, sea cucumbers, brittle stars, cushion stars, 11-armed sea stars and gastropods, including saw shells.	Blue cod, leather jacket, tarakihi
Patchy low-relief bryozoan-sponge habitat	Low-profile biogenic clumps of encrusting / erect bryozoans (*C. agglutinans*, *C. elegans*) and erect / encrusting sponges (including *D. varia, C. incrustans*), interspersed with areas of coarse gravel with shell and sand. Cushion stars were consistently observed throughout the area. Other fauna, including kina, sea cucumbers, brittle stars, 11-armed sea stars and gastropods such as saw shells, were occasionally recorded in association with biogenic clumps.	Blue cod, leather jacket, tarakihi

Ecologically valuable and sensitive seabed habitats

No taxa classified as Threatened or At Risk in the New Zealand Threat Classification System (NZTCS; Freeman et al. 2013), or as Threatened in the International Union for Conservation of Nature (IUCN) Red List were observed in the proposal area or wider survey area.[4] However, several habitat types within the wider survey area meet definitions of sensitive marine habitats under Aotearoa New Zealand guidance (MacDiarmid et al. 2013; Anderson et al. 2019). These habitats are sustained by structure-forming and functionally important organisms such as reef-building bryozoans, sponges and calcareous tube worms. Many of these taxa are slow-growing or physically fragile, making them particularly vulnerable to disturbance from processes such as sedimentation, smothering and direct mechanical damage. While some habitat-forming species may grow in isolation, when they occur in aggregations they form three-dimensional structures that visually and functionally characterise an area, providing shelter, protection and resources for other marine flora and fauna (Anderson et al. 2019). These living structures enhance ecological function and resilience, and often support higher biodiversity than surrounding areas, making them critical hotspots for ecosystem services and species interactions.

The sensitivity of these habitats arises both from their structural fragility and the vulnerability of the biological communities they support. Loss or degradation of these structures can result in long-term ecological consequences, including cascading effects on associated species and ecosystem services.

Occurrence within the proposal area

No sensitive habitats were observed within the proposal area.[5]

Occurrence across the wider survey area

The following habitats identified in the survey area are considered of high ecological value and sensitive to physical disturbance from seabed-altering activities such as marine farming. As clarified earlier, none of these habitats were observed within the proposal area, aside from minor flecks considered to be model artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance. For clarity, all subsequent references to 'biogenic habitat' in this report refer to ecologically significant or extensive areas of these habitat types, as observed and verified in the broader survey area. Minor artefactual flecks within the proposal area are excluded from this definition.

Bryozoan-sponge reefs (7% of the survey area)

- **Description** – These reefs are formed by dense aggregations of bryozoans and sponges, creating complex, three-dimensional biogenic structures. Bryozoans occur in massive, encrusting, erect branching and bushy forms. Reef-building species of particular ecological importance include *Celleporaria agglutinans* (Bradstock and Gordon 1983) and *Cinctipora elegans*, which together with sponges provide the structural framework for reef development. Sponges were also an ecologically important component of these reefs, performing roles such as habitat provision, substrate stabilisation and nutrient cycling (Bell 2008). Species include (likely) *Dactylia varia* and *Crella incrustans* and (possibly) *Iophon minor* and *Chondropsis* cf. *kirkii*.

[4] Although we cannot exclude this possibility given, for example, the cryptic nature and naturally low abundances of some 'Threatened' or 'At Risk' taxa.

[5] Only minor flecks considered to be model artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance were predicted within the proposal area; no true biogenic habitat was observed.

- **Associated fauna** – These reefs support diverse fauna, including tube worms (Serpulidae, likely *Galeolaria hystrix*), brachiopods (*Neothyris lenticularis* or *Magasella sanguinea*), large bivalves (*Ostrea chilensis, Pecten novaezelandiae, Tucetona laticostata, Modiolus areolatus*) and fish such as blue cod, tarakihi and leather jackets. Additional fauna, such as anemones, sea cucumbers and sea stars, are summarised in Table 1.
- **Ecological significance thresholds** – Bryozoan beds are defined as areas where large frame-building bryozoans (> 50 mm in three dimensions) occur at > 4% mean cover over broad areas, or dominate small areas (> 50% cover per m^2) (MacDiarmid et al. 2013). Sponge gardens are defined as areas with ≥ 25% cover of sponges across ≥ 100 m^2. Observed reef areas had ~ 30% bryozoan cover (patches with > 50%) and ~ 28% sponge cover, exceeding these thresholds.
- **Sensitive taxa** – Reef-building bryozoans (*C. agglutinans*, *C. elegans*), large erect / encrusting sponges, fragile branching bryozoans (*Hornera foliacea*), tube worms (Serpulidae), brachiopods and large bivalves. These taxa are long-lived, slow to recover and / or provide key structural complexity.
- **Ecological context** – Bryozoan-sponge reefs occur along Aotearoa New Zealand's continental shelf (e.g. Otago Shelf, Foveaux Strait, and parts of the Tasman Bay / Te Tai-o-Aorere and Hauraki Gulf / Tīkapa Moana regions [Jones et al. 2016]) but have been substantially reduced in Foveaux Strait by historical oyster dredging (Bradstock and Gordon 1983; Cranfield et al. 1999). While degraded or isolated patches may not function as full ecosystems in the pre-disturbance sense (Morrisey 2022), the Department of Conservation (DOC) submission on the previous application suggests that even remnants are recognised under Policy 11(a)(iii) of the New Zealand Coastal Policy Statement 2010 (NZCPS 2010) as indigenous ecosystems that are threatened in the coastal environment, which requires avoidance of adverse effects. Regardless of this distinction, the reefs remain ecologically significant and warrant protection due to their structural complexity, role as habitat for associated fauna and contribution to regional biodiversity. Bryozoan-sponge reefs also meet several Policy11(b) criteria, including (and see Morrisey 2022) requiring the avoidance of significant adverse effects:
    - Policy 11(b)(ii) – habitats in the coastal environment that are important during the vulnerable life stages of indigenous species [e.g. dredge oysters and blue cod].
    - Policy 11(b)(iii) – indigenous ecosystems and habitats that are only found in the coastal environment and are particularly vulnerable to modification [e.g. damage by anchoring, bottom-contact fishing, sedimentation, and other anthropogenic and climate-related factors].
    - Policy 11(b)(iv) – habitats of indigenous species in the coastal environment that are important for recreational, commercial, traditional or cultural purposes [e.g. for harvesting of blue cod and oysters].

Bushy-bryozoan thickets (4% of the survey area)

- **Description** – Dominated by *Orthoscuticella innominata*, with smaller *Cinctipora elegans*, calcareous tube worms (*Galeolaria hystrix*), and erect / encrusting sponges. These thickets create three-dimensional habitats supporting diverse epifauna, including brachiopods, large bivalves and fish. Tube worms form dense calcareous tubes that provide structural habitat for other organisms.
- **Ecological significance thresholds** – While *O. innominata* thickets are not explicitly classified as significant in MacDiarmid et al. (2013), they meet functional criteria for sensitive marine habitat by

providing three-dimensional structure and supporting diverse benthic assemblages. Bushy bryozoans reached up to 80% cover in places (average ~ 22%). Tube worm abundance varied from patchy to > 60% of video frame coverage.

- **Sensitive taxa** – Bushy bryozoans (*O. innominata*), small reef-building bryozoans (*C. elegans*), erect / encrusting sponges, tube worms (Serpulidae), brachiopods and large bivalves. These taxa are long-lived, slow to recover and / or provide key structural complexity.
- **Ecological context** – These thickets meet functional criteria for a sensitive seabed habitat due to their structural complexity and support of diverse assemblages. Associated fauna further highlight their ecological value. Bushy-bryozoan thickets also meet several Policy 11(b) criteria, requiring the avoidance of significant adverse effects:
    - Policy 11(b)(ii) – habitats in the coastal environment that are important during the vulnerable life stages of indigenous species [e.g. dredge oysters and blue cod].
    - Policy 11(b)(iii) – indigenous ecosystems and habitats that are only found in the coastal environment and are particularly vulnerable to modification [e.g. damage by anchoring, bottom-contact fishing, sedimentation, and other anthropogenic and climate-related factors].
    - Policy 11(b)(iv) – habitats of indigenous species in the coastal environment that are important for recreational, commercial, traditional or cultural purposes [e.g. for harvesting of blue cod and oysters].

Macrofaunal communities

The macrofaunal community within the proposal area is broadly comparable to those found in other coarse sandy sediments in the wider region, such as at the entrance to Big Glory Bay (James et al. 2018). Across 31 sediment samples, a total of 1,060 individuals representing 90 taxa were recorded. Total macrofaunal abundance was low, ranging from 1 to 88 individuals per core, with an average of 34 individuals per core (Figure 7). No taxa of high ecological importance were identified.

Low macrofaunal abundance is characteristic of coarse sandy sediments with limited organic matter (McLachlan et al. 1984), and values observed here are consistent with other sandy environments in the region. For instance, the entrance to Big Glory Bay typically supports 12–60 individuals per core (James et al. 2018).

Nematode worms were the most abundant taxa (177 individuals), followed by oligochaetes (153 individuals), which were present in all but one sample (sample 14). This pattern is typical of high-energy sandy habitats, where nematodes often dominate (Urban-Malinga et al. 2006).

Species richness ranged from 1 to 26 taxa per core, averaging 11 taxa per core (Figure 8). This is similar to values reported from other sandy sites in the region; for example, the entrance to Big Glory Bay had 9–18 taxa per core between 2012 and 2017 (James et al. 2018).

Polychaetes were highly diverse, with 33 taxa recorded. The most abundant families included Exogoninae, Cirratulidae, Syllidae, Spionidae and Terebellidae. Twelve bivalve taxa were also recorded, although in low numbers (1–5 individuals per taxon across all sites). The presence of a diverse assemblage of polychaetes and bivalves is consistent with communities found at the Big Glory Bay entrance, suggesting the assemblages observed here are representative of the broader Rakiura region.

Taxa were grouped to phylum level, ranked by relative abundance, and their spatial distributions were mapped (Figure 9). A complete taxa list and summary indices are provided in Appendices 5 and 6, respectively.

Multivariate analysis of macrofaunal assemblages identified two distinct groups (Group e and Group b), with 33% similarity, excluding three outlier samples (8, 15 and 38) (Figure 10). Sediment grain size appears to be the primary driver of community structure, as indicated by vector overlays and sample distributions (Figure 10). Group b corresponded to sand-dominated sites, while Group e included areas with greater proportions of gravel and mud. Group e also showed higher species richness and total abundance than Group b. Although grab samples were not collected from the northeastern part of the proposal area, bathymetry, backscatter and video imagery indicates that this area shares similar depth and substrate characteristics with Group e sites (e.g. Stations 16, 29 and 20; Figure 4). Based on this, it is reasonable to infer that the northeastern area is compositionally similar and may support a comparable macrofaunal community to Group e.[6]

Key taxa distinguishing the two groups included polychaetes from the families Cirratulidae, Exogoninae, Syllidae, Terebellidae and Spionidae, as well as nematodes, oligochaetes, ribbon worms (Nemertea), Munnidae isopods and cnidarians, which were more common in Group e samples.

[6] Targeted sediment sampling for this area is recommended as part of the baseline monitoring programme.

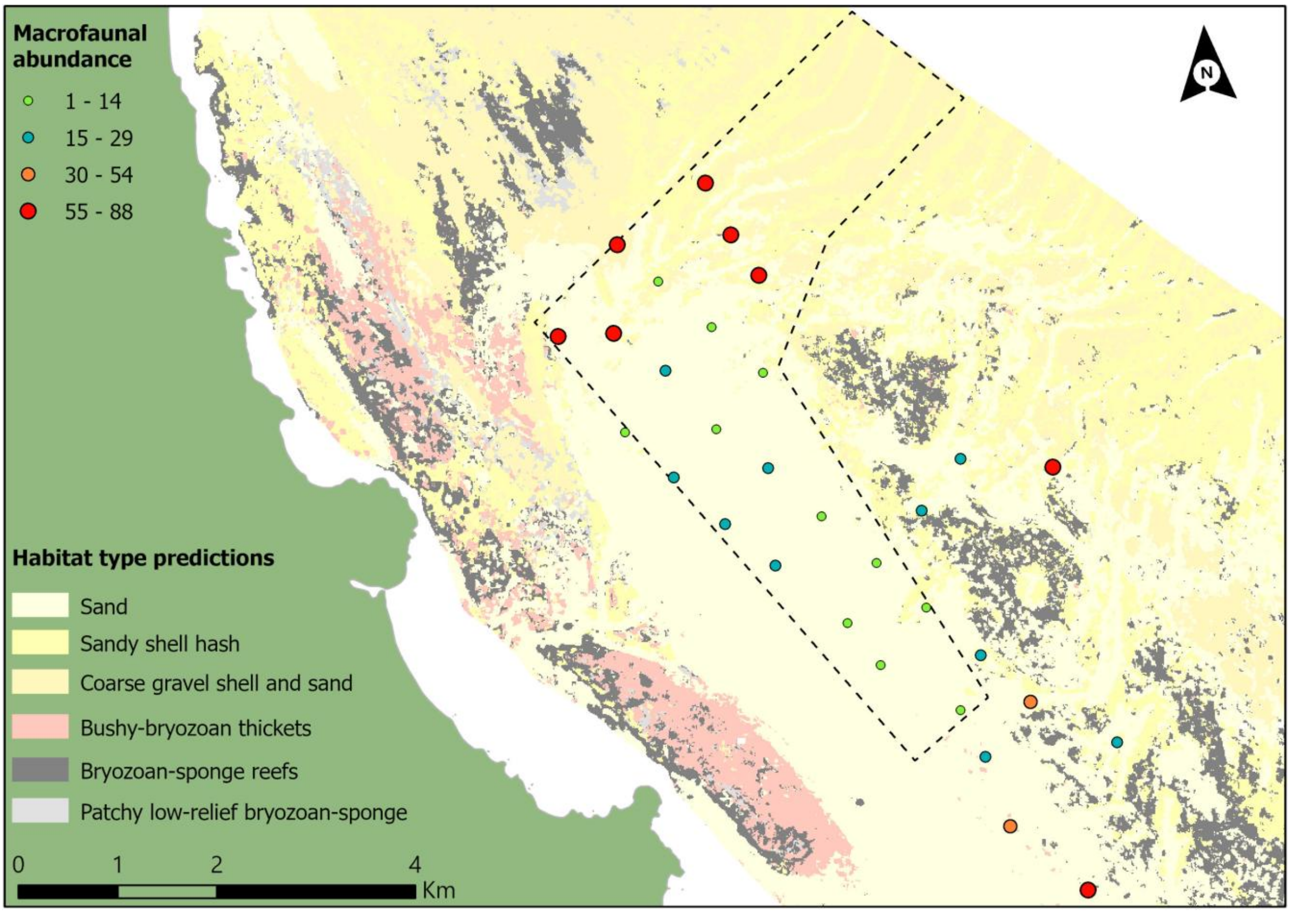

Figure 7. Macrofauna abundance per sediment core (10 cm deep and 113 cm^2 surface area) sampled across the survey area.

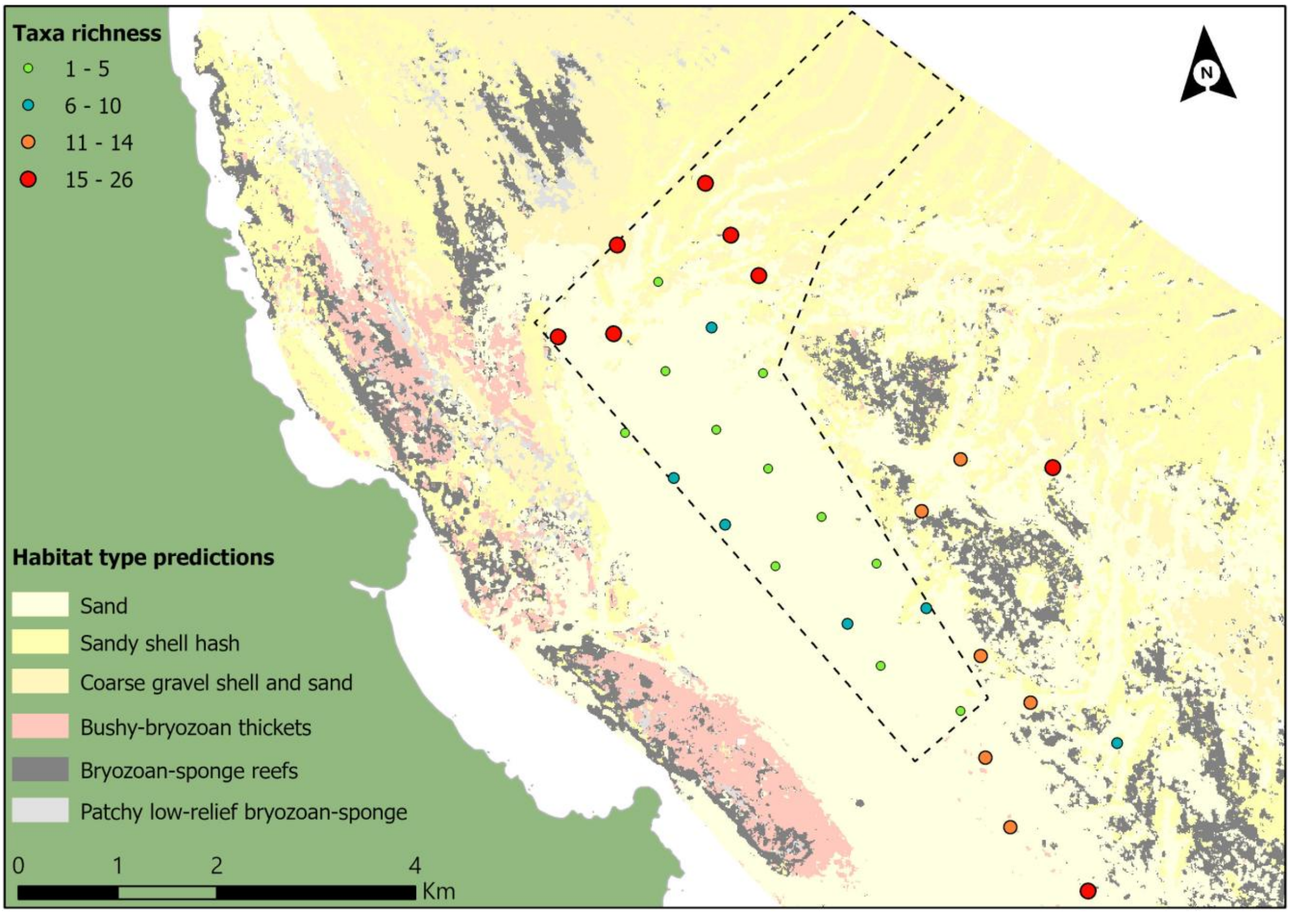

Figure 8. Macrofauna taxa richness (number of taxa) per sediment core (10 cm deep and 113 cm^2 surface area) sampled across the survey area.

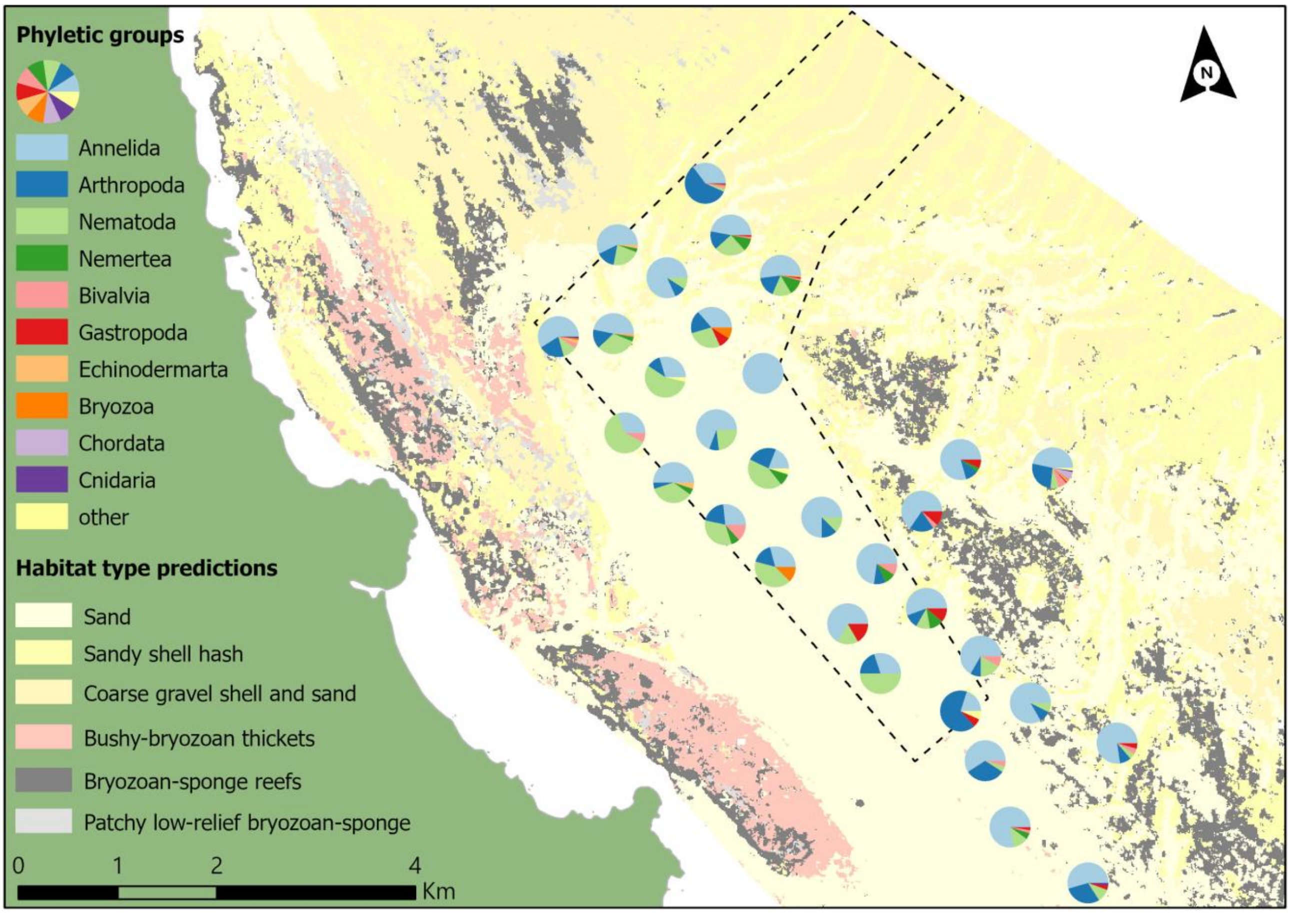

Figure 9. Macrofauna abundance per sediment core (10 cm deep and 113 cm² surface area) grouped by phylum, sampled across the survey area. For clarity, only the most numerically abundant phyla are shown. The category 'Other' includes Porifera, Phoronida and Sipuncula.

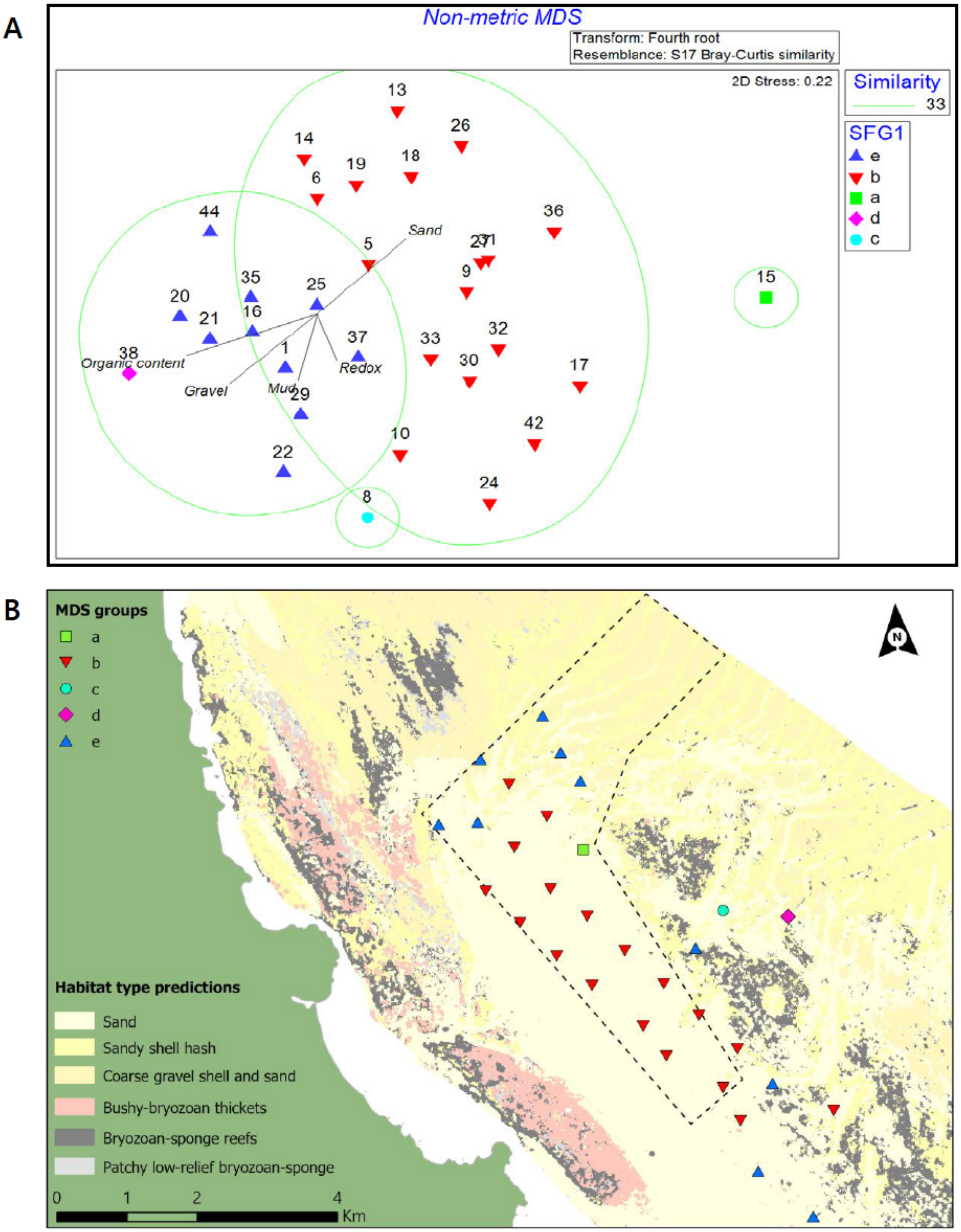

Figure 10. Similarity (%) among macrofauna communities from core sample contents across the survey area (A) Non-metric multi-dimensional scaling (MDS) plot of similarities between macrofaunal assemblages (clustered at 33% similarity), overlaid with vectors of sediment properties (particle grain size, organic content and redox). Abundance data were 10-root transformed. Resemblance based on Bray–Curtis similarities. (B) Location of grab samples, indicating location of MDS groupings relative to habitat across the survey area.

3.3 Seabed ecological value summary

Additional habitat observations across the proposal area and wider survey area since the previous assessment (Bennett et al. 2022) have increased confidence in, and understanding of, the boundaries and extent of habitat types within and adjacent to the Hananui proposal area. The seabed comprises six primary habitat types that differ in physical structure, biotic assemblages and ecological function.

The proposal area is dominated by sand, sandy shell hash, and coarse gravel with shell and sand. These habitats are widespread and structurally simple, and support sparse epifaunal communities. Sandy shell hash and coarse gravel are slightly more structurally complex and support marginally higher epifaunal diversity, but remain typical of common regional habitats.

In contrast, biogenic habitats such as bushy-bryozoan thickets, patchy low-relief bryozoan-sponge assemblages and bryozoan-sponge reefs located outside the proposal area are comparatively rare, functionally significant and sensitive to disturbance. These habitats provide three-dimensional structure, support diverse epifaunal communities and fulfil key ecological roles, including nursery and refuge functions, nutrient cycling, and habitat for culturally and commercially important species. Rock outcrops outside the proposal area similarly support diverse epifauna and are also functionally important.

Ecological values were assigned to each habitat type using the Environment Institute of Australia and New Zealand Ecological Impact Assessment Module 1 (EIANZ 2024), based on rarity, structural complexity, associated biota and functional importance. Bryozoan-sponge reefs are rare compared to soft sediment habitats, are slow to recover once disturbed, and considered to meet the definition of an indigenous ecosystem that is threatened in the coastal environment under Policy 11(a)(iii) of the NZCPS 2010 (DOC 2019). These reefs are therefore assigned a 'High to Very High' ecological value.

Bushy-bryozoan thickets meet functional criteria for sensitive marine habitats by providing important structural habitat for associated fauna, including fish, brachiopods and large bivalves. They contain small colonies of reef-building bryozoans and other biogenic taxa, include elements of threatened reef ecosystems and perform key ecological functions, supporting a 'Moderate–High' ecological value.

Patchy low-relief bryozoan-sponge assemblages represent an early successional stage in reef development. Although less structurally complex than mature reefs, they provide important ecological functions and contribute to the broader successional trajectory of biogenic reef habitats. These assemblages are assigned a 'Moderate–High' ecological value.

In contrast, the common soft sediment habitats (sand, sandy shell hash, and coarse gravel with shell and sand) are widespread, comparatively homogenous, and generally resilient to localised physical disturbance and organic enrichment. They provide limited structural complexity and support relatively low ecological function, and are therefore assigned 'Low' to 'Low–Moderate' ecological values. The sand and sandy shell hash habitats are naturally simple and widespread, supporting low abundances of macrofauna and epifauna. The EIANZ (2024) guidelines often associate 'Low' ecological value with impacted or degraded soft sediment habitats. In this case, however, the 'Low' ecological value rating reflects inherent characteristics rather than any signs of degradation. While sand provides important ecosystem functions such as habitat for infauna, foraging by demersal fish and nutrient cycling, these functions are relatively modest and do not increase the habitat's ecological value above 'Low'. Coarse gravel with shell and sand supports slightly higher macrofaunal abundance and diversity compared with bare sand, due to the greater structural complexity of shell and gravel fragments. However, the area has been subject to historical dredging, which has modified the habitat and potentially reduced its structural heterogeneity and faunal richness. Consequently, the habitat is assigned a 'Low–Moderate' ecological value, reflecting both its intrinsic characteristics and the influence of anthropogenic modification.

This habitat and ecological value synthesis provides the basis for assessing potential effects of the proposed activities, and for prioritising mitigation and adaptive management. A full summary of habitat types and their ecological value assignments is provided in Table 2.

Table 2. Summary of seabed habitat types within the Hananui proposal area (sand, sandy shell hash and coarse gravel with shell and sand only) and wider survey area (all six habitat types), their key characteristics, ecological attributes and assigned ecological value (EIANZ 2024), and supporting rationale.

Habitat type	Key characteristics	Ecological attributes	Assigned ecological value (EIANZ 2024)	Rationale
Sand (66% of proposal area, 39% of survey area)	Well-sorted fine to medium sand, low organic matter, oxygenated. Very sparse epifauna. Macrofaunal communities with low abundance and moderate diversity. Assemblages typical of regional sandy habitats. No taxa of high ecological importance were identified.	Homogenous sediment, low biotic diversity / abundance, limited structural complexity, relatively resilient.	**Low**	Widespread and common, low diversity, resilient to disturbance; provides limited ecological functions beyond basic benthic habitat.
Sandy shell hash (16% of proposal area, 27% of survey area)	Similar to sand but with higher shell content and slightly higher epifaunal diversity / abundance. Macrofauna comparable to sand habitat. No taxa of high ecological importance were identified.	Increased heterogeneity compared to sand, patchy biogenic elements, moderate epifaunal use.	**Low**	Widespread and common, low diversity, resilient to disturbance; provides limited ecological functions beyond basic benthic habitat.
Coarse gravel with shell and sand (18% of proposal area, 22% of survey area)	Mixed coarse substrate with slightly higher epifaunal diversity / abundance than sand and sandy shell hash, with occasional erect bryozoans and sponges. Macrofaunal abundance and richness slightly higher in gravel-rich samples.	Greater physical heterogeneity than sand; provides stable substrate for settlement; patchy but persistent biogenic features.	**Low–Moderate**	Slightly higher diversity and habitat complexity than sand / sandy shell hash. Still relatively widespread and resilient, but with some sensitive elements.
Bushy-bryozoan thickets (0%* of proposal area, 4% of survey area)	*Orthoscuticella innominata* and calcareous tube worm mounds forming moderate to high topographic complexity. High epifaunal diversity (sponges, bryozoans, hydroids, large bivalves, brachiopods, echinoderms), with sensitive taxa	Moderate to high complexity supports moderate to high biodiversity, regionally distinctive, vulnerable to disturbance. Possible	**Moderate–High**	Regionally distinctive and structurally important habitat. Supports sensitive taxa and fish. Vulnerable to disturbance.

Habitat type	Key characteristics	Ecological attributes	Assigned ecological value (EIANZ 2024)	Rationale
	present (e.g. tube worm mounds). Include elements of threatened reef ecosystems (possible successional stage). Moderate to high diversity and abundance of fish. Moderate spatial extent and patchiness.	successional stage for more complex reef habitat.		
Bryozoan-sponge reefs (0%* of proposal area, 7% of survey area)	Reef-building bryozoans (*Celleporaria* sp., *Cinctipora* sp.) and sponges. High epifaunal diversity (ascidians, hydroids, bivalves, brachiopods, anemones, echinoderms). Declining habitat type in Foveaux Strait.	Complex, developing to mature biogenic reef; high species richness; important ecological role; very sensitive to physical disturbance. Limited habitat modification where mature examples exist, however developing or earlier successional stages modified (by dredging).	**High–Very High**	Key habitat-forming species, regionally significant, threatened habitat in Foveaux Strait (considered to meet Policy 11(a) criteria for threatened indigenous ecosystem type), slow recovery, high ecological value.
Patchy low-relief bryozoan-sponge habitat (0%* of proposal area, 1% of survey area)	Sparse, low-profile bryozoans / sponges in coarse sediments. Early successional stage (Cranfield et al. 2004 model). Lower biodiversity and complexity than developing and mature reefs.	Transitional biogenic habitat with limited structural complexity; lower diversity than more mature reefs but with potential to develop into reefs with higher topographic complexity and diversity.	**Moderate–High**	Provides some complexity and ecological function, but less than reefs. Ecologically significant as precursor to more complex stages; moderate habitat modification.

* Values < 0.1% were rounded to 0% for reporting clarity. These are considered model artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance and not representative of verified biogenic habitat.

4. Site selection

Oceanographic features of a site, including depth, currents and wave climate, influence the deposition and resuspension rates of salmon farm waste. The proposal area is a high-flow environment where wastes should be readily dispersed. Mean current speeds are strongest near the surface ($0.56\ m \cdot s^{-1}$) and weakest near the seabed ($0.34\ m \cdot s^{-1}$). The predominant current flows along a northwest / southeast axis. The mean and maximum significant wave heights predicted for the area (from a 37-year regional wave hindcast model) are approximately 1 m and 3 m, respectively. See (Wilson 2025) for a detailed account of the oceanographic features of the proposal area.

4.1 Farm placement

Farm locations within the proposal area were selected to avoid known areas of ecologically valuable and sensitive marine habitats (as discussed in Section 3.2). Proposed farms are at least 500 m from bushy-bryozoan thickets, bryozoan-sponge reefs and areas of patchy low-relief bryozoan-sponge habitat (Figure 11). Along the tidal axis (southeast–northwest), farm blocks maintain a minimum separation of 1,000 m from these habitat types.

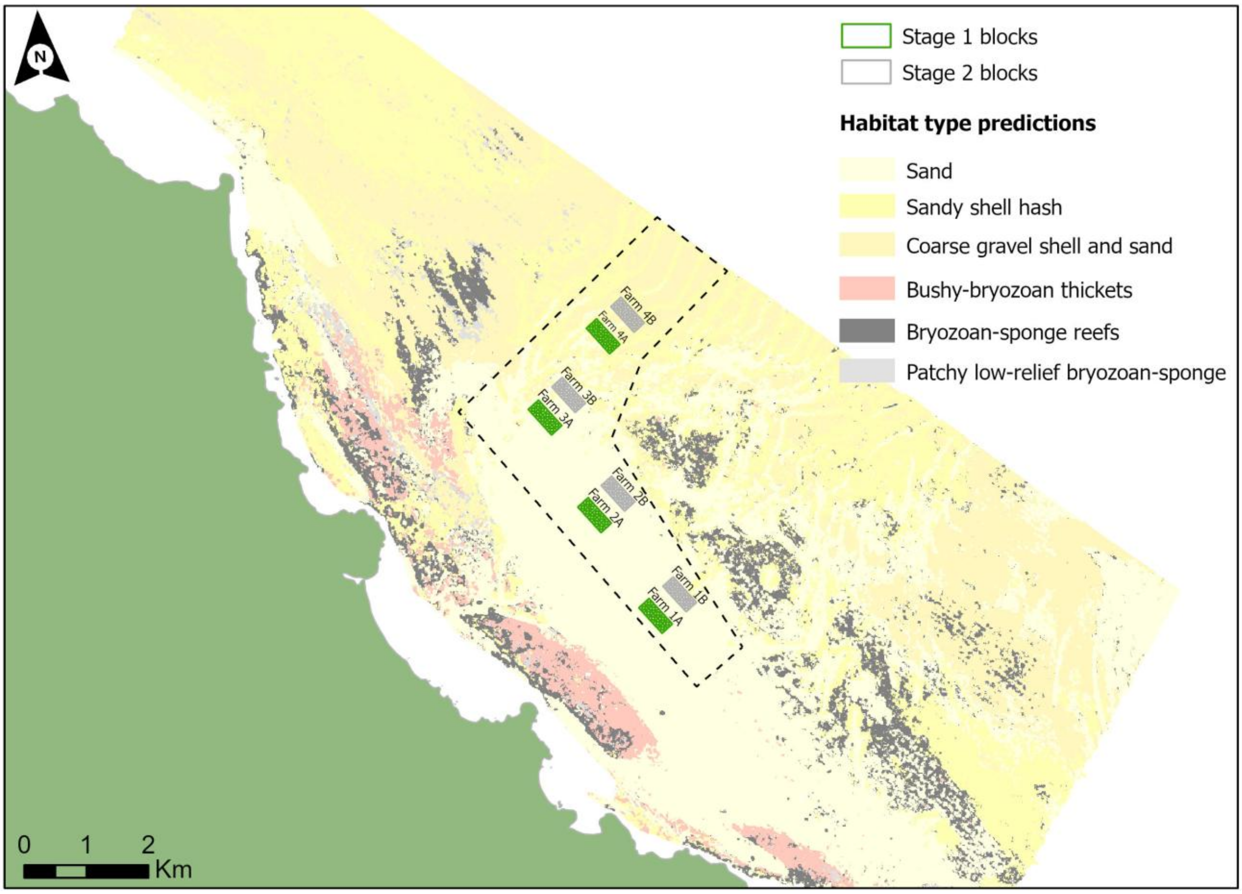

Figure 11. Proposed farm placement across the Hananui proposal area (black outline) over predicted habitat types.

4.2 Planned operations and development staging

The proposed development comprises four marine farms (referred to as Farms 1–4; Figure 11), each divided into two blocks (referred to as A and B blocks). Each block will contain ten 168 m-circumference pens (in a 5 × 2 configuration). Water depths across the proposed farm sites range from 25 m to 36 m. The maximum net depth will be 22 m; however, a minimum 5 m clearance will be maintained between the bottom of the net and the seabed. As a result, in some locations the net depth will be less than 22 m. Each 10-pen grid will be secured to a submerged mooring grid, which itself will be moored to the seabed with an array of moorings and anchors. Each farm will be serviced by an on-site barge that will be moored using this same system. The proposed farm layout at full production (i.e. Stage 2) is shown in Figure 11.

Ngāi Tahu Seafood proposes to develop the ~ 1,285 ha area in two stages. Stage 1 involves establishing one block of 10 pens at each of the four farm sites (A blocks), with a combined total feed discharge of up to 15,000 tonnes per annum. Stage 2 would increase the feed discharge to 25,000 tonnes per annum by adding a second block of 10 pens at each site (B blocks). Progression to Stage 2 will depend on the outcomes of environmental monitoring conducted over at least two full production cycles at each farm at the Stage 1 feed level, as set out in the proposed consent conditions and monitoring plan.

5. Assessment of environmental effects

Seabed impacts beneath salmon farms are inevitable, arising both from the physical presence of farm structures and from operational discharges. This section evaluates the potential effects of these activities on the seabed and outlines measures proposed to avoid, remedy or mitigate those effects where practicable. For each type of effect, a summary table is provided at the end of the relevant sub-section, outlining predicted effects and corresponding mitigation. An overall summary of all predicted effects, associated mitigation measures and the residual level of significance is then presented in Table 10 (Section 5.6).

5.1 Seabed effects associated with the initial site development

Ngāi Tahu Seafood proposes to install pens using an anchoring system similar to that used on overseas salmon farms (e.g. Storm Bay, Tasmania) and at New Zealand King Salmon's proposed Blue Endeavour site. Each 10-pen grid will be secured to a submerged mooring grid, which is itself moored to the seabed with a combination of dual-shank anchors, chain and 5-tonne mooring blocks. Disturbance from mooring installation may cause physical damage to the seabed and short-term resuspension of sediments. Both factors could affect biota in the area directly surrounding the installation area. Consequences for sensitive species include habitat destruction, smothering, sedimentation-induced reductions in feeding efficiency, and possible mortality (Clark et al. 2011; Anderson et al. 2019). A summary of potential environmental effects associated with installation of mooring systems, and options to avoid, remedy or mitigate these, where applicable, is provided in Table 3. An overall summary of all predicted effects, associated mitigation measures and the residual level of significance is presented in Table 10 (Section 5.6).

Destruction and displacement of species and habitats

Substrates beneath the proposed farms are primarily sand with varying amounts of shell hash (Figure 11). The proposed farm footprints do not overlap with areas of significant biogenic habitat. However, very occasionally potentially sensitive species such as sponges, bryozoans, brachiopods and large bivalves (e.g. oysters, scallops, dog cockles and horse mussels) may occur within these sand-dominated areas (see Table 1 in Section 3).

Anchor installation in proximity to these taxa may result in a range of effects, from localised reductions in species densities to complete displacement or exclusion within the anchor footprint. Any organisms present at the point of installation are likely to be displaced or damaged. However, the potential ecological significance of these impacts is reduced by the relatively low diversity and very patchy distribution of epibiota within the areas proposed for farm development.

Resuspension of sediments

Anchor installation may result in temporary sediment resuspension, which could pose a short-term risk of smothering nearby epibiota. However, this disturbance is expected to be brief, limited to areas directly around anchor sites and to the installation period and the hours or days immediately following, and is therefore unlikely to have lasting impacts on benthic communities within the area. Additionally, strong currents in the region will help minimise these effects by rapidly dispersing suspended sediments.

Seabed effects associated with the initial site development are mitigated by appropriate site selection that avoids sensitive biogenic habitat. Farm development will be carried out by experienced crew using best-practice installation methods. It is also worth noting the presence of existing anchorages in proximity to the proposed site, which are used by large vessels. These anchor deployments may cause more frequent and widespread disturbance compared to the proposed moorings, where seabed disturbance is limited to a one-off installation event. Additionally, dredge fishing occurs in the region and is known to generate significant sediment disruption. As such, the area is already subject to a degree of anthropogenic disturbance.

Table 3. Summary of potential environmental effects from the installation of mooring systems, and measures to avoid, remedy or mitigate these effects, where practicable.

Potential effect	Environmental implications	Options to avoid, remedy or mitigate (where applicable)
Destruction / displacement of species and habitats	Installation of each anchor is likely to displace or disturb organisms within a small footprint of seabed.	Anchor sites are in sand-dominated habitats with low epibiotic diversity; sensitive biogenic habitats (e.g. bryozoan-sponge reefs) are avoided.
Short-term resuspension of sediments	Localised resuspension and resettlement of fine sediments may occur during anchor installation, but is expected to be limited in duration (hours to days) and extent. Dispersive site conditions will reduce the potential for far-field effects.	Installation by qualified and experienced personnel will minimise disturbance.

5.2 Seabed effects associated with the presence of farm structures

Once established, farm structures may reduce light penetration to the seabed and create new habitat for fouling organisms, including potential marine pests (Forrest and Johnston 2025). Conversely, the presence of these structures can also provide a degree of protection from physically destructive fishing activities such as bottom trawling or dredging. Table 4 summarises the potential environmental effects associated with the ongoing presence of farm infrastructure, along with options to avoid, remedy or mitigate these effects, where applicable. An overall summary of all predicted effects, associated mitigation measures and the residual level of significance is presented in Table 10 (Section 5.6).

Shading

Shading from farm structures can reduce sunlight reaching the seabed, potentially affecting photosynthetic organisms and associated food webs (Keeley 2013). However, within the proposed farm area, photosynthetic taxa are scarce, with only drift algae observed in locations likely to be affected by shading (i.e. directly beneath the pens). As such, the risk of shading having a discernible impact on the seabed environment is considered negligible.

Provision of habitat

Farm structures can create new habitat for fouling organisms, which may alter local ecological dynamics. Fouling biomass dropping from structures can change the physical and biological composition of the seabed, potentially contributing to organic enrichment through decomposition. Scavenging species such as brittle stars and sea cucumbers may aggregate beneath farms to feed on fallen material, which could influence the composition of the benthic epifaunal community.

However, as farms are sited over predominantly sandy habitats with varying degrees of shell and gravel, these effects are expected to be minor and can be effectively managed through regular monitoring and maintenance of farm infrastructure, including periodic removal of fouling organisms (NTS 2025).

Farm structures may also be colonised by native or exotic pest species, which could act as propagule sources for surrounding habitats. These risks are specifically addressed in the biosecurity assessment of effects (Forrest and Johnston 2025).

Protection

The presence of farm structures may offer protective benefits for certain organisms and habitats, particularly those tolerant of organic deposition, by limiting access to destructive fishing activities such as bottom trawling or dredging (Fletcher 2015). Although sensitive taxa, including bryozoans, sponges, brachiopods and large bivalves, have been observed directly beneath or adjacent to the proposed farm sites only occasionally, they are known to be highly vulnerable to dredging impacts (Anderson et al. 2019). This consideration is especially relevant in Foveaux Strait, where dredge fishing occurs frequently and can pose a significant threat to benthic communities.

Table 4. Summary of potential environmental effects associated with the presence of farm structures, and measures to avoid, remedy or mitigate these effects, where practicable.

Potential effect	Environmental implications	Options to avoid, remedy or mitigate (where applicable)
Shading of seabed	Farm structures may reduce the amount of sunlight reaching the seabed, potentially lowering food availability for photosynthetic organisms. However, few, if any, photosynthetic taxa are present at the site, so impacts are expected to be negligible.	Not applicable.
Biofouling drop-off	Colonisation of farm structures by fouling organisms is likely. Natural sloughing or net cleaning may result in this material falling to the seabed, potentially altering sediment characteristics and contributing to organic enrichment.	Regular inspection and maintenance of farm structures to manage biofouling, with a focus on early operational stages to assess extent and nature of fouling. Refer to the Biosecurity Management Plan (NTS 2025).
Colonisation by pest species	Farm structures may be colonised by marine pests, which could act as propagule sources and facilitate spread to nearby habitats.	As above. See also the biosecurity assessment of effects (Forrest and Johnston 2025) and Biosecurity Management Plan (NTS 2025).
Protection from destructive activities	Farm structures will provide protection from destructive activities, including bottom trawling and dredging.	Not applicable.

5.3 Seabed effects resulting from organic deposition during farm operation

The primary seabed impacts of finfish aquaculture arise from the deposition of faeces and uneaten feed, which can lead to localised over-enrichment of sediments (Keeley 2013). Where deposition rates are high, microbial decomposition of this organic material can significantly alter sediment chemistry and benthic ecology (Holmer et al. 2005). In this section, the mechanisms and ecological consequences of organic deposition are first described. Predictive depositional models are then used to estimate the spatial extent and magnitude of seabed deposition from the proposed operation. Finally, the model outputs are applied to assess potential changes in sediment characteristics and benthic communities across both the proposal area and the wider survey area.

Summary of seabed effects from organic deposition

High organic loading can result in the accumulation of biodeposits that smother benthic organisms, reduce sediment permeability and shift benthic community structure towards opportunistic, enrichment-tolerant species (Wildish and Pohle 2005). Elevated organic inputs also increase food supply for epifauna, and biodeposition of feed and faeces (including material sloughed from fouling organisms) can attract scavengers and predators such as sea cucumbers, sea stars, crabs and isopods (Kutti et al. 2007; Woodcock et al. 2018).

In addition to organic deposition, nutrient release can influence benthic processes. Dissolved nitrogen and phosphorus released from salmon farms may locally elevate nutrient concentrations and, where suitable substrates occur, support benthic algal growth (Sanderson et al. 2012; Oh et al. 2015). Surveys undertaken as a part of this assessment, indicate that photosynthesising taxa are sparse or absent across the proposal area, possibly reflecting both nutrient limitation and physical factors such as strong currents, wave exposure, depth and a lack of stable substrate that could restrict algal colonisation. While nutrient enrichment from farm operations could promote algal growth on any available stable surfaces, the likelihood of substantial proliferation remains low because dissolved nutrients are likely to be rapidly dispersed, light is limited with depth and high-energy conditions may constrain settlement and persistence of algal growth.

Increased turbidity caused by resuspension of fine particulates may reduce light penetration to the seabed and potentially limit primary productivity (Keeley 2013). However, this is unlikely to be significant at the proposal site, where photosynthetic organisms are sparse or absent. Excessive organic loading can also lead to depletion of near-bottom oxygen levels, creating hypoxic conditions harmful to benthic fauna (Sanz-Lázaro and Marín 2008; Zhang et al. 2018).

The magnitude and spatial extent of seabed effects depend on multiple interrelated factors, including farm operational parameters and environmental characteristics (Keeley et al. 2013). Organic loading is influenced by stocking density, feed type and delivery system, fish faeces settling velocity, pen design, and hydrodynamic effects such as flow reduction caused by farm structures.

Environmental factors such as water depth and current velocity play a key role in flushing and assimilation of waste materials (Keeley 2013). Additionally, seabed topography and seasonal variables such as water temperature may modulate local hydrodynamics and waste dispersion (Keeley 2013). Increased flushing reduces localised sedimentation and biodeposition while enhancing sediment oxygenation, mitigating negative impacts (Findlay and Watling 1997; Plew 2019).

Seabed impacts are generally most pronounced directly beneath farm pens and diminish with distance. In 'deepwater' sites (water depth > ~ 30 m) with strong currents (depth-averaged speeds > 15 $cm{\cdot}s^{-1}$), such as the proposal area, biodeposit footprints are more dispersed with less concentrated enrichment compared to shallower, low-flushing sites (Keeley et al. 2019).

Sediment texture strongly influences waste resuspension and dispersal (Law et al. 2016; Law 2019). The coarse sandy sediments typical of the proposal area are less cohesive and more prone to resuspension than finer substrates. While sandy sediments can retain some organic material, their higher mobility increases the potential for particle dispersal (Law et al. 2016).

Resuspension of deposited material leads to broader dispersion, reducing localised seabed impacts beneath the farm. Recent studies have shown that waste particles can disperse up to 1.3 km from farms

(Keeley et al. 2019, 2024), but beyond 600 m ecological effects are generally weak or absent due to effective dilution and assimilation by the environment (Keeley et al. 2024).

Although understanding of far-field effects remains limited (Law et al. 2016), far-field deposition is expected to occur at reduced rates, with waste largely assimilated without measurable ecological changes (Bannister et al. 2016; Keeley et al. 2024). Nonetheless, particulate organic matter may accumulate in nearby depositional zones such as low-flow areas, seafloor depressions or structurally complex habitats (e.g. reefs and bivalve beds). Elevated inputs in these areas could lead to localised enrichment, increased availability of organic particulates and dissolved nutrients, and heightened turbidity (Woodcock et al. 2018; Weitzman et al. 2019), extending beyond the immediate depositional footprint.

Given the relatively shallow depth, strong currents and non-cohesive sandy sediments in the proposal area, significant resuspension and dispersal of biodeposits are anticipated. This will likely result in wide spatial distribution of organic particles beyond the primary farm footprint (Keeley et al. 2019) and lower accumulation of organic material immediately beneath the farm. A summary of potential environmental effects associated with organic deposition during farm operation, and options to avoid, remedy or mitigate impacts, where applicable, is provided in Table 5. An overall summary of all predicted effects, associated mitigation measures and the residual level of significance is presented in Table 10 (Section 5.6).

Table 5. Summary of potential environmental effects arising from organic deposition during farm operation, and measures to avoid, remedy or mitigate these effects, where practicable.

Potential effect	Environmental implications	Options to avoid, remedy or mitigate (where applicable)
Aggregation of mobile epibiota	Biodeposition of feed and faeces (including from fouling organisms) may attract scavengers and predators (e.g. sea cucumbers, sea stars, crabs, isopods). This can alter community structure through increased predation pressure.	• Avoid footprint overlap with biogenic habitats. • Monitor epifaunal composition and apply effects-based management (e.g. link monitoring results to operational responses).
Alterations to sediment properties	Microbial decay of deposited organic material can change sediment chemistry (e.g. oxygen depletion, elevated sulphides, reduced redox potential). Visible bacterial cover may occur.	• Apply effects-based management (e.g. fallowing and rotational site use) to prevent undesirable changes. • Optimise feed to minimise waste.
Alteration to macrofaunal communities	Elevated organic matter (uneaten feed, faeces) alters food availability and sediment conditions. Effects may include: • Reduced species richness and dominance of opportunistic taxa. • Increased total biomass and community assimilative capacity.	• Apply effects-based management (e.g. fallowing and rotational site use). • Optimise feed to minimise waste. • Carry out particle dispersal modelling to identify enrichment hotspots.

Potential effect	Environmental implications	Options to avoid, remedy or mitigate (where applicable)
	• Under high enrichment, azoic conditions from anoxia or smothering. Effects decrease with distance from the farm, but resuspension and dispersal in sandy sediments may cause localised far-field enrichment. Low-flow areas and deeper pockets are more sensitive.	
Alteration to epifaunal communities, particularly sensitive taxa	Increased organic matter alters food availability and sediment conditions, leading to: • Positive effects (enhanced food supply). • Negative effects (smothering, reduced reproductive success, larval settlement failure). • Increased turbidity may reduce light penetration and primary productivity, although few primary producers occur in the proposal area.	• Avoid siting farms where biogenic habitats or sensitive communities could be impacted. • Apply effects-based management (e.g. fallowing, rotational site use) to ensure sensitive communities remain healthy. • Optimise feed to minimise waste. • Carry out particle dispersal modelling to identify enrichment hotspots.
Oxygen depletion of overlying waters	Localised oxygen depletion may occur near the seabed beneath pens. Significant depletion is unlikely due to the dispersive nature of the site but could occur at high farming intensity (see Wilson [2025] for further discussion).	• Apply effects-based management (e.g. fallowing, rotational site use). • Apply best-practice feed management (optimise feed, minimise waste).
Algal overgrowth and enrichment from increased nutrients	Nutrients (primarily nitrogen and phosphorus) released from organic material can stimulate algal growth, and if excessive, may lead to algal overgrowth that alters seabed community structure and ecological functioning. Photosynthesising taxa are sparse or absent at the site, suggesting environmental conditions are not favourable. The likelihood of substantial benthic algal proliferation across the seabed is considered low, although some localised increases cannot be ruled out.	• Avoid siting farms where biogenic habitats or sensitive communities could be impacted. • Apply effects-based management (e.g. fallowing, rotational site use) to ensure sensitive communities remain healthy. • Optimise feed to minimise waste. • Carry out particle dispersal modelling to identify enrichment hotspots.

Sediment properties

Microbial decay of waste from salmon farming can significantly alter sediment chemistry, leading to oxygen depletion, elevated sulphide concentrations and reduced redox potential. These changes may also cause anoxic conditions in the overlying water and increase dissolved nutrient levels in both sediment porewater and the water column (Holmer et al. 2005; Keeley et al. 2013). Such chemical shifts are typically accompanied by changes in the composition of benthic communities.

The magnitude, persistence and spatial extent of these changes are influenced by seabed type and hydrodynamic regime. In muddy, low-energy environments, fine cohesive sediments have low permeability and higher capacity to retain deposited particulates (Papageorgiou et al. 2009). This leads to sustained oxygen depletion, higher organic carbon accumulation, and persistent anoxia and sulphide enrichment in surface layers.

However, in high-flow environments such as the proposal area, excessive accumulation of organic waste on the seabed is unlikely. Furthermore, the coarse sandy sediments found across most of the site are more readily oxygenated, which may facilitate the decomposition of farm wastes (Martinez-Garcia et al. 2015). More consolidated substrates, such as coarse gravel, sand and shell in the northeastern section of the proposal area, may have slightly greater potential for temporary waste retention compared to the more mobile, unconsolidated sandy habitats. Nonetheless, these coarser substrates remain far less cohesive than muddy sediments.

Waste distribution and accumulation can also be shaped by spatial variation in substrate and seabed structure, both within and outside the proposal area. Within the proposal area, troughs and flanks of subtidal dune and ridge features may act as temporary retention zones for organic material. Outside the proposal area, biogenic habitats may similarly retain fine particulates due to their structural complexity. However, given the high-flow conditions across the region and the considerable distance between these reef habitats and the proposed farm locations, the likelihood of significant deposition or adverse effects in these areas remains low.

Overall, while some localised accumulation may occur in certain areas, the seabed across the proposal area is expected to support relatively efficient waste dispersal and oxygenation. Consequently, the adverse effects typically associated with organic waste build-up are expected to be less pronounced here than at lower-flow, muddier sites.

Sediment macrofauna

Sediment macrofaunal communities generally follow the successional response to increasing organic enrichment described by Pearson and Rosenberg (1978). Initially, as organic matter is deposited, total abundance tends to increase gradually. When organic loading reaches higher levels, abundance rises more sharply while species diversity declines. Continued organic input leads to a peak that is dominated by opportunistic species, after which abundance sharply decreases as oxygen concentrations fall and hypoxic conditions develop.

The macrofaunal communities across the proposal area are typical of coarse sandy sediments with low organic content, characterised by naturally low overall abundance and moderate diversity (see Section 3.2). This inherently low abundance suggests a limited assimilative capacity for additional organic inputs, since fewer organisms are present to rework and consume this material (Pearson and Rosenberg 1978; Hyland et al. 2005). Although sediments with low organic content can accommodate some organic enrichment (Papageorgiou et al. 2009), even small increases can shift community composition (Hyland et al. 2005). Species adapted to low-food conditions, as found here, may struggle to process excess organic matter effectively (Macleod et al. 2007). However, deposit feeders such as oligochaete worms, various polychaetes and scavenging amphipods are present and could potentially increase in abundance, partially compensating by enhancing the waste-processing capacity of the community.

Ultimately, the low macrofaunal abundance observed at the site indicates a generally reduced assimilative capacity for organic waste, although this is moderated by species traits, sediment type, hydrodynamics and community composition (Pearson and Rosenberg 1978; Hyland et al. 2005).

Epibiota

Research to date has primarily focused on the ecological effects of salmon farms on sediment macrofaunal communities, while the direct depositional effects on large epibiota (i.e. > 0.5 mm) are poorly documented. Accordingly, there is limited information on the direct farm-related impacts on these important taxa.

Generally, if organisms consume farm waste, there may be positive effects such as increased growth due to an enhanced food supply (George and Parrish 2015; Bergvik et al. 2019), which may lead to increased population densities. Sub-lethal adverse effects are also possible; for example, a reduction in food quality due to epibiota consuming salmon feed may lower growth or reproduction potential (White et al. 2016), resulting in reduced densities. In addition, smothering may directly displace some organisms. Ultimately, however, effects are likely to be species-specific. Mobile scavengers and deposit feeders may also aggregate in areas where farm wastes and drop-off of fouling organisms provide a food supply (see also Section 5.2). Known responses to organic deposition of specific epibiota found in the proposal area are discussed below.

Bryozoans and sponges

Bryozoans and sponges are generally considered sensitive to organic deposition and sedimentation (Clark et al. 2011; Morrisey et al. 2015; Dunlop et al. 2021). Frame-building bryozoans, such as those that form the bryozoan-sponge reefs outside of the proposal area, are slow-growing, and recovery from wide-scale disturbance can take decades (Batson and Probert 2000). Whereas bushy bryozoans are comparatively fast-growing, suggesting they may be less sensitive to the effects of salmon farming (Smith 2021). Some bryozoan species have shown tolerance to organic deposition from salmon farms; for example, bryozoans settled and grew on artificial structures near farms in the Gulf of Aqaba, Red Sea, and in Iceland (Angel et al. 2002; Israel et al. 2016), and the biomass of epiphytic bryozoans on seaweed stipes increased with levels of farm-derived organic waste (Haugland et al. 2021). Sponges also display variable sensitivity to organic enrichment and sedimentation (Sutherland et al. 2018; Laroche et al. 2021; Laroche et al. 2022). In general, both the survival of translocated specimens and the occurrence of studied species is reported to be lowest near finfish pens, increasing at distances of approximately 80 to 200 m from the pens (Sutherland et al. 2018; Dunlop et al. 2021; Laroche et al. 2022).

Tube worms

Tube worms are suspension feeders that are susceptible to increased suspended sediments, which can block feeding appendages and affect respiration (Kupriyanova et al. 2001), and to sedimentation, which can smother and kill colonies. While tube worms are likely to be sensitive to high levels of farm-related deposition, the presence of tube worm (*Galeolaria* sp.) reefs near salmon farms in Big Glory Bay suggests that these organisms are tolerant to low levels of deposition and salmon farm-related enrichment effects (Anderson et al. 2019). Tube worm casings have also been observed on a few mussel shells adjacent to a finfish pen in Beatrix Bay, Marlborough Sounds (Millar 2025). Although it is unclear from video whether the worms were alive, their presence indicates persistence in areas influenced by farm activities.

Brachiopods and large bivalve species

Brachiopods and large bivalves are ciliary filter-feeders that draw in suspended particles and expel wastes using filament movements, making them potentially vulnerable to organic enrichment, suspended sediments and sedimentation (Hewitt and Pilditch 2004; Lohrer et al. 2006). Recent laboratory and field studies have demonstrated that brachiopods and large bivalves exhibit species-specific and generally sub-lethal responses to salmon farm-related enrichment (Elvines et al. 2024; McMullin et al. 2025). All tested species assimilated salmon faeces, indicating exposure pathways, but significant effects were typically observed only under moderate to high levels of organic deposition. At low enrichment, responses were subtle, mainly limited to changes in fatty acid composition and often below statistical detection thresholds. The horse mussel (*Atrina zelandica*) showed the strongest sensitivity, with consistent negative effects, including altered fatty acid profiles, increased respiration, stress-related gene expression and digestive gland atrophy. In contrast, the scallop (*Pecten novaezelandiae*) exhibited weaker or mixed responses, while the brachiopod *Neothyris lenticularis* appeared comparatively resilient, with delayed metabolic changes and minimal tissue-level effects. No mortality or oxidative damage attributable to enrichment was observed, but increasing inter-individual variability at higher enrichment levels suggests that these taxa approach their tolerance limits under elevated deposition. Collectively, the evidence suggest that brachiopods and large bivalves are tolerant to low levels of organic inputs, but that their ability to adapt becomes compromised with moderate to high enrichment.

Effects of fallowing on seabed enrichment

Fallowing farms between production cycles is a common management practice used internationally to reduce seabed impacts by allowing sediments time to recover before farming resumes (Black et al. 2008). Recovery rates depend on the extent of impact, such as the quantity and timing of organic matter deposition, site-specific characteristics such as sediment type and dispersal capacity, and the duration of the fallow period (Macleod et al. 2006; Zhulay et al. 2015). In Norway, standard fallow periods of 6–8 weeks are applied at the end of an 18-month production cycle (Black et al. 2008). During this period, partial recovery has been observed (Zhulay et al. 2015); however, full seabed recovery (i.e. a return to pre-farming conditions) typically takes several years (Keeley et al. 2014, 2019). Ngāi Tahu Seafood is planning to fallow each farm in the proposal area for 3 months between production cycles, allowing partial recovery of sediment chemistry and benthic communities. Interim monitoring of fallowed or low-production sites is recommended to track recovery dynamics and improve understanding of temporal seabed responses to enrichment.

5.4 Predicted spatial extent and magnitude of seabed deposition

Depositional modelling is a tool to estimate the intensity and spatial extent of solid waste from the proposed development on the seabed (i.e. the depositional footprint). Depositional modelling involves calculating the trajectories of millions of virtual particles representative of fish faeces and wasted feed pellets as they sink to the seabed. In the model, these particles are continuously released from fish pens and are dispersed by water currents as they sink to the seabed.

At high-flow sites such as the proposal area, particles that land on the seabed can be lifted back off it by water currents and subsequently be redispersed (resuspension). Additionally, macrofaunal and microbial consumption of particles, as well as shear forces acting on the particles, act to reduce the mass of waste particles over time (decay). Modelled trajectory data are compiled upon completion of the simulation and used to derive depositional footprints for the proposed salmon farms. These footprints are estimates of the extent and intensity of solid waste that will be deposited to the seabed under different peak feed scenarios. The modelled footprints are then used to inform expected ecological enrichment on the seabed and inform design of an appropriate environmental monitoring strategy.

Overview of models

Depositional modelling undertaken for the proposal area used the OceanTracker model to simulate particle behaviour (Vennell et al. 2021), in conjunction with a 2-year-duration three-dimensional (3D) hydrodynamic hindcast model of the Foveaux Strait region. The 3D hydrodynamic hindcast model provides a 3D time history of water current velocities throughout Foveaux Strait. Data from this model are used by OceanTracker to infer water current velocities at particle locations (Oceanum 2025).

OceanTracker (previously called VenOM in Bennett et al. [2022]) is a Lagrangian particle-tracking model that has been previously used to model solid waste deposition at other prospective and existing aquaculture sites. These sites include the Blue Endeavour site in Cook Strait (Elvines et al. 2021a), existing farms in Tory Channel / Kura Te Au (hereafter Tory Channel) (Elvines et al. 2021b), and proposed aquaculture settlement areas that from part of central government’s Treaty of Waitangi settlement commitments (Elvines et al. 2025a, 2025b). This model was also used in the initial application for this site (Bennett et al. 2022), although since that time several changes have been made to model features to more accurately represent the physical processes that determine solid waste deposition (Table 6).

The OceanTracker model is modular in design, such that different features can be added to the core model to simulate different particle behaviours (sinking velocities, resuspension dynamics, particle decay) and particle properties (feed inputs, feed conversion). Changes to the model since the previous effects assessment at this site include modelling faecal particles as three different size classes with different sinking velocities, varying particle resuspension dynamics according to benthic habitat, and varying feed inputs to farms over the course of a production cycle. More detailed methods regarding the parametrisation of OceanTracker for this assessment are provided in Appendix 7.

The underlying hydrodynamic hindcast model (Oceanum 2025) used to provide OceanTracker with water current velocity data at particle locations is a 2-year sub-sample (1 June 2018 to 31 May 2020) of the 10-year-duration hydrodynamic model used in the water column assessment (Wilson 2025). It was necessary to sub-sample these data because of computational limits on model run-time and output data size. The spatial coverage of the hydrodynamic model extends from Steep Head to Cave Point on the western boundary, and East Cape / Koromere to Waipapa Point on the eastern boundary (Figure 12). Model resolution is ~ 50 m at the farm, 150 m near the coast and 1 km at the open boundary. This model was parametrised using spatially varying seabed bottom roughness coefficients according to benthic habitat categories taken from the most recent habitat model of this area (see Section 3). Full details concerning the parametrisation of the model and its validation against measured data are available in Oceanum (2025).

Table 6. Summary of key differences made to the depositional modelling approach since the last application and their implications.

Updated modelling	Previous modelling	Implications
Model duration of 24 months.	Model duration of 6 weeks.	Modelling for a longer duration incorporates a greater range of ocean conditions into model predictions to create more robust statistics and depositional footprint estimates.
Feed inputs to farms vary over time to mimic intended operating conditions.	Farms modelled with constant feed input at peak feed.	Using more realistic feed inputs leads to more realistic estimates of solid waste. New modelling includes effects from feed loading prior to peak feed periods, and feed cessation during harvest and fallow periods.
Model outputs based on 3-month average around operational peaks. No model outputs produced during initial 3-month spin-up period.	Model outputs based on 2-week averages at peak feed, with 4-week spin-up period.	Using a longer averaging period around peak feed periods captures more variability in ocean conditions and waste distribution. Allowing the model to spin up for the first 3 months leads to more accurate calculations of waste.
Three classes of faecal particles modelled with different sinking rates.	One class of faecal particles modelled using a single mean sinking rate.	Improved resolution of the near-field footprint through inclusion of fast-sinking particles, and improved resolution of the far-field footprint through inclusion of slow-sinking fine particles.
Inclusion of variable bed roughness in hydrodynamic model and resuspension dynamics.	Uniform bed roughness assumed based on soft sediment habitat.	Inclusion of variable bed roughness based on benthic habitat provides a more realistic representation of variability in near-seabed currents and resuspension dynamics.

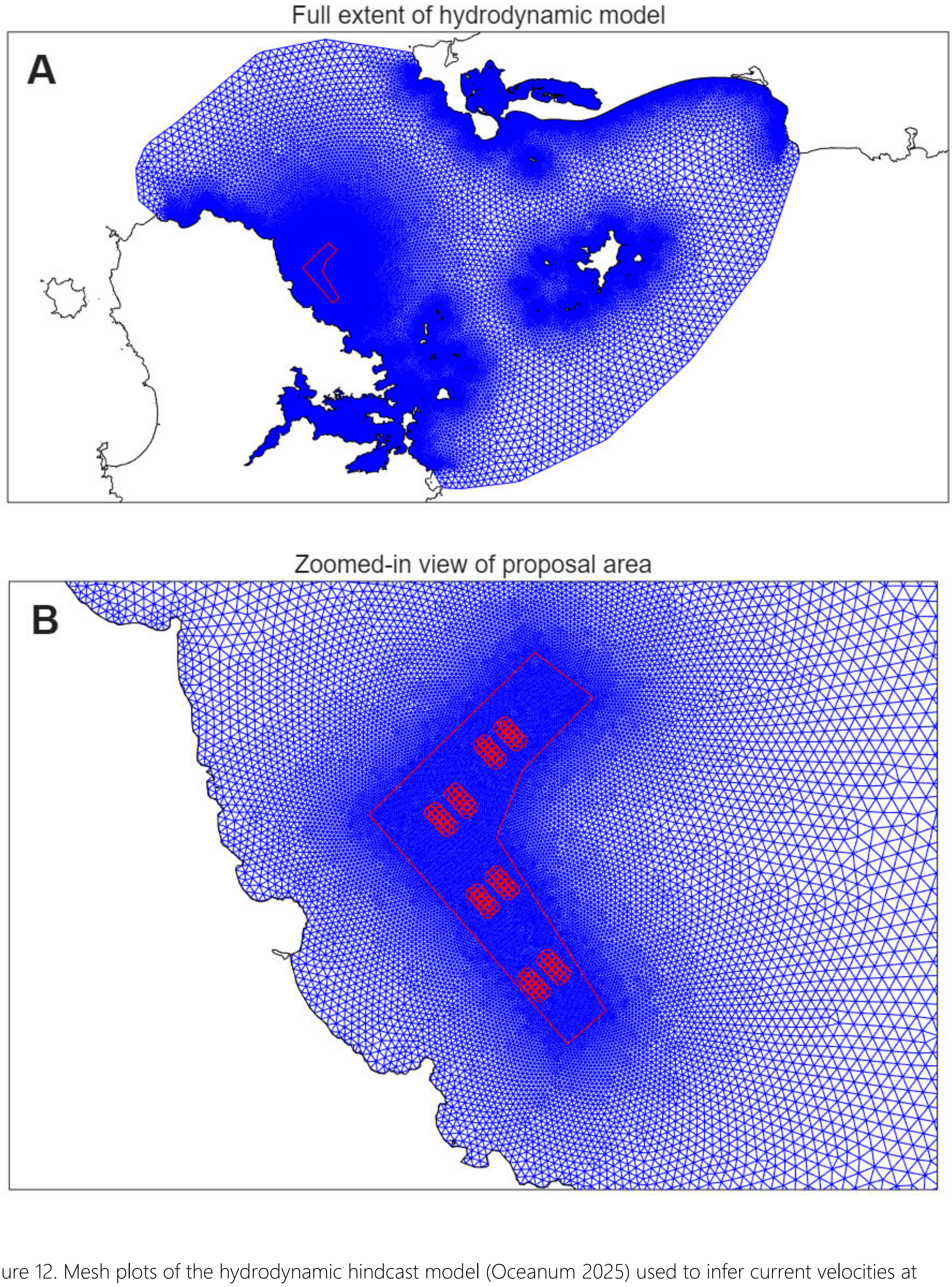

Figure 12. Mesh plots of the hydrodynamic hindcast model (Oceanum 2025) used to infer current velocities at particle locations in the depositional modelling. (A) The full extent of the hydrodynamic model domain. (B) A zoomed-in view of the proposal area. The perimeter of the proposal area and pen locations are denoted in red.

Hydrodynamic forcing at the site

Hydrodynamics at the site are dominated by tides and then weather systems. Differences in tidal elevation between the ends of Foveaux Strait generate strong tidal currents along the NW-SE axis with a predictable semi-diurnal oscillation. These tidal flows typically result in a symmetric waste dispersion pattern along the tidal axis around fish pens. Residual currents (non-tidal) created by local winds and regional weather systems cause a net drift underneath the back-and-forth motion of the tidal currents. This net drift is transient, changing in direction and magnitude based on weather conditions. Oceanum (2025) estimate the magnitude of residual flow components to be 10% of the tidal components.

Depositional modelling here uses a hydrodynamic model for Foveaux Strait to infer water currents at particle locations when calculating particle trajectories. The boundary conditions of this Foveaux Strait model were inherited from a larger hydrodynamic model for New Zealand and therefore include both tidal and weather-related forcing on water currents. The impacts of storm surges (i.e. changes in sea level due to differences in atmospheric pressure) on water currents are included in the New Zealand model and are therefore inherently built into the Foveaux Strait hydrodynamic model and the depositional modelling.

Effects of waves on deposition

The effects of waves on farm waste deposition have not been considered here and to the best of our knowledge have not been incorporated into other depositional modelling studies to date. There are no established and validated methods for including waves in depositional models and including them in our analysis would have required us to develop new methods based on multiple assumptions that introduce new sources of error into our predictions. Given these uncertainties we chose to exclude waves from our analysis but provide an estimate of how waves may impact on particle resuspension.

Wave-induced orbital motion would have a negligible effect on particle trajectories in the sense that the circular motion would return particles to their original location. That said, wave orbitals can increase resuspension and promote further dispersion of particles by tidal currents. Using wave model data for the same period as our hydrodynamic model, we estimate that waves would result in an 8% increase in the percentage of time that particles are resuspended. High-energy wave events would likely resuspend particles throughout the entirety of the site. Further details on how waves resuspend particles and the impact this has on different parts of the proposal area are provided in Appendix 8. Strong tidal currents at the site are by far the most important mechanism of both waste transport and resuspension, and occur at least four times every day.

Modelled feed scenarios

For each model scenario, the amount of feed going through each farm varies monthly according to a feed schedule. This schedule was provided by Ngāi Tahu Seafood, but was simplified to constrain the computational effort required for the simulation. For each farm in the model, all pens in a block start receiving feed at the same time, and the monthly total feed for each farm is distributed evenly across all pens within that farm.

Ngāi Tahu Seafood intends to stagger stocking windows for blocks of pens over time, with pens 'coming online' consecutively rather than all at once. At the start of Stage 1, farms are stocked consecutively, with 6-month gaps between stocking (e.g. Farm 1 is farmed for 6 months before Farm 2 is stocked). We exclude this start-up period from our analysis and model the farms only once all are operating fully.

The modelled feed scenarios (Figure 13) were designed to capture the 3-month period surrounding peak feed inputs to each farm, while also allowing for a 3-month model spin-up period at the start of the simulation. This spin-up period is important because there are no particles in the simulation when the model starts and there is no mass on the seabed. As virtual particles are released from the pens, the total mass in the simulation increases until the amount of mass going into the system is balanced by the amount of mass decaying. At this point, the effect of starting the model with no mass in the system no longer influences the concentrations of solid waste in the model. For Stage 1, we were able to capture the 3-month window around each farm peaking, as well as allow for a 3-month spin-up period in one 24-month model simulation. For Stage 2, we captured six of the eight feed peaks in one 24-month period, and then modelled an additional 24-month period to capture the remaining two peaks. In total, we undertook three model scenarios: one scenario for Stage 1, and two scenarios for Stage 2 (Figure 13).

It is difficult to correctly allocate monthly feed inputs to individual farms over time that sum to the annual feed limits for each stage. In response to this, we slightly overestimated the amount of feed going through the farms in our model simulations and thus slightly overestimated the amount of waste released from the farms. The proposed feed limits for Stage 1 and Stage 2 are 15 $kt{\cdot}yr^{-1}$ and 25 $kt{\cdot}yr^{-1}$, respectively. In our model scenarios, the feed inputs for Stage 1 and Stage 2 are 16 $kt{\cdot}yr^{-1}$ (7% overestimate) and 27 $kt{\cdot}yr^{-1}$ (8% overestimate), respectively. This adds an additional layer of conservatism on top of the other measures taken in our modelling approach to safeguard against uncertainty (Table 11).

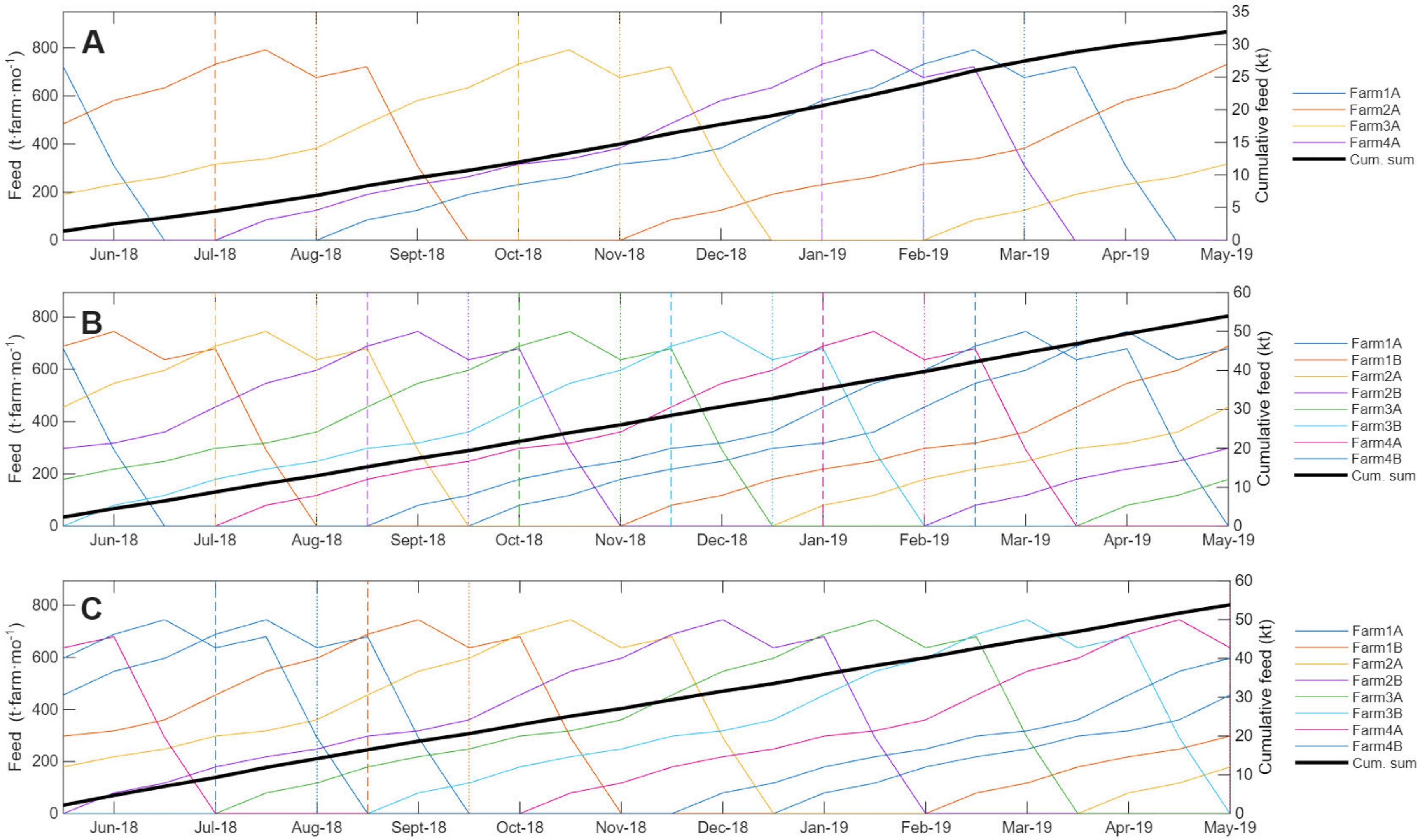

Figure 13. Modelled feed scenarios for Stage 1 (A) and Stage 2 (B and C). The start and end of the 3-month peak feed periods used to calculate depositional footprints for each farm are indicated by coloured dashed and dotted lines, respectively. The cumulative sum of feed across all farms is indicated by the thick black line and secondary axis.

Model outputs and predicting ecological enrichment

Our assessment of benthic effects is based on two measures of deposition: residual solids ($g \cdot m^{-2}$), and monthly solids flux ($g \cdot m^{-2} \cdot mo^{-1}$). Here, we describe these different metrics, explain how and why we have used both metrics, and discuss how these metrics can be related to different levels of ecological enrichment.

Residual solids

To calculate residual solids, we divided the seabed into a grid of 50 m × 50 m squares and summed the mass of solid waste within each grid cell at the end of every model day. This includes waste particles that have fallen from the pens that day, as well as older decaying particles that continue to disperse due to resuspension. For each day of the 24-month simulation, we produced gridded data for the accumulated waste in each 50 m × 50 m cell. The residual solids footprint for each farm was then calculated by averaging these daily residual solids data over the 3 months of peak feed input to each farm. One way to think of residual solids is that it is what you would expect to see visually if you dived below the farm and looked for solid waste.

Solids flux

To calculate monthly solids flux, the seabed was also divided into a grid of 50 m × 50 m squares. However, instead of summing the mass of accumulated waste within each grid cell, only the amount of new mass that had fallen from the pens was considered (no resuspended mass). For every day of the 24-month simulation, we produced gridded data for the daily flux of waste. Any mass that fell the day before (or earlier) was not counted, nor were any resuspended particles (as they were counted in the grid cell they first landed in). The mean daily solids flux footprints for each farm were then calculated by averaging the daily gridded flux data over the 3 months of peak feed input for each farm. Mean daily flux data were then converted into monthly units (mean daily flux × 365 $days \cdot yr^{-1}$ ÷ 12 $mo \cdot yr^{-1}$). One way to think of monthly solids flux is that it is what you would measure with a sediment trap elevated above the seabed during the course of a month (or a year for annual solids flux).

Using solids flux to determine the zone of maximum effect

The residual solids footprint considers both the initial deposition of waste particles from the pens, as well as the secondary transport of waste particles via resuspension. It is a more realistic metric for determining deposition at high-flow sites such as this one. Traditionally, however, finfish farming in Aotearoa New Zealand has been undertaken in relatively low-flow sites, where resuspension is not an important waste transport mechanism. During this time, methods were developed to relate modelled flux values to an ecological ‘enrichment stage’ by correlating modelled flux values to benthic field measurements at salmon farms in the Marlborough Sounds (Keeley et al. 2013). Enrichment stages defined in Keeley et al. (2013) and corresponding flux values are provided below (Table 7) for dispersive sites (mid-water current speeds ≥10 $cm \cdot s^{-1}$, cf. 39–45 $cm \cdot s^{-1}$ in the proposal area).

Table 7. Modelled flux values and corresponding enrichment stage definitions taken from high-flow site examples in Keeley et al. (2013).

Monthly solids flux (annual equivalent)	Enrichment stage	Characteristics
0	Pristine end of spectrum	Environmental variables comparable with an unpolluted, pristine reference station.
< 83 $g \cdot m^{-2} \cdot mo^{-1}$ (< 1 $kg \cdot m^{-2} \cdot yr^{-1}$)	2 – Minor enrichment	Richness usually greater than for reference conditions. Zone of 'enhancement' – minor increases in abundance possible. Mainly a compositional change. Sediment chemistry unaffected or with only very minor effects.
83 $g \cdot m^{-2} \cdot mo^{-1}$ (1 $kg \cdot m^{-2} \cdot yr^{-1}$)	3 – Moderate enrichment	Notable abundance increase; richness and diversity usually lower than at reference station. Opportunistic species (i.e. capitellid worms) begin to dominate,
417 $g \cdot m^{-2} \cdot mo^{-1}$ (5 $kg \cdot m^{-2} \cdot yr^{-1}$)	4 – High enrichment	Diversity further reduced; abundances usually quite high, but clearly sub-peak. Opportunistic species dominate, but other taxa may still persist. Major sediment chemistry changes (approaching hypoxia).
10,833 $g \cdot m^{-2} \cdot mo^{-1}$ (13 $kg \cdot m^{-2} \cdot yr^{-1}$)	5 – Very high enrichment	Very high numbers of one or two opportunistic species (i.e. capitellid worms, nematodes). Richness very low. Major sediment chemistry changes (hypoxia, moderate oxygen stress). Bacteria mat usually evident. Out-gassing occurs on disturbance of sediments.

For the purposes of interpreting depositional modelling outputs at the proposal area, we determine a **zone of maximum effect (ZME) as the area where monthly solids flux is greater than 41.67 $g \cdot m^{-2} \cdot mo^{-1}$ (equivalent annual flux of 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$)**. We have conservatively chosen this value by taking half of the flux value used to classify minor enrichment in Keeley et al. (2013). We have adopted this conservative approach considering the differences in environmental conditions and seabed composition at the proposal site compared to the Marlborough Sounds sites on which Keeley et al. (2013) based their work.

Using residual solids to determine the primary footprint

To relate residual solids values to a level of environmental effect, we rely on a study undertaken retrospectively at the Clay Point and Te Pangu salmon farms in Tory Channel (Elvines et al. 2021b). In this study, depositional modelling was undertaken using an older version of the OceanTracker model (VenOM) and includes the effects of resuspension and particle decay. Historical feed data were used in conjunction with past monitoring data and a historical footprint mapping exercise to identify levels of residual solids at which ecological enrichment was likely. While the resuspension module used here is more sophisticated than Elvines et al. (2021b) because it includes the effects of different habitat on resuspension dynamics, the underlying particle-tracking model is the same. Moderately enriched (see Table 7) soft sediment sites in this study corresponded to residual solids values between 7 $g \cdot m^{-2}$ and 18 $g \cdot m^{-2}$. There were no enriched reef sites in the study area of Elvines et al. (2021b), although modelled residual solids at the reef sites was as high as 9 $g \cdot m^{-2}$. Elvines et al. (2021b) determined residual solids values of 12.5 $g \cdot m^{-2}$ (the midpoint value of 7 $g \cdot m^{-2}$ and 18 $g \cdot m^{-2}$) and 9 $g \cdot m^{-2}$ as suitable thresholds above which enrichment effects are likely in soft sediment and reef habitats, respectively.

In a similar fashion to determining the ZME, we conservatively set a threshold value of **7 g·m^{-2} as our primary footprint** by taking the lower bound of moderately enriched soft sediment sites in Elvines et al. (2021b). By taking the lower bound of this range, our footprint also includes the 9 g·m^{-2} threshold proposed for reef habitats. Finally, we set our **outer limit of effects (OLE) footprint as 0.7 g·m^{-2}**, based on an order of magnitude reduction in deposition from our primary footprint. The OLE footprint is intended to represent the maximum spatial extent at which farm-related effects might be detectable.

Predicted depositional footprint

Stage 1

The modelling outputs are presented in Figures 14–17 and represent solids flux (kg·m^{-2}·yr^{-1}) and residual solids deposition (g·m^{-2}) averaged over the 3 months of peak feed discharge for each farm (i.e. in the lead-up to harvest). Under the single year-class farming regime proposed by Ngāi Tahu Seafood, these peak inputs will be followed by periods of low to no deposition, with feed declining to zero after harvest and sites left fallow for at least 3 months before the next cycle begins.

Within the ZME (defined by the 0.5 kg·m^{-2}·yr^{-1} solids flux footprint), no areas are predicted to receive solids flux above 13 kg·m^{-2}·yr^{-1} (very high enrichment) during the 3 months of peak production at Stage 1. It is predicted that 10–19 ha of sandy seabed (0.8–1.5% of the proposal area) could receive a high level of deposition of organic waste (deposition greater than 5 kg solids·m^{-2}·yr^{-1}) during periods of peak feed input at each farm, extending up to 150 m from the pen edges (Figure 14, Table 8). The total ZME is predicted to extend up to 680 m from the pen edges across sandy seabed habitats.

Across the primary footprint (defined by the 7 g·m^{-2} residual solids footprint, and including the ZME), 510–533 ha of the seabed (40–42% of the proposal area) could receive a moderate level of deposition at Stage 1 per farm (Figure 15, Table 8). The primary footprint is predicted to mostly remain within the proposal area boundary and does not overlap with known areas of ecologically valuable and sensitive marine habitats.

Outside the primary footprint, resuspension modelling indicates that trace amounts of farm-derived organic material may disperse more than 17 km northwest and southeast of the proposal area boundary (beyond the model domain; Figure 15). Much of this wider dispersal zone is predicted to have very low residual solids, well below levels expected to cause ecological effects. The OLE boundary, approximated using the 0.7 g·m^{-2} residual solids contour (Figure 15), is indicative only; it is an order of magnitude lower than thresholds associated with moderate enrichment of soft sediment sites elsewhere and is unlikely to result in measurable ecological change in reef sites. Under this conservative representation, a further 958–1,262 ha of mostly sandy seabed habitats could receive minor enrichment during periods of peak production at each farm and there is minimal overlap with biogenic habitats at Stage 1 (Figure 15):

- < 2.7 ha of bryozoan-sponge reef (0.3% of the mapped extent)
- < 5 ha of bushy-bryozoan thickets (1% of the mapped extent)
- 0.7 ha of patchy low-relief bryozoan-sponge habitat (0.1% of the mapped extent).

Overall, these overlaps are small, and residual solids are predicted to remain well below levels expected to cause adverse effects.

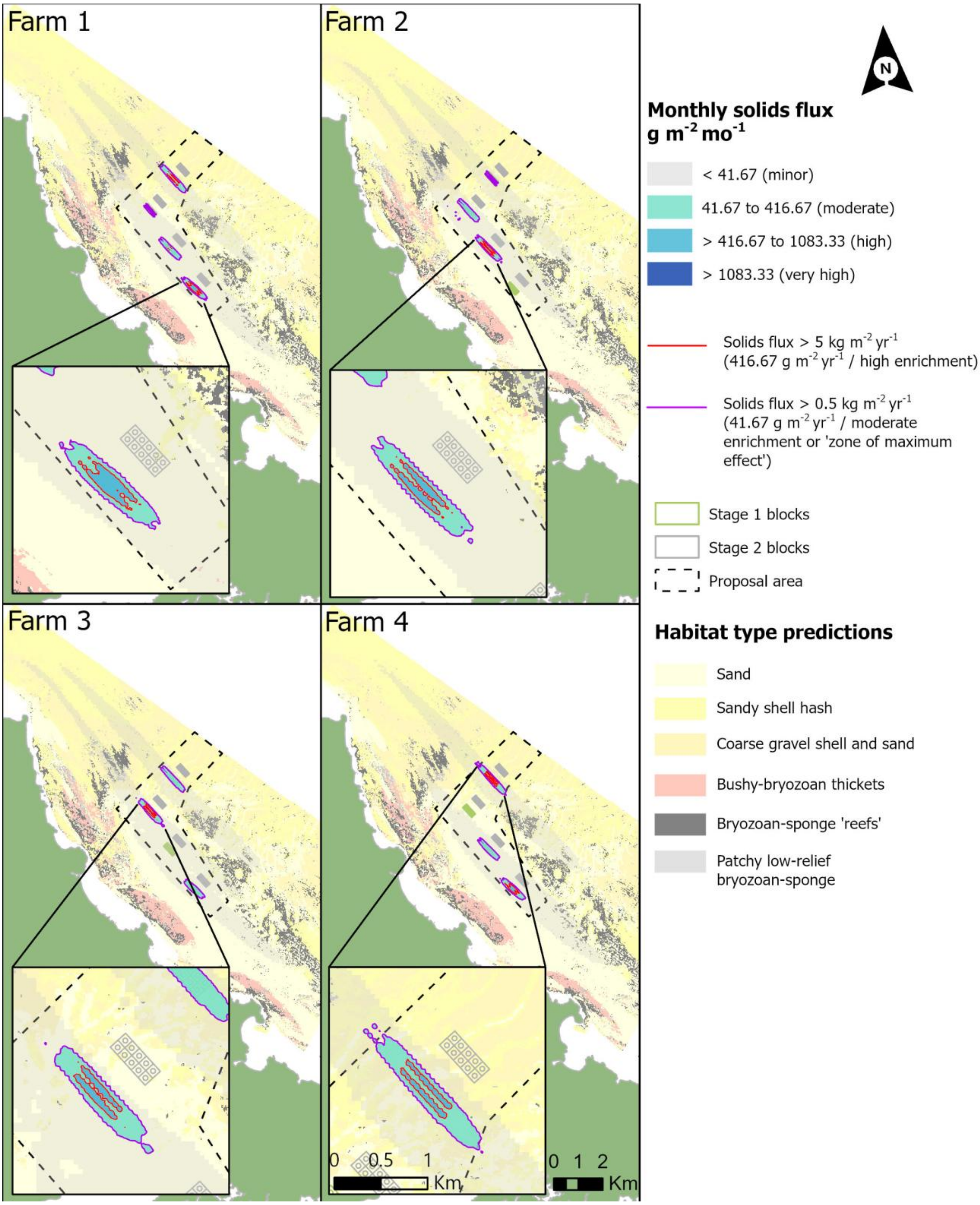

Figure 14. Predicted depositional footprint of farm-derived solids ($g \cdot m^{-2} \cdot mo^{-1}$ and equivalent $kg \cdot m^{-2} \cdot yr^{-1}$ footprints) for Stage 1 of the Hananui proposal (~ 15,000 tonnes feed per year), averaged over the 3 months of the production cycle with the highest feed discharge, without accounting for resuspension. The conservative 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ contour defines the zone of maximum effect (ZME), while areas of high enrichment (≥ 5 $kg \cdot m^{-2} \cdot yr^{-1}$) within the ZME are indicated for each farm. Total areas within the ZME footprints are summarised in Table 8.

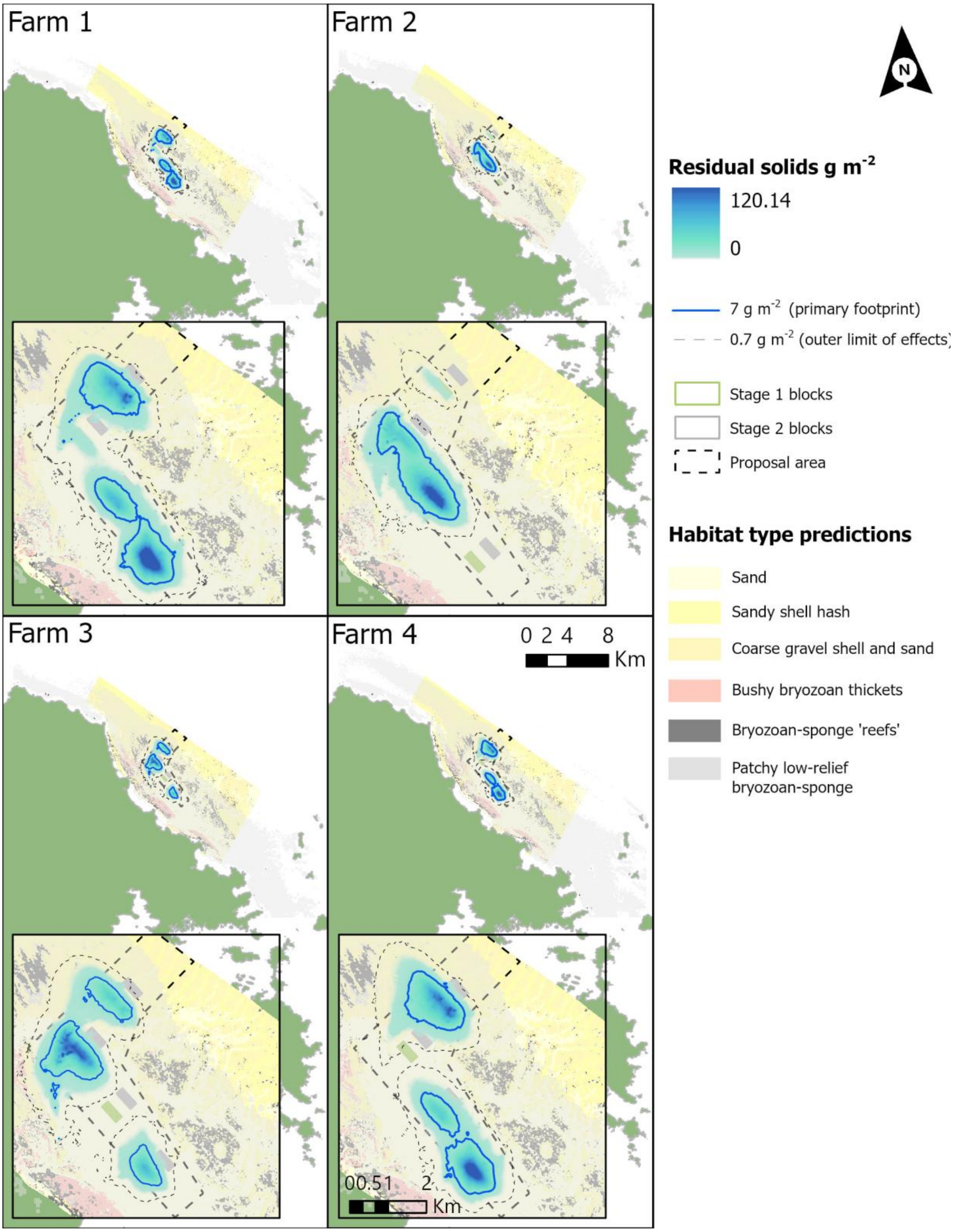

Figure 15. Predicted residual solids (g·m^{-2}) for Stage 1 of the Hananui proposal, averaged over the 3 months of the production cycle with the highest feed discharge, after accounting for resuspension and decay processes. The primary footprint, defined by the 7 g·m^{-2} residual solids contour, indicates areas of potential moderate enrichment. The outer limit of effects (OLE), approximated using the 0.7 g·m^{-2} residual solids contour, is indicative only; it represents an order of magnitude lower than thresholds associated with moderate enrichment in the Marlborough Sounds. Total areas within the primary and OLE footprints are summarised in Table 8.

Stage 2
A similar pattern of waste deposition as that observed for Stage 1 is predicted for Stage 2.

Within the ZME (defined by the 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ solids flux footprint), no areas are predicted to receive solids flux above 13 $kg \cdot m^{-2} \cdot yr^{-1}$ (very high enrichment) during peak production at Stage 2. It is predicted that 10–19 ha of sandy seabed (0.8–1.5% of the proposal area) could receive a high level of deposition of organic waste (deposition greater than 5 kg $solids \cdot m^{-2} \cdot yr^{-1}$) during periods of peak feed input for each farm block, extending up to 250 m from the pen edges (Figure 16, Table 8). The total ZME is predicted to extend up to 680 m from the pen edges, across sandy seabed habitats.

Across the primary footprint (defined by the 7 $g \cdot m^{-2}$ residual solids footprint, and including the ZME), 473–533 ha of the seabed (37–42% of the proposal area) could receive a moderate level of deposition at Stage 2 during peak production for each farm block (Figure 17, Table 8). The primary footprint is predicted to mostly remain within the proposal area boundary and does not overlap with known areas of ecologically valuable and sensitive marine habitats.

Outside the primary footprint, resuspension modelling indicates that trace levels of farm-derived organic material may disperse more than 17 km northwest and southeast of the proposal area boundary (beyond the model domain). As for Stage 1, much of this wider dispersal zone is predicted to have very low residual solids, well below levels expected to cause ecological effects. The OLE boundary, approximated using the 0.7 $g \cdot m^{-2}$ residual solids contour, indicates that a further 868–1,195 ha of mostly sandy seabed habitats could receive minor enrichment while each farm block is at peak production. There is some overlap of this footprint (where residual solids are predicted to be less than ~ 2 $g \cdot m^{-2}$) with biogenic habitat at Stage 2 (Figure 17):

- < 43 ha of bryozoan-sponge reef (5% of the mapped extent)
- < 16 ha of bushy-bryozoan thickets (3% of the mapped extent)
- < 25 ha of patchy low-relief bryozoan-sponge habitat (17% of the mapped extent).

These values represent the maximum predicted overlap under conservative assumptions for the far-field dispersal of residual solids and are, overall, relatively small and below levels expected to cause adverse effects.

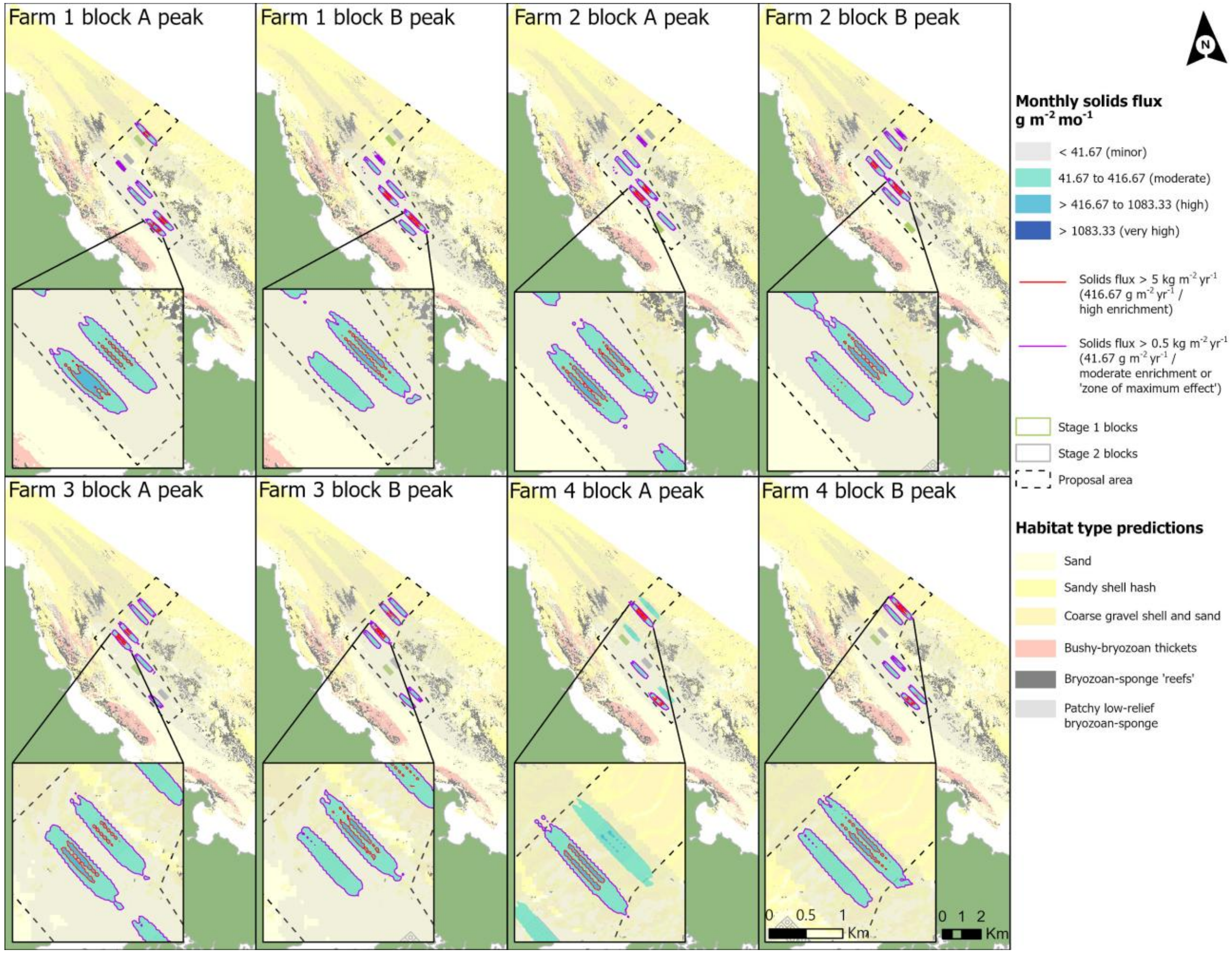

Figure 16. Predicted depositional footprint of farm-derived solids ($g{\cdot}m^{-2}{\cdot}mo^{-1}$ and equivalent $kg{\cdot}m^{-2}{\cdot}year^{-1}$ footprints) for Stage 2 of the Hananui proposal (~ 25,000 tonnes feed per year), averaged over the 3 months of the production cycle with the highest feed discharge, without accounting for resuspension. The conservative 0.5 $kg{\cdot}m^{-2}{\cdot}yr^{-1}$ contour defines the zone of maximum effect (ZME), while areas of high enrichment (≥ 5 $kg{\cdot}m^{-2}{\cdot}yr^{-1}$) within the ZME are indicated for each farm. Total areas within the ZME footprints are summarised in Table 8.

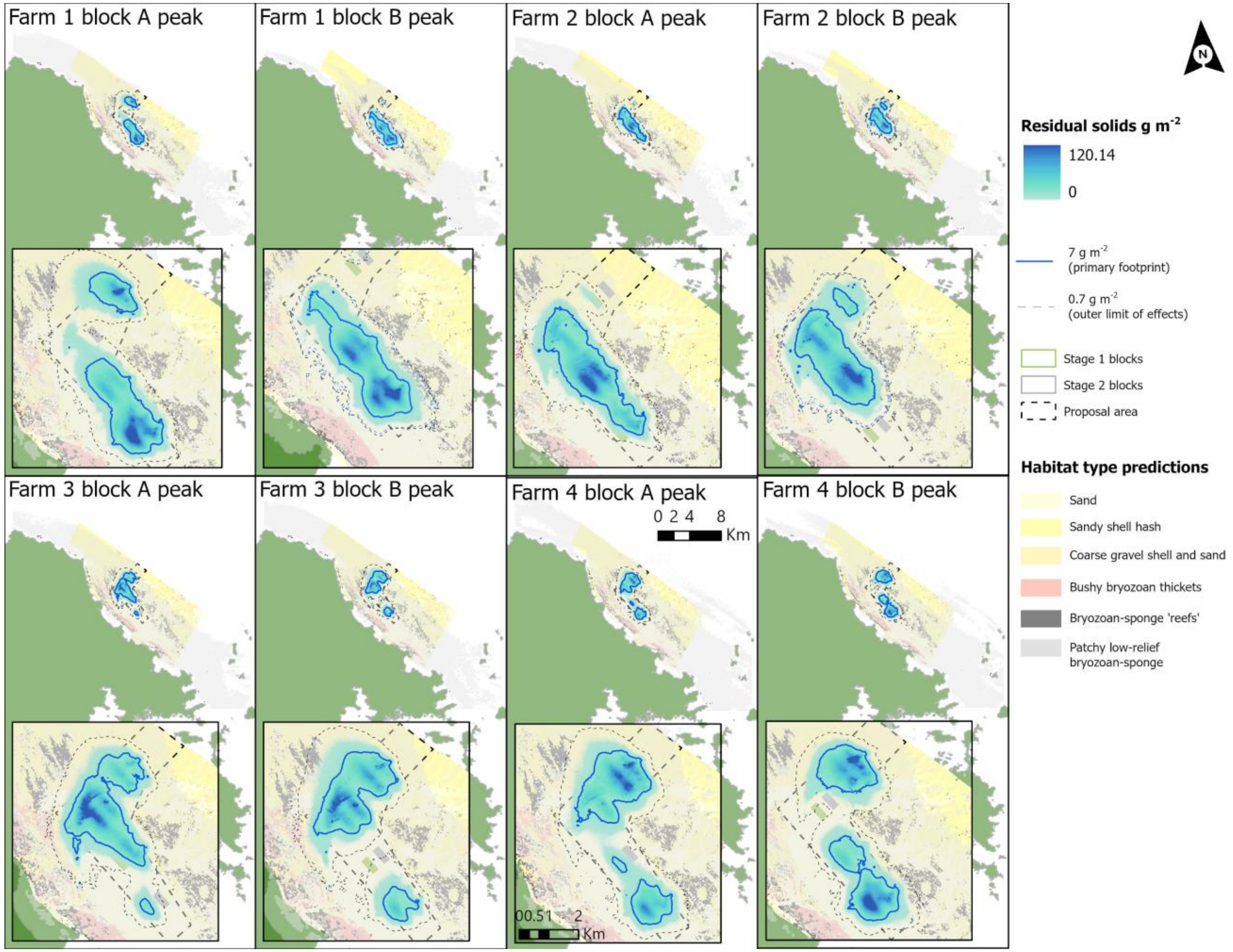

Figure 17. Predicted residual solids (g·m^{-2}) for Stage 2 of the Hananui proposal, averaged over the 3 months of the production cycle with the highest feed discharge, after accounting for resuspension and decay processes. The primary footprint, defined by the 7 g·m^{-2} residual solids contour, indicates areas of potential moderate enrichment. The outer limit of effects (OLE), approximated using the 0.7 g·m^{-2} residual solids contour, is indicative only; it represents an order of magnitude lower than thresholds associated with moderate enrichment in the Marlborough Sounds. Total areas within the primary and OLE footprints are summarised in Table 8.

Table 8. Predicted footprint areas (ha) at Stage 1 and Stage 2 of development for each farm at peak production (average over the 3 months of highest feed input). Footprints are reported as: the zone of maximum effect (ZME; solids flux ≥ 0.5 kg $m^{-2} \cdot yr^{-1}$), the primary footprint (residual solids ≥ 7 g m^{-2}, including the ZME), and the outer limit of effects (OLE) footprint (residual solids ≥ 0.7 g m^{-2} but < 7 g m^{-2}, i.e. not including the primary footprint). The ZME is further broken down into areas predicted to receive solids flux levels corresponding to moderate enrichment (0.5–5 kg $m^{-2} \cdot yr^{-1}$) and high enrichment (> 5–13 kg $m^{-2} \cdot yr^{-1}$), based on thresholds for high-flow (dispersive) sites (Keeley et al. 2013; Fletcher et al. 2022).[7]

		Footprint area (ha)				
		ZME			Primary footprint	OLE
		ZME total	ZME moderate enrichment	ZME high enrichment		
Stage 1	**Farm 1**	140	121	19	518	1,262
	Farm 2	139	126	13	510	958
	Farm 3	151	141	10	533	1,036
	Farm 4	155	145	11	530	1,215
Stage 2	**Farm 1 A block**	140	121	19	518	1,262
	Farm 1 B block	135	123	12	473	917
	Farm 2 A block	139	126	13	510	958
	Farm 2 B block	142	127	15	487	868
	Farm 3 A block	151	141	10	533	1,036
	Farm 3 B block	150	137	13	523	1,195
	Farm 4 A block	155	145	11	530	1,215
	Farm 4 B block	151	132	20	518	1,080

[7] These relationships were developed based on farming multiple year classes where feed inputs are relatively constant. In contrast, farms in the Hananui proposal will hold a single year class at any given time, so periods of high feed discharge will be followed by periods of low to no feed input. Additionally, the sandy seabed of the proposal area differs from that for which these relationships were derived (i.e. sandy-mud and muddy-sand habitats in the Marlborough Sounds).

Indirect seabed deposition from water column enrichment
In addition to direct deposition of feed and faeces, salmon farming can increase phytoplankton productivity via elevated nitrogen in the water column. Dead phytoplankton or waste from grazers may contribute additional organic material to the seabed. However, water quality modelling (Wilson 2025) indicates that predicted increases in total nitrogen concentrations are small and unlikely to result in notable increases in phytoplankton biomass. The strong tidal currents in Foveaux Strait limit the potential for bloom development by rapidly dispersing nutrients, preventing phytoplankton cells from remaining in enriched waters long enough to metabolise and reproduce (Wilson 2025). While some organic material from dead phytoplankton or grazers may still reach the seabed, the contribution from water column enrichment is expected to be minor compared to direct deposition. Seabed monitoring is expected to capture any measurable effects, ensuring that indirect deposition from water column enrichment is detected as production scales up.

Predicted effects of the proposed development on the seabed

During periods of peak feed input, the total seabed area where moderate or higher enrichment is expected (ZME) ranges from 135 ha to 155 ha (Table 8). Within this, the area where high enrichment is expected ranges from 10 ha to 20 ha (up to 1.6% of the proposal area), and the area of moderate enrichment ranges from 121 ha to 145 ha (up to 11% of the proposal area). It is important to note that during periods of peak feed input to individual farm blocks, other blocks within the proposal area are simultaneously operating at various production stages (Figures 15 and 17). Accordingly, deposition areas reported in Table 8 represent cumulative input from all active farms at that time, not just from the block receiving peak feed.

Within the primary footprint (defined by the 7 $g{\cdot}m^{-2}$ residual solids footprint), up to 533 ha (42% of the proposal area) could receive a moderate level of deposition during periods of peak feed input for each farm block (Table 8). Note that there is overlap between the primary footprint and the ZME. Outside of the primary footprint, low levels of farm-related organic waste may be dispersed more than 17 km northwest and southeast of the proposal area boundary. The OLE footprint (defined by the 0.7 $g{\cdot}m^{-2}$ residual solids footprint) indicates the maximum spatial extent at which farm-related effects might be detectable; within this footprint, minor enrichment is possible. Outside of the OLE, farm waste might be detectable, but effects are unlikely to be discernible. At Stage 1 and Stage 2, up to a further 1,262 ha could receive 0.7 $g{\cdot}m^{-2}$ residual solids during periods of peak production at each farm (and farm block). Beyond that (where residual solids are < 0.7 $g{\cdot}m^{-2}$), farm-related effects are unlikely.

To assess the potential effects of the predicted deposition levels on the seabed within and beyond the proposal area, enrichment classifications based on observations from dispersive sites in the Marlborough Sounds are applied (Fletcher et al. 2022).

Sediment properties
Seabed conditions directly beneath the pens (i.e. within the middle of the ZME footprints) are expected to undergo major changes in sediment chemistry (Fletcher et al. 2022), including substantially elevated total free sulphides and reduced redox potential as a result of increased microbial activity from waste

decay (Hamoutene 2014). Sediment organic content is also expected to increase and patches of bacteria may be visible. Out-gassing (of methane and hydrogen sulphide) is considered unlikely.

Beyond the ZME, sediment chemistry may exhibit moderate changes relative to baseline and reference conditions (Fletcher et al. 2022), including elevated sulphide concentrations and reduced redox potential across the primary footprint. While some increase in organic content is possible, these effects are expected to be less pronounced and more spatially variable than those observed directly beneath the pens. This is particularly true at a high-flow site like this, where organic material is likely to be readily metabolised, dispersed, or exported when present in dilute form, and typically assimilated without measurable consequences (Keeley et al. 2019; Keeley et al. 2024).

Outside the primary footprint and the Hananui proposal area, minor enrichment is possible. However, sediment chemistry is expected to show only very minor effects or remain unaffected (Fletcher et al. 2022).

It is important to reiterate that differences exist between the proposal area and the muddier sediments in the Marlborough Sounds from which these relationships were derived. It is possible that the sandy sediments across the predicted footprint are less cohesive and therefore more easily oxygenated, so the same level of waste build-up and response may not be observed. Furthermore, under the single year-class farming regime proposed by Ngāi Tahu Seafood, periods of higher deposition will be followed by periods of low to no deposition, allowing for some seabed recovery between production cycles.

Finally, as noted earlier, troughs and flanks of subtidal dune and ridge features within the proposal area may act as temporary retention zones for organic material. Depositional modelling indicates that while waste can accumulate in the troughs of these features, it is periodically flushed by strong tidal currents. This reflects a dynamic in which temporary accumulation may occur along flanks or in troughs, but regular tidal flushing helps prevent the formation of significant accumulation zones. Furthermore, while the modelling does not indicate increased retention of waste in the more consolidated coarse gravel, sand and shell substrates of the northeastern section of the proposal area, monitoring across all areas will be important to confirm that environmental effects align with predictions.

Sediment macrofauna

Within the middle of the ZME, where high levels of enrichment are possible, substantial shifts in sediment macrofaunal community composition are expected. Under such conditions, overall abundance is likely to be elevated, with opportunistic taxa (e.g. capitellid worms) becoming dominant, although other taxa may still persist (Fletcher et al. 2022). During other stages of the production cycle, when feed discharges are low or absent, macrofaunal communities in these areas would be expected to show reduced effects and some potential for recovery.

Across the primary footprint, where moderate enrichment is predicted, macrofaunal abundance is also likely to be elevated, but species richness and diversity may be reduced compared with baseline and reference conditions (Fletcher et al. 2022). Opportunistic and tolerant species (e.g. capitellids, dorvilleids) are expected to increasingly dominate communities under these conditions.

Outside of the Hananui proposal area, where only minor enrichment is predicted, macrofaunal communities may be 'enhanced' with slightly higher richness and abundance compared to reference conditions. It is important to note that this enhancement effect may also occur within the primary footprint, particularly during the early stages of development, as organic inputs stimulate benthic productivity.

As discussed earlier, macrofaunal communities across the proposal area may have a limited inherent capacity for waste assimilation, reflecting their naturally low abundance and diversity. Nevertheless, the presence of deposit-feeding taxa provides some capacity to recycle organic material. These species are likely to increase under farm conditions and contribute to organic matter processing.

Epibiota

The substratum within the proposal area where ZME and primary depositional footprints overlap is sand with varying amounts of shell hash. Epibiota are patchy within this habitat type. Beyond the proposal area, patches of high biogenic cover occur, supporting sensitive species. The depositional models indicate that low levels of farm-related organic waste may be dispersed up to 17 km from the proposal area boundary. However, effects are unlikely to be discernible beyond the OLE footprint (where minor enrichment may occur, but adverse ecological effects are unlikely). The OLE footprint is mostly constrained to sandy habitats with varying amounts of shell hash and gravel. At Stage 1, the outer edge of this footprint (where residual solids are ~ 0.7 $g \cdot m^{-2}$) overlaps slightly with < ~ 1.4% of the total mapped biogenic habitat (< 8.4 ha) per farm block during peak feed discharge. At Stage 2, parts of the lower end of this footprint (i.e. where residual solids are < ~ 2 $g \cdot m^{-2}$) are predicted to overlap with < 5% (43 ha) of the total mapped bryozoan-sponge reef, 17% (25 ha) of the total mapped patchy low-relief bryozoan-sponge habitat, and 3% (16 ha) of the total mapped bushy-bryozoan thickets. Beyond this zone, residual solids are predicted to fall below 0.7 $g \cdot m^{-2}$, where farm-related effects are unlikely.

Effects on epibiota within the ZME and primary depositional footprints

Farm placement is such that the ZME and primary depositional footprints are not predicted to overlap with ecologically valuable and sensitive seabed habitats. However, where potentially sensitive epibiota (such as bryozoans, sponges, tube worms, large bivalves and brachiopods) occur occasionally (and at very low densities) within predicted footprints, they may be affected by increased sedimentation associated with farm-related deposition. This could reduce suspension and filter-feeding efficiencies, and in some cases direct smothering may displace individual organisms. Long-term exposure to elevated organic inputs may also lead to sub-lethal stress responses. For example, reduced food quality from the consumption of salmon feed particulates may impair growth and reproductive potential (White et al. 2016), contributing to lower population densities. Horse mussels have demonstrated sensitivity to high enrichment levels, showing increased respiration, altered fatty acid profiles, stress-related gene expression and digestive gland atrophy (Elvines et al. 2024; McMullin et al. 2025). These effects could translate into reduced energy reserves, impaired digestion and diminished reproductive success. Other species, such as scallops and brachiopods, may tolerate lower levels of enrichment. However, increasing variability in individual responses at higher deposition levels suggests that these taxa may approach their tolerance limits under sustained exposure at levels of deposition predicted within the ZME. Additionally, localised increases in epibiota abundance in response to enhanced food availability may attract scavengers and predators (e.g. sea cucumbers, crabs, cushion stars, snake stars). Such aggregations may compete for settlement space or prey on sensitive sessile species, particularly juveniles, further reducing recruitment potential and altering community dynamics.

Effects on epibiota within the OLE zone

The OLE footprint is intended to represent the maximum spatial extent at which farm-related effects might be detectable. The upper boundary of this zone is 7 $g{\cdot}m^{-2}$, equivalent to the lower bound of moderate enrichment effects in the Marlborough Sounds and 2 $g{\cdot}m^{-2}$ below the 9 $g{\cdot}m^{-2}$ threshold predicted to reach rocky reefs in Tory Channel where no effects have been observed (Elvines et al. 2021b). The lower boundary is less certain and has been conservatively defined, based on professional judgement, at approximately an order of magnitude lower than levels where moderate enrichment effects are observed in the Marlborough Sounds (i.e. 0.7 $g{\cdot}m^{-2}$).

This zone is predicted to overlap mainly sandy seabed habitats with varying levels of shell hash and gravel. At Stage 1, during periods of peak feed discharge, the outermost edge of the OLE footprint, where residual solids are predicted to be ~ 0.7 $g{\cdot}m^{-2}$, may overlap with:

- < 0.3% (2.7 ha) of the total mapped bryozoan-sponge reef
- 0.1% (0.7 ha) of the total mapped patchy low-relief bryozoan-sponge habitat
- 1% (5 ha) of the total mapped bushy-bryozoan thickets

At Stage 2, the footprint extends slightly further, and the outer swath, where residual solids range from ~ 2 $g{\cdot}m^{-2}$ down to 0.7 $g{\cdot}m^{-2}$, may overlap with:

- < 5% (43 ha) of the total mapped bryozoan-sponge reef
- 17% (25 ha) of the total mapped patchy low-relief bryozoan-sponge habitat, and
- 3% (16 ha) of the total mapped bushy-bryozoan thickets

These taxa are suspension feeders, and slight increases in organic particulates may provide an enhanced food supply. Many are likely to tolerate, and in some cases benefit from, such conditions. For example, tube worm 'reefs' near salmon farms in Big Glory Bay host diverse epibiota (Smith et al. 2005). In subtropical coral communities, intermediate nutrient enrichment has been linked with higher diversity, even though high impacts reduce it (Huang et al. 2011). Bryozoans have also been shown to increase in diversity in moderately enriched environments (Koçak 2008).

Effects on epibiota beyond the OLE boundary

Beyond the OLE boundary are extensive areas of biogenic habitat, including bryozoan-sponge reefs and bushy-bryozoan thickets. Residual solids in these areas are predicted to be < 0.7 $g{\cdot}m^{-2}$, a level at which farm-related effects are considered unlikely. By the time waste reaches this distance, it is expected to be sufficiently diluted and readily assimilated, making any enrichment signal difficult to distinguish from natural variability (Keeley et al. 2024).

The uneven and rough nature of these reef habitats has been represented in the depositional modelling through the adoption of variable resuspension thresholds. Specifically, the threshold applied to bryozoan-sponge reef habitat was based on a value determined for fragmented rock, which shares a similar structural scale to the reef features observed near the site. However, particles could still become more trapped than modelled, particularly if they become wedged in cracks or gaps between reef structures. Modelling indicates that deposition levels in these reef habitats remain low (<0.7 $g{\cdot}m^{-2}$), and it is considered unlikely that any localised pockets of deposition would result in meaningful enrichment or adverse effects on epibiotic communities.

Risk to biogenic habitat: what we know

The 7 $g{\cdot}m^{-2}$ contour (below the 9 $g{\cdot}m^{-2}$ threshold at which no effects have been observed on rocky reefs in Tory Channel) is predicted to remain largely within the proposal area and does not overlap with extensive areas of biogenic habitat. During Stage 1, residual solids of ~ 0.7 $g{\cdot}m^{-2}$ are predicted to only just reach the fringes of the nearest areas of biogenic habitat at peak production. At Stage 2, residual solids of ~ 2 $g{\cdot}m^{-2}$ are expected to extend to the habitat edges, with the remaining area of OLE overlap limited to very low levels (< 2 $g{\cdot}m^{-2}$), all well below the Tory Channel reef threshold.

While there are differences between Tory Channel reefs and the bryozoan-sponge reefs outside of the Hananui proposal area (e.g. community composition and substrate), both are dispersive environments that support many of the same taxa (e.g. bryozoans, sponges, tube worms, anemones, ascidians, sea stars, oysters, scallops) and functional groups (e.g. suspension-feeding bivalves, sessile suspension feeders, mobile deposit and suspension feeders, mobile scavengers / predators; McMullin and McGrath 2023). Tory Channel therefore provides a useful point of reference, suggesting that biogenic habitats outside of the proposal area are unlikely to be exposed to deposition levels that would cause stress. In practice, deposition levels reaching the areas of biogenic habitat may be too low to generate a detectable ecological response. Consistent with this, monitoring near salmon farms in Storm Bay, Tasmania, has also found no evidence of system-wide effects on adjacent biogenic habitats, including inshore reef, deep reef and seagrass communities (Aquenal 2024).

Current knowledge indicates that responses of epibiota to salmon farm-derived deposition are species-specific. Tube worms, brachiopods and large bivalves generally appear tolerant to low-level deposition, exhibiting only subtle or sub-lethal effects in laboratory and field studies at higher levels of waste exposure. Bryozoans and sponges show more variable sensitivity, whereby slow-growing, frame-building bryozoans and some dense or slow-growing sponges are likely more vulnerable, whereas faster-growing or opportunistic bryozoans and certain sponge species can tolerate, or even benefit from, low-level enrichment. However, it is currently unknown how reef-building bryozoans will respond to any level of farm-derived deposition, and no direct evidence exists for these taxa in comparable environments.

International studies highlight this variability. Epiphytic bryozoans increased in biomass under elevated organic inputs (Haugland et al. 2021), and other bryozoan species settled and grew on artificial structures near farms in the Gulf of Aqaba, Red Sea, and in Iceland (Angel et al. 2002; Israel et al. 2016). Declines in soft corals and some sponges have been observed near farms, but these taxa were common again within 200 m, at flux levels far higher than those predicted outside the Hananui proposal area (Dunlop et al. 2021). Conversely, maerl beds (formed by coralline red algae) are highly sensitive even to low deposition (Hall-Spencer et al. 2006).

Overall, while species-specific responses and some uncertainties remain, particularly for reef-building bryozoans, the evidence suggests that most suspension-feeding taxa within the biogenic habitats outside of the proposal area are likely to tolerate the low levels of deposition predicted to reach those areas during periods of peak feed input. Given these low levels, any influence on food availability or community composition in these areas is also expected to be minor. Consequently, adverse effects on biogenic habitats are considered unlikely, although monitoring remains important to detect any subtle or cumulative changes in community structure over time.

Protection provided by the proposal area

While the Hananui proposal area partially overlaps with the southwestern extent of the Foveaux Strait dredge oyster fishery area (Hill et al. 2010), it lies predominantly outside the main oyster beds and over the last decade only a low distribution of catch has come from within the proposal area (Michael 2019). Dredging still occurs within this area. Bryozoans, sponges, brachiopods and large bivalve species are all highly sensitive to the effects of dredging (Anderson et al. 2019), with benthic fishing activity already causing significant loss to bryozoan thickets in this region (Michael 2007). Therefore, although far-field effects associated with organic deposition from farm operations are possible, farm infrastructure and low oyster densities may deter dredging within the Hananui proposal area.

5.5 Seabed effects resulting from farm additives

In addition to organic enrichment effects, there is potential for seabed impacts associated with additives in feed (e.g. zinc), antifoulants (typically copper-based) and therapeutants used to treat stock (e.g. antibiotics, parasiticides, anaesthetics). Each is summarised below, with potential environmental hazards and options to avoid, remedy or mitigate adverse effects. A summary of potential environmental effects arising from farm additives during farm operation and options to avoid, remedy or mitigate these, where applicable, is provided in Table 9 .

Metals (copper and zinc)

Historically, copper contamination has arisen from antifouling paints applied to nets and other farm structures. However, antifouling paints are now used far less frequently in finfish aquaculture and are not proposed for nets at this site (Thomas Hildebrand, Ngāi Tahu Seafood, pers. comm., 2025). Zinc is routinely included in fish feeds as an essential additive for fish health. Potential effects of metal contamination include:

- Accumulation of metals in sediments to concentrations that may cause toxic effects on benthic communities.
- Persistence of elevated metal concentrations in sediments for longer time frames than organic enrichment effects.
- Bioaccumulation within marine organisms and potential transfer through food webs.
- Local effects on reef communities via direct (water column) or indirect (trophic) pathways.

Deposition of metals generally follows the same pattern as for organic material, with most concentrated directly beneath pens and diminishing with distance. At the proposal area, dispersive hydrodynamic conditions make excessive accumulation less likely. Avoiding antifouling paints will also substantially reduce risks. Nonetheless, metals are persistent and may resuspend with finer particles, enabling dispersal. Experience from high-flow farms in the Marlborough Sounds suggests that concentrations typically return to background within ~ 200 m of farm centres (Sneddon and Tremblay 2011).

Therapeutants (antibiotics and parasiticides)

The use of therapeutants such as antibiotics, antibacterials and parasiticides in Aotearoa New Zealand finfish aquaculture is currently minimal, and information on their fate and effects in local marine environments is limited. Most therapeutants are water-soluble and degrade relatively rapidly (Forrest et al. 2007), suggesting low environmental persistence. The use of antibiotics on finfish farmed in the proposal area would be considered only if unavoidable; for example, if pathogens such as a *Piscirickettsia* spp. were detected and associated with clinical signs such as skin lesions, liver pathology and increased mortality, and in the absence of an effective vaccine (Forrest and Johnston 2025).

Potential risks of therapeutant use relate to impacts on non-target organisms (e.g. phytoplankton, zooplankton, sediment bacteria) and the development of resistant bacteria or parasites (Champeau 2013). Non-water-soluble compounds would be expected to disperse in a similar pattern to organic material, concentrating beneath farms. However, if accumulation of more persistent compounds such as zinc is managed effectively, effects from less persistent therapeutants are likely to be minimal.

Best-practice measures, including rotational use of sites, fallowing and good husbandry, reduce the likelihood that the use of therapeutants will be required. Should use increase, site-specific assessment, monitoring and management responses will be necessary.

Table 9. Summary of potential environmental effects arising from farm additives, and measures to avoid, remedy or mitigate these effects, where practicable.

Potential impact	Environmental implications	Options to avoid, remedy or mitigate (where applicable)
Toxic effects on seabed biota	Metals (zinc, copper) and other contaminants from feed or antifoulants may accumulate in sediments. Persistent contaminants may remain for years, although strong site currents should limit accumulation to areas directly beneath pens.	• Select sites that avoid areas with conspicuous epifaunal communities or sensitive taxa. • Avoid use of antifouling paints on nets; instead, use latest antifouling technologies where required. Prioritise net cleaning over chemical treatments. • Monitor physio-chemical and biological properties of sediment until contaminant levels are well understood. • Limit use of therapeutants through fallowing, rotational site use and good husbandry.
Long-term accumulation and persistence of metals	Elevated concentrations may persist beyond the duration of organic enrichment effects, with potential for resuspension and dispersal.	• Monitor sediment physico-chemical and biological properties until contaminant levels are well characterised. • Select sites that avoid areas with conspicuous epifaunal communities or sensitive taxa.
Impacts of therapeutants	Potential effects on non-target species (e.g. plankton, sediment bacteria) and risk of resistant bacteria or parasites. Effects in Aotearoa New Zealand marine environments are poorly documented.	• Limit use of therapeutants through fallowing, rotational site use, and good husbandry. • Assess and monitor any future use on a case-by-case basis (e.g. for treatment of pathogens such as *Piscirickettsia* spp. as per Forrest and Johnston [2025]).

5.6 Summary of effects

The potential seabed effects of the proposed operation are summarised in Table 10. Effects are described by activity type, zone of influence and their likely environmental implications. Likelihood of effect and significance are expressed using terminology aligned with the Resource Management Act 1991. In parallel, the magnitude, ecological value and level of effect are assessed following the EIANZ Ecological Impact Assessment (ECIA) guidelines (Roper-Lindsay et al. 2018; EIANZ 2024). This structure allows ecological assessments (magnitude, value, level of effect) to be presented alongside significance, providing a link between technical ecological analysis and regulatory decision-making. Ecological value is assigned based on the highest-value habitat potentially affected within each footprint. Key uncertainties and the management measures in place to address them are summarised in Table 11.

This assessment addresses prior panel comments regarding uncertainty in depositional effects and the sensitivity of biogenic habitats. Compared with the previous application, this assessment incorporates several improvements that increase confidence in predictions and reduce residual uncertainty. Modelling refinements include the use of long-duration hydrodynamic data to capture variability in ocean conditions, variable feed loading to replicate intended operations, explicit resolution of different faecal particle sizes and sinking rates, and incorporation of benthic substrate characteristics to better represent resuspension dynamics. Collectively, these advances provide more robust estimates of solids flux and residual solids. Further layers of conservatism have been applied to the modelling approach by basing particle mass decay on macrofaunal consumption only and modelling slightly higher feed inputs than those proposed. Additional habitat observations across the wider survey area have refined the mapping of bryozoan-sponge reefs, bushy-bryozoan thickets and patchy low-relief bryozoan-sponge habitats, improving confidence in their spatial extent and boundaries. Farm sites have also been repositioned to increase separation from extensive areas of biogenic habitat, avoiding overlap with the primary depositional footprint. In addition, this application applies more conservative thresholds for defining zones and levels of effect, providing a precautionary basis for assessment.

Overall, potential effects on the seabed have been avoided or reduced through farm siting on sandy seabed habitats (avoiding mapped biogenic habitats) and in areas with strong flushing potential. The main potential effects identified are:

- **Effects on macrofaunal communities from deposition of organic material and potential contaminants**. Within the immediate vicinity of the pens, high resuspension and dispersion are expected to limit impacts, while moderate deposition may occur across the primary footprint. Opportunistic and tolerant taxa are likely to dominate areas of higher enrichment, with reductions in diversity and abundance of sensitive taxa possible. Monitoring should target areas within the ZME and primary depositional footprints, as well as outside them to capture far-field effects.
- **Effects on sensitive epibiota and biogenic habitats, including bryozoan-sponge reefs, bushy-bryozoan thickets and patchy low-relief bryozoan-sponge habitats, from deposition of organic material and, to a lesser extent, other farm additives**. These habitats are mostly outside the OLE boundary, where predicted deposition is very low (< 0.7 $g{\cdot}m^{-2}$) and ecological effects are unlikely. Predicted overlap is restricted to a small area exposed to residual solids at levels an order of magnitude lower than in the primary footprint (~ 0.7 $g{\cdot}m^{-2}$) during periods of peak feed discharge at Stage 1, with a slightly bigger area predicted to receive ~ 2 $g{\cdot}m^{-2}$ at Stage 2. This assessment indicates that significant ecological effects are unlikely; however, a targeted monitoring programme is recommended to provide assurance, confirm the absence of far-field effects, and improve understanding of species-specific responses to low-level enrichment.

Remaining uncertainties include the sensitivity of biogenic habitats to low-level deposition, thresholds for ecological response, and potential cumulative effects with other activities. These uncertainties are addressed through a staged approach to development, an effects-based management framework, and a robust monitoring and adaptive management programme (recommendations are currently in preparation). This framework will ensure that any unforeseen effects or departures from predicted outcomes are detected early and managed effectively, thereby minimising residual ecological risk. Stage 1 operations will provide empirical evidence to further constrain predictions and validate assumptions, further reducing uncertainty around ecological risks to biogenic habitats.

Table 10 (overleaf). Summary of potential seabed effects from the proposed operation. The table outlines activity types, environmental implications, spatial extent, intensity, duration, affected zone, likelihood, magnitude (pre-mitigation), ecological value and significance (pre-mitigation). Mitigation measures are described alongside residual magnitude, level of effect and significance after mitigation. For consistency, the highest ecological value habitat predicted to be affected within each footprint is used in the assessment. Magnitude (Roper-Lindsay et al. 2018) describes the extent of change, while ecological value (EIANZ 2024; Table 2) reflects the importance of the affected habitat or community; together, these determine the level of effect and underpin the significance rating. Residual effects represent predicted impacts after mitigation, with residual magnitude × ecological value used to assign residual level of effect. Abbreviations: ZME = zone of maximum effect (solids flux > 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$); Primary = primary footprint (residual solids > 7 $g \cdot m^{-2}$); OLE = outer limit of effects (residual solids < 0.7 $g \cdot m^{-2}$); n/a = not applicable; CGSS = coarse gravel with shell and sand; BS-Reef = bryozoan-sponge reef.

Definition of terms used in the table

Spatial extent of effect: Localised (tens of metres), Medium (hundreds of metres), Large (> 1 km).

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

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

Likelihood of effect: n/a (not applicable / not expected), Low (unlikely), Moderate (possible), High (probable).

Magnitude of effect (as per Roper-Lindsay et al. [2018]): Nil (no effects at all; not included in Roper-Lindsay et al. [2018]), Negligible (barely distinguishable change), Low (minor, discernible change), Moderate (partial change), High (major change), Very high (near-total loss or fundamental change.

Ecological value (as per EIANZ [2024] and see Section 3.3 of this report): Low, Moderate, High, Very high.

Significance of effect: Nil (no effects at all), Negligible (effect too small to be discernible or of concern), Less than minor (discernible effect but very limited in area or duration, or only a very slight change in existing conditions), Minor (noticeable but will not cause any significant adverse effects), More than minor (noticeable that may cause adverse impact), Significant (noticeable and will have serious adverse impact).

Level of effect (assigned following Roper-Lindsay et al. [2018], whereby residual magnitude × ecological value = residual level of effect): Low, Moderate, High, Very high.

Activity	Effect	Environmental implications	Spatial extent, intensity, duration	Zone	Likelihood	Magnitude (pre)	Ecological value	Significance (pre)	Mitigation	Residual magnitude	Level of effect	Residual significance
Mooring installation	Resuspension and disturbance	Short-term resuspension and smothering; localised damage to habitat / biota beneath moorings.	**Spatial extent: Localised** (limited to areas directly around anchor sites). **Intensity: High** (displacement and loss of habitat). **Duration: Persistent** (short-term impact of installation but ongoing impacts from chain sweep. Recovery on scale of months to years for sensitive epifaunal species when farm is removed).	ZME	High	Moderate	Low–Moderate (CGSS)	Minor	Avoid biogenic habitats; use experienced crew and best-practice installation methods.	Low	Very low–Low	Less than minor
				Primary	n/a	Nil	Low–Moderate (CGSS)	Nil		Nil	Nil	Nil
				OLE	n/a	Nil	Very high (BS-reef)	Nil		Nil	Nil	Nil
Presence of structures	Biofouling drop-off	Fouling material may alter seabed composition and contribute to enrichment. Fouling biota may include pest species. Beyond ZME, structures may act as propagule sources for pests.	**Spatial extent: Medium** (affected area limited mainly to seabed beneath pens, mooring lines and around anchors). **Intensity: High** (if pest species are introduced to areas of biogenic habitat). **Duration: Persistent** (recovery on the order of months to years when farm is removed, unless persistent colonisation occurs).	ZME	High	Moderate	Low–Moderate (CGSS)	More than minor	Avoid biogenic habitats; manage fouling; carry out biosecurity monitoring and management.	Low	Low	Minor
				Primary	Low	Moderate	Low–Moderate (CGSS)	Minor		Low	Very low	Negligible
				OLE	Low	High	Very high (BS-reef)	Significant		Low	Low	Negligible
	Shading	Localised reduction in light to seabed; very	**Spatial extent: Medium** (affected area limited largely to seabed beneath pen perimeters,	ZME	Moderate	Negligible	Low–Moderate (CGSS)	Negligible	Avoid placement over areas	Negligible	Very low	Negligible

Activity	Effect	Environmental implications	Spatial extent, intensity, duration	Zone	Likelihood	Magnitude (pre)	Ecological value	Significance (pre)	Mitigation	Residual magnitude	Level of effect	Residual significance
		few multicellular primary producers in area.	mooring lines and, to a lesser extent, underneath pens). **Intensity: Low.** **Duration: Persistent** (recovery on the order of days to months when farm is removed).	Primary	n/a	Nil	Low–Moderate (CGSS)	Nil	identified as having primary producers.	Nil	Nil	Nil
				OLE	n/a	Nil	Very high (BS-reef)	Nil		Nil	Nil	Nil
Deposition	Attraction of scavengers / predators	Increased scavenger/predator abundance around biodeposits and enriched areas, altering interactions.	**Spatial extent: Medium** (limited mainly to the farm footprint). **Intensity: High** (possible displacement of sensitive epibiota). **Duration: Persistent** (effects reversible within months following cessation of farming and dispersal of predators, depending on level of effect).	ZME	Moderate	Moderate	Low–Moderate (CGSS)	More than minor	Limit deposition on biogenic habitats; apply effects-based management and triggers.	Low	Low	Minor
				Primary	Moderate	Moderate	Low–Moderate (CGSS)	More than minor		Low	Low	Less than minor
				OLE	Low	Negligible	Very high (BS-reef)	Negligible		Negligible	Low	Negligible
	Macrofaunal communities	Alteration of macrofaunal communities and loss of sensitive taxa. Effect reversible if feed inputs cease. Recovery back to background state is on the order of years following farm removal.	**Spatial extent: Large** (effect will occur throughout the entire deposition zone, but with a gradient of intensity). **Intensity: High** (displacement of sensitive infaunal species in the ZME and primary footprint). **Duration: Persistent** (for the duration of farm, but the highest magnitude of effects will be periodic in accordance with feed use cycles, or less intense if constant discharge. Recovery back to background state is on	ZME	High	High	Low–Moderate (CGSS)	Significant	Carry out monitoring and adaptive management, minimise feed waste, apply fallowing.	Moderate	Moderate	More than minor
				Primary	Moderate	High	Low–Moderate (CGSS)	More than minor		Moderate	Moderate	Minor
				OLE	Low	Moderate	Low–Moderate (CGSS)	Negligible		Low	Low	Negligible

Activity	Effect	Environmental implications	Spatial extent, intensity, duration	Zone	Likelihood	Magnitude (pre)	Ecological value	Significance (pre)	Mitigation	Residual magnitude	Level of effect	Residual significance
			the order of years following farm removal).									
	Epibiota (epifaunal communities)	Alteration of epifaunal communities and sensitive taxa within the ZME. Changes to abundance/growth/recruitment possible within the primary footprint Beyond that less certain, expect slight changes (e.g., increased food for filter-feeders).	**Spatial extent: Large** (effect will occur throughout the entire deposition zone, but with a gradient of intensity) **Intensity: High** (note: low epifaunal abundance in ZME / primary footprints but high impact; high epifauna abundance in small areas of OLE but low impact). **Duration: Persistent** (effects reversible if feed inputs cease, recovery on the order of several years if habitat-forming biota are displaced, depending on the impact level).	ZME	High	Very high	Low–Moderate (CGSS)	Significant	Avoid biogenic habitats; carry out monitoring and adaptive management.	Moderate	Moderate	Minor
				Primary	High	High	Low–Moderate (CGSS)	More than minor		Moderate	Moderate	Minor
				OLE	Moderate	High	Very high (BS-reef)	More than minor		Low	Moderate	Minor
	Oxygen depletion	Potential near-bottom oxygen reduction beneath pens under high enrichment.	**Spatial extent: Medium** (would occur only in areas with highest deposition). **Intensity: High** (displacement of fauna). **Duration: Moderate** (effects reversible within months following farm removal).	ZME	Moderate	High	Low–Moderate (CGSS)	More than minor	Avoid biogenic habitats; apply effects-based management.	Moderate	Low	Minor
				Primary	Low	Nil	Low–Moderate (CGSS)	Nil		Nil	Nil	Nil
				OLE	Low	Nil	Very high (BS-reef)	Nil		Nil	Nil	Nil

Activity	Effect	Environmental implications	Spatial extent, intensity, duration	Zone	Likelihood	Magnitude (pre)	Ecological value	Significance (pre)	Mitigation	Residual magnitude	Level of effect	Residual significance
	Nutrients	**Nutrients (mainly nitrogen and phosphorus) released from organic material resulting in increased algal growth.**	**Spatial extent: High** (fertilisation effect possible across the depositional footprints). **Intensity: Moderate** (waste deposition and accumulation relatively low). **Duration: Persistent** (effects reversible within months following farm removal).	ZME	Low	Moderate	Low–Moderate (CGSS)	Minor	Carry out monitoring and adaptive management, minimise feed waste, apply fallowing.	Low	Low	Less than minor
				Primary	Low	Moderate	Low–Moderate (CGSS)	Minor		Low	Low	Less than minor
				OLE	Low	Moderate	Very high (BS-reef)	Minor		Low	Moderate	Minor
	Farm additives	Possible accumulation of persistent compounds in areas with highest deposition; sub-lethal effects uncertain. Negligible accumulation expected beyond primary footprint.	**Spatial extent: Medium** (would occur only in areas with highest deposition). **Intensity: High** (displacement of fauna). **Duration: Moderate** (effects reversible within months following farm removal, although accumulations in the sediment may be apparent for several years after farm removal).	ZME	High	Very high	Low–Moderate (CGSS)	Significant	Avoid biogenic habitats; monitor sediments / epibiota; apply best-practice chemical use / reduce contaminant inputs (e.g. net cleaning / avoid use of copper in antifoulants, minimise treatments).	Moderate	Moderate	Minor
				Primary	Low	Low	Low–Moderate (CGSS)	Less than minor		Low	Low	Less than minor
				OLE	Low	Negligible	Very high (BS-reef)	Negligible		Negligible	Low	Negligible

Table 11. Key areas of uncertainty, mitigation responses and improvements since the earlier application.

Area of uncertainty	Why uncertain	Mitigation / management response	Improvements since earlier application
Model predictions	Hydrodynamic and depositional models simplify complex processes (especially resuspension and benthic assimilation).	• Conservative use of depositional footprints (ZME at 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ solids vs 1 $kg \cdot m^{-2} \cdot yr^{-1}$ solids, moderate enrichment at 7 $g \cdot m^{-2}$ residual solids vs 12.5 $g \cdot m^{-2}$). • Effects assessment based on peak feed inputs (these will be followed by periods of low to no feed). • Modelled with slightly higher annual feed inputs than will be used. • Conservative decay rate based on macrofaunal consumption. Waste will also be consumed by micro-organisms. • Use of effects-based monitoring and adaptive management.	Updated and refined modelling with long-duration hydrodynamic data to include variability in ocean conditions, variable feed loading that replicates intended operations, resolution of different-sized faecal particles and sinking rates, and consideration of benthic substrate in determining resuspension dynamics.
Ecological thresholds	Thresholds for ecological responses are based on relationships developed for the Marlborough Sounds.	• Conservative use of depositional footprints (ZME at 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ solids, moderate enrichment at 7 $g \cdot m^{-2}$ residual solids). • Effects assessment based on peak feed input (this will be followed by periods of low to no feed). • Modelled with slightly higher annual feed inputs than will be used. • Use of effects-based monitoring and adaptive management.	Adoption of effects-based management framework consistent with Aotearoa New Zealand standards (e.g. enrichment stage thresholds).

Area of uncertainty	Why uncertain	Mitigation / management response	Improvements since earlier application
Effects on sandy seabed	The Aotearoa New Zealand evidence base is stronger for muddy habitats; there are fewer studies on sandy systems and their response to organic loading.	• Conservative use of depositional footprints (ZME at 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$ solids, moderate enrichment at 7 $g \cdot m^{-2}$ residual solids). • Use effects-based monitoring and adaptive management of sandy habitats within and outside footprint. • Sandy sediments are more easily oxygenated, making organic waste less likely to accumulate. • Fallowing to support recovery.	No areas predicted to receive very high enrichment.
Biogenic habitats and low deposition	Sensitivity of biogenic taxa to very low levels of organic enrichment less well understood.	• Farms sited at least 1 km from biogenic habitat along the tidal axis (southeast–northwest) and 500 m in other directions, in line with best-practice guidance and expert advice (e.g. Smith 2021). • Use of effects-based monitoring and adaptive management to monitor biogenic habitat.	Farm relocation and refined siting to avoid primary depositional footprint overlap with areas of biogenic habitat.
Recovery time frames	Recovery speed depends on level of enrichment, sediment type, hydrodynamics and species present.	• Fallowing. • Use of interim monitoring to track recovery; adaptive management.	No change.
Farm additives / contaminants	Persistence and biological uptake of antifoulants and therapeutants in marine sediments in Aotearoa New Zealand not fully characterised.	• Minimal use of antifoulants (prefer washing over treating nets). • Monitoring of sediment chemistry and epibiota.	Farm relocation and refined siting to avoid primary depositional footprint overlap with areas of biogenic habitat.

6. Monitoring and management of seabed effects

While seabed effects of salmon farming are well documented in sheltered, soft sediment environments, the coarse-grained, highly dispersive conditions at the Hananui site present specific challenges for monitoring and management. Organic wastes at dispersive sites are transported further from net pens, increasing the potential for far-field effects. This is particularly relevant given the presence of sensitive biogenic habitats adjacent to the proposal area.

Ngāi Tahu Seafood proposes a two-stage development, allowing the effects of Stage 1 farming activity to be assessed before progressing to Stage 2. Staging provides a mechanism to ensure development remains within acceptable environmental limits and allows operational practices to be adaptively adjusted as required.

An effects-based management approach is recommended to guide farm operations and minimise seabed impacts. Effects-based management involves defining acceptable environmental outcomes or thresholds for effects on the seabed and associated communities (Fletcher et al. 2022). These thresholds may include limits on organic deposition fluxes and the spatial extent of enriched seabed areas, and can be informed by best management practice guidelines developed for managing impacts to seabed environments for inshore salmon farms (Fletcher et al. 2022) and open-ocean finfish culture (Giles et al. 2021). For example, in the Marlborough Sounds seabed effects are managed through limits on deposition intensity (enrichment stage [ES] 5.0, ~ 13 kg solids·m^{-2}·yr^{-1}) and on the spatial extent of seabed experiencing ES > 3.0 (~ 1 kg solids·m^{-2}·yr^{-1}) (Fletcher et al. 2022). While the Hananui site differs in sediment type and hydrodynamics, these guidelines provide a useful framework that can be adapted locally.

Farm siting has been designed to avoid dense biogenic habitats and positions farms over soft sediments with high flushing potential, reducing baseline risk. Effects-based management will also guide operational decisions, including feed rates and stocking densities, to ensure effects remain within acceptable limits.

Adaptive management complements effects-based management by providing a structured, iterative approach to managing uncertainty. Under adaptive management, monitoring results inform whether effects approach or exceed thresholds, enabling timely operational adjustments. Adaptive management is particularly important at dispersive sites, where far-field effects and sensitive taxa responses are less predictable, and supports refinement of monitoring during the two-stage development to detect ecological responses early.

Effects-based management measures to avoid and minimise seabed effects, while reducing residual uncertainty, include:

- **Site fallowing and rotation**: Each farm site will be fallowed for at least 3 months at the end of each production cycle to reduce cumulative impacts and support recovery.
- **Ongoing monitoring, model validation and adaptive management**: Monitoring data will guide operational adjustments to ensure effects remain within acceptable limits, including:

- *reduction of farming intensity* – feed inputs and stocking densities may be adjusted to limit deposition and footprint extent
- *relocation options* – if required, farms may be shifted to alternative sites to protect sensitive habitats.

Monitoring at Hananui will need to consider both near-field and far-field effects, particularly where sensitive biogenic habitats occur outside the primary depositional footprint. Monitoring could include community-level metrics (abundance, cover, diversity) and species-specific responses. Early-warning indicators, such as reductions in key sensitive species, can signal low-level enrichment before broader community changes occur. Habitat-specific monitoring at other farms, including rocky reefs and biogenic habitats (e.g. Dunmore 2022), highlights the importance of targeted assessment.

In the absence of established guidelines for the proposal area, site-specific limits of acceptable effect will be developed in consultation with iwi, stakeholders and scientific advisors. Detailed and flexible implementation will be captured in environmental monitoring plans (EMOPs), allowing methods and thresholds to evolve as knowledge, technology and understanding of environmental conditions advance.

7. Key findings

The main findings of our seabed assessment are given below.

Site characteristics

- Water depths in the proposal area range from 20 m to 40 m (with strong mean currents 0.56 $m \cdot s^{-1}$ near the surface and 0.34 $m \cdot s^{-1}$ near the seabed, Wilson 2025).
- Sediments are predominantly coarse-grained sands with shell hash, gravel and minimal mud.
- Substrates are well oxygenated, with low organic content.
- Macrofaunal communities are naturally low in abundance but moderately diverse.

Habitat types – six primary seabed habitat types were mapped across the survey area

Within the Hananui proposal area

- Three primary seabed habitat types were recorded:
  - *sand* – 66% of the proposal area; sparse epifauna
  - *sandy shell hash* – 16% of the proposal area; sparse epifauna
  - *coarse gravel with shell and sand* – 18% of the proposal area; sparse epifauna with occasional isolated biogenic clumps.
- While habitat modelling predicted trace amounts of biogenic habitat within the proposal area (< 0.1%), these are considered modelling artefacts, shell fragments, or isolated biogenic clumps with limited ecological significance, occurring as flecks within a sand-dominated substrate.

Across the wider survey area

- Six primary seabed habitat types were recorded:
  - *sand* – 39% of the total survey area
  - *sandy shell hash* – 27% of the total survey area
  - *coarse gravel with shell and sand* – 22% of the total survey area
  - *bushy-bryozoan thickets* – 4% of the total survey area; high epifaunal diversity, including erect bryozoans, sponges, brachiopods and large bivalves
  - *bryozoan-sponge reefs* – 7% of the total survey area; highly biodiverse, dominated by erect and encrusting bryozoans, sponges and tube worms
  - *patchy low-relief bryozoan-sponge habitat* –1% of the total survey area; areas of higher biodiversity interspersed with coarse gravel with shell and sand.

Predicted deposition – Stage 1

- **Zone of maximum effect** (ZME; > 0.5 $kg \cdot m^{-2} \cdot yr^{-1}$): No areas are predicted to exceed very high enrichment (> 13 $kg \cdot m^{-2} \cdot yr^{-1}$). Around 10–19 ha (0.8–1.5% of the proposal area) may exceed 5 $kg \cdot m^{-2} \cdot yr^{-1}$ for each farm during peak feed input, extending up to ~ 150 m from pen edges.

- **Primary footprint** (≥ 7 g·m^{-2} residual solids): 510–533 ha for each farm at peak feed input (40–42% of the proposal area), largely contained within the proposal area and not overlapping mapped biogenic habitats.
- **Outer limit of effects** (OLE; < 7 g·m^{-2} > 0.7 g·m^{-2} residual solids): A further 958–1,262 ha could receive minor enrichment during periods of peak production at each farm and there is minimal overlap with biogenic habitats at Stage 1.
- **Far-field**: Trace farm waste (< 0.7 g·m^{-2}) may disperse more than 17 km northwest and southeast of the proposal area boundary. At this distance, waste is expected to be diluted and assimilated, with any enrichment being indistinguishable from natural variability.

Predicted deposition – Stage 2

- **ZME**: No areas are predicted to exceed very high enrichment (> 13 kg·m^{-2}·yr^{-1}). Around 10–19 ha (0.8–1.5% of proposal area) may exceed 5 kg·m^{-2}·yr^{-1} for each farm during peak feeding, extending up to ~ 250 m from pen edges.
- **Primary footprint**: 473–533 ha (37–42% of proposal area) for each farm at peak feed input, largely within the proposal area and not overlapping mapped biogenic habitats.
- **During periods of peak production**: The OLE overlaps with up to 43 ha of bryozoan-sponge reef (5% of total mapped habitat), 25 ha of patchy low-relief habitat (17% of total mapped habitat) and 16 ha of bushy-bryozoan thickets (3% of total mapped habitat) per farm/ farm block.
- **Far-field**: Trace farm waste (< 0.7 g·m^{-2}) may disperse more than 17 km northwest and southeast of the proposal area boundary. Again, at this level waste is expected to be diluted and assimilated, with any enrichment unlikely to be distinguishable from natural variability.

Predicted seabed effects by zones

- **Within the ZME:** Moderate to high enrichment predicted across predominantly sandy seabed habitats.
  - *effects to macrofaunal communities* = 'More than minor' due to the intensity and spatial extent. Substantial shifts in macrofaunal community composition expected.
  - *effects to epibiota* = 'Minor', as likely to result in displacement of the few epibiota present and / or increase in scavengers / predators.
- **Within the primary footprint:** Moderate enrichment predicted across predominantly sandy seabed habitats.
  - *effects to macrofaunal communities* = 'Minor' due to the magnitude of change across a large spatial extent. Macrofaunal communities are likely to show shifts in abundance and composition, but community function will be largely retained.
  - *effects to epibiota* = 'Minor', as community shifts are possible across a large area, although only few epibiota are present.
- **Within the OLE**: Minor enrichment possible across mostly sandy seabed habitats and a small proportion of biogenic habitat.

- *effects to macrofaunal communities* = 'Negligible' due to a possible 'enhancement' effect across a large spatial scale.
- *effects to epibiota* = 'Minor' due to low-level enrichment (approximately an order of magnitude lower than the primary footprint) reaching a small area with high epifaunal abundance.

Confidence and residual uncertainty

- The assessment is supported by refined depositional modelling and an improved understanding of biogenic habitat boundaries.
- Conservative thresholds were applied in defining zones and levels of effect.
- Results are based on periods of peak feed discharge, with additional confidence provided by the fact that these are interspersed with periods of low or no discharge, allowing recovery between production cycles.
- Effects-based management, combined with a staged development approach, will enable monitoring of potential effects of concern. Farming practices can be adapted as needed to minimise risk, ensuring the development remains within acceptable environmental limits.
- Key uncertainties remain regarding ecological responses in dispersive, coarse-sediment environments and potential far-field effects on biogenic habitats; however, adverse ecological effects to these habitats are considered unlikely.

Overall residual significance: With conservative siting, thresholds, and a robust effects-based management and monitoring framework, residual ecological effects on seabed habitats are expected to be low and manageable. Stage 1 operations will provide empirical data to further reduce uncertainty and validate predictions.

8. Acknowledgements

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9. Appendices

Appendix 1. Laboratory analytical methods for sediment samples processed by Hill Labs

Analyte	Method	Default detection limit
Particle grain size	Wet sieving using dispersant with gravimetric calculation applied. Seven size classes applied based on the Udden–Wentworth scale: ≥ 2 mm = Gravel ≥ 1 mm to < 2 mm = Very Coarse Sand ≥ 500 µm to < 1 mm = Coarse Sand ≥ 250 µm to < 500 µm = Medium Sand ≥ 125 µm to < 250 µm = Fine Sand ≥ 63 µm to < 125 µm = Very Fine Sand < 63 µm = Mud (Silt & Clay)	Particle grain size
Organic matter (as ash-free dry weight [AFDW])	Ignition in muffle furnace 550 °C, 6 hr, gravimetric. APHA 2540 G 22nd ed. 2012.Calculation: 100 – ash (dry weight)	0.04 g/100 g dry wt

Appendix 2. Definitions of community measures calculated for macrofaunal communities (see Appendix 6)

Indicator	Calculation and description	Reference
N	Sum (*n*) Total macrofauna abundance = number of individuals per 13 cm-diameter core.	–
S	Count (taxa) Taxa richness = number of taxa per 13 cm-diameter core.	–
d	$(S - 1) / \log N$ Margalef's diversity index. Ranges from 0 (very low diversity) to ~ 12 (very high diversity).	Margalef (1958)
J'	$H' / \log S$ Pielou's evenness index. A measure of equitability, or how evenly the individuals are distributed among the different species. Values can range from 0.00 to 1.00. A high value indicates an even distribution and a low value indicates an uneven distribution or dominance by a few taxa.	Pielou (1966)
H'	$- \sum_i p_i \log(p_i)$ where p is the proportion of the total count arising from the ith species. Shannon–Wiener diversity index (SWDI). A diversity index that describes, in a single number, the different types and amounts of animals present in a collection. Varies with both the number of species and the relative distribution of individual organisms among the species. The index ranges from 0 for communities containing a single species, to high values for communities containing many species where each is represented by a small number of individuals.	–
AMBI	= [(0 × %GI + 1.5 × %GII + 3 × %GIII + 4.5 × %GIV + 6 × %GV)] / 100 where GI, GII, GIII, GIV and GV are ecological groups. Azites Marine Biotic Index: relies on the distribution of individual abundances of soft-bottom communities according to five ecological groups (GI–GV). GI includes species sensitive to organic pollution and present under unpolluted conditions, whereas, at the other end of the spectrum, GV species are first-order opportunists adapted to pronounced unbalanced situations (e.g. *Capitella capitata*). Index values are between 1 (normal) and 6 (extremely disturbed).	Borja et al. (2000)
M-AMBI	Uses AMBI, *S* and *H'*, combined with factor analysis and discriminant analysis (see source reference). Multivariate-AMBI. Integrates the AMBI with measures of species richness and SWDI using discriminant analysis (DA) and factorial analysis (FA) techniques. Uses reference conditions for each parameter (based on 'pristine conditions') that allows the index to be tailored to accommodate environments with different base ecological characteristics. Scores are from 1 (high ecological quality) to 0 (low ecological quality).	Muxika et al. (2007)

Indicator	Calculation and description	Reference
BQI	$= \left(\sum_{i=1}^{n}\left(\frac{A_i}{totA} \times \mathrm{ES50}_{0.05i}\right)\right) \times {}^{10}\log(S+1)$ Where ES50 = expected number of species, as per Hurlbert (1971), and $\mathrm{ES50}_{0.05}$ is the species tolerance value, given here as the 5th percentile of the ES50 scores for the given taxa as per Rosenberg et al. (2004). Benthic quality index uses species-specific tolerance scores ($\mathrm{ES50}_{0.05}$), abundance and diversity factors. Results can range from 0 (being highly impacted) and 20 (reference conditions). Application of this index at this site will require the calculation of regional species $\mathrm{ES50}_{0.05}$ values.	Rosenberg et al. (2004)

Appendix 3. Predictive habitat map – multibeam data ground-truthing and analysis

A3.1 Introduction

High-resolution multibeam echosounders (MBES) have been instrumental in a number of studies that have demonstrated the relationships between backscatter intensity and sediment type, as well as bathymetry and bathymetric derivatives, with seafloor features and corresponding biogenic habitats (reviewed in Brown et al. 2011). Using machine learning and other statistical approaches, these environmental variables (e.g. backscatter, sediment type and bathymetry) can be used to model the extent of, or suitability of, an area for biogenic habitats (Trzcinska et al. 2020; Xu et al. 2021).

A3.2 Methodology

Multibeam data processing / model inputs

Predictor variables used in the predictive habitat model developed for this study included the bathymetry and backscatter data gathered from the MBES survey, as well as geomorphic statistics extracted from the bathymetry datasets and the maximum current velocities in the area. Bathymetric predictors included the slope, curvature (calculated from the bathymetry surface using the Benthic Terrain Modeler toolbox for ArcGIS; Walbridge et al. 2018) and Terrain Ruggedness Index rugosity (height difference between a central cell and the adjacent cells in the raster, calculated using the spatialEco package in the R environment). In addition, textural statistics from the bathymetry, bathymetry derivatives and backscatter data were derived from grey-level co-occurrence matrices (including 'mean', 'variance', 'homogeneity', 'contrast', 'dissimilarity', 'entropy', 'second moment' and 'correlation') using the package glcm (Zvoleff 2020) within the R environment. The texture analysis allowed the variability of the seafloor to be quantified and included as predictor variables.

To include current velocities as a predictor in the model, a raster of maximum current velocities was calculated from a dataset of ocean currents derived from two spring neap cycles (29.2 days) from 1 December 2018. Because the data points from the current dataset were not evenly distributed across the area of interest, we interpolated the maximum values using an inverse distance weighted method (Pebesma 2004; Benedikt et al. 2016).

All predictor variables were initially calculated as 1 m^2 resolution rasters and resampled to a resolution of 10 m^2 using a bilinear interpolation, where the new value of a cell is based on a weighted distance average of the four nearest input cell centres. This was to account for inaccuracies in remotely operated vehicle (ROV) data due to the way in which the ultra-short baseline (USBL, a method of underwater positioning) calculates the position of the ROV. The USBL does this by measuring signals between the unit attached to the ROV and the transceiver, and calculates the range (time between signals) and angle between signals. Typically, accuracy is 2% of the slant range (distance across planes; ~ 100 m in our case or a 2 m error); however, other factors (e.g. uncalibrated boat movements and the signal frequency) can increase the error, leading us to use a 10 m estimate of error and consequently resample the predictor variables to 10 m^2 rasters. These were used as predictor variables to train the model, using drop-camera and ROV imagery of habitats as response variables.

Ground-truthing

Ground-truthing data came from six different surveys:

- Drop-camera surveys designed to ground-truth side-scan sonar collected across six surveys from 2018 to 2019
- ROV survey designed to observe the seabed under potential farm locations (October 2021)
- ROV survey designed to ground-truth the Discovery Marine Limited (DML) multibeam imagery (July 2022)
- ROV survey to validate initial model results (September 2022)
- Drop-camera survey (further ground-truthing of DML multibeam imagery) designed to target small areas predicted to be biogenic habitat within the vicinity of the proposed farm locations and predicted primary depositional footprints (March 2023)
- Drop-camera survey (further ground-truthing of DML multibeam imagery) designed to target under-revised farm locations and collect more data from areas of biogenic habitat to the northwest (April 2025).

All the imagery was collected with an associated geolocation. For the earlier drop-camera surveys (2018–21), this was the GPS on the boat, whereas for the later drop-camera (2023 onwards) and ROV surveys, a USBL system was used, enabling the ROV and imagery to be tracked separately from the boat. The USBL sends a location ping back to the receiver every 3 seconds. The positions of each of these pings were used as the input points for the model.

The imagery from these surveys were grouped into six classes:

- Bryozoan-sponge reefs
- Patchy low-relief bryozoan-sponge habitat
- Bushy-bryozoan thickets
- Sand
- Sandy shell hash (SSH)
- Coarse gravel with shell and sand (CGSS).

These habitat types are described in the main body of the report (Section 3.2). From the surveys, 10,531 points labelled with the habitats observed, time stamps and locations were collected (Figure A3.1).

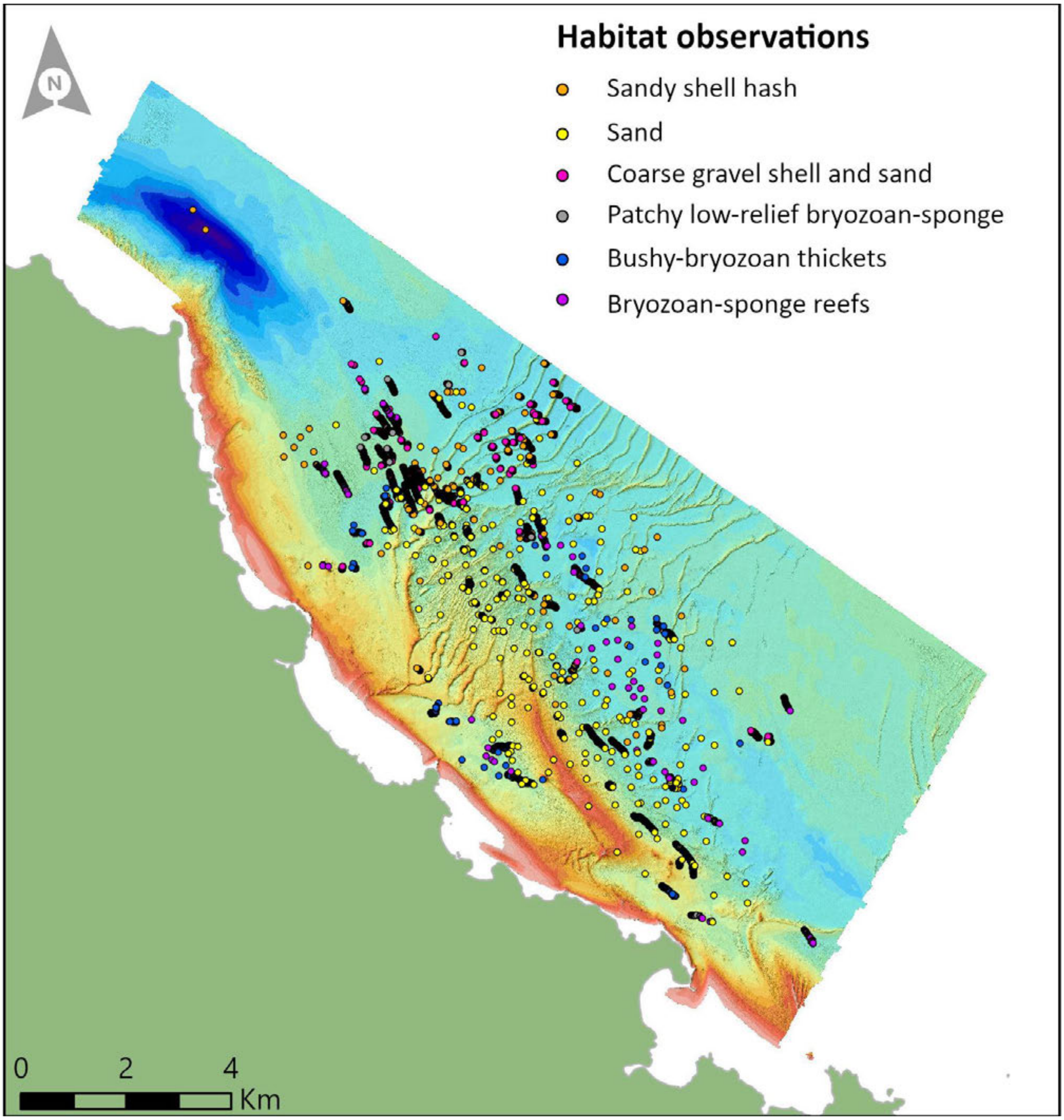

Figure A3.1. All drop-camera and ROV imagery collected and used in the ground-truthing of the predictive model.

Model validation / performance

The raster layers for each predictor variable were sampled using the locations of the ground-truthing points to obtain the inputs for the benthic habitat predictive model. The modelling approach used an extreme gradient boosting model. This model is an implementation of gradient-boosted decision trees, a class of machine learning algorithms that can be used for classification or regression predictive modelling problems. This technique works by sequentially creating an ensemble of weak predictive models, with each new model attempting to correct for the deficiencies in the previous model (Friedman 2001). For this, packages xgboost (Chen et al. 2022) and caret (Kuhn 2022) were used.

To cross-validate the model outputs, the ground-truthing data was divided into two equal datasets and used 80% of the data for training the models and 20% of the data to test the resulting best model. From this cross-validation two model evaluation metrics were analysed, accuracy and Cohen's kappa. Accuracy is the percentage of correct classifications out of all instances, and Cohen's kappa is the accuracy normalised at the baseline of random chance on the dataset (agreement between categories). Cohen's kappa is useful because accuracy can be unclear when the classification problem has more than two classes; we also analysed the confusion matrices that visualise the model performance. Lastly, metrics concerning the variable importance for the models were obtained. Variable importance is determined by calculating the relative influence of each variable depending on whether the variable was used during the tree-building process, and how much the squared error (over all trees) improved or decreased as a result. The best-performing model was then used to predict the habitat classes for the whole area, using the 10 m raster information (bathymetry, bathymetry derivatives, backscatter and their texture variables).

Because the 80% of data used for training purposes was randomly selected, the resulting best model and predictions could potentially be influenced by the choice of training set. Therefore, the above process was run 84 times to analyse the average model evaluation metrics and the average confusion matrix, and determine the uncertainty in the results for each class of habitat. The average variable importance, and their standard deviation were also calculated. In the resulting percentage-based predicted benthic habitat raster, each pixel represents the most common class that was predicted out of 84 iterations of the process. Additionally, a probability raster was created for each class representing the number of times that each pixel was predicted to be that class from the 84 iterations.

A3.3 Results

Mapping / discrimination of ground-truth samples

From the inputs, the model classified 12,469 ha of the benthic environment into the habitat categories (Figure A3.2). Of this area, 818 ha (6.6%) was classified as bryozoan-sponge reef, 146 ha (1.2%) was classified as patchy low-relief bryozoan-sponge habitat, 497 ha (4.0%) was classified as bushy-bryozoan thickets, 2,778 ha (22.3%) was classified as coarse gravel with shell and sand, 4,907 ha (39.3%) was classified as sand, and 3,324 ha (26.7%) was classified as sandy shell hash.

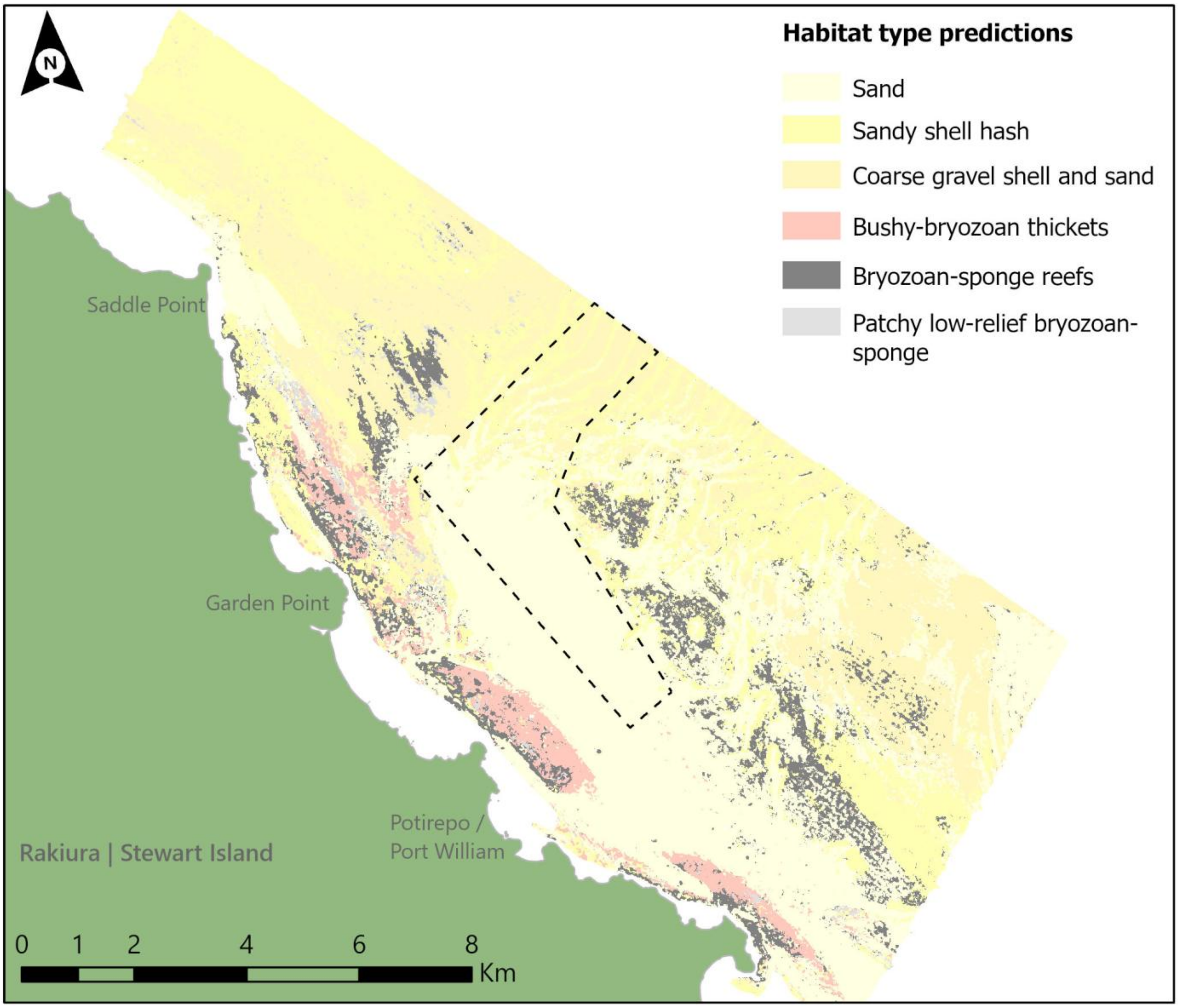

Figure A3.2. Map of the predicated habitats using 80% of the ground-truthed data as the training set.

Accuracy and model performance

When using 80% of the data as a training dataset, the model accurately predicted 82.1 ± 0.004% of the test data points (the remaining 20% of the ground-truthing dataset), with a Cohen's kappa of 0.76 ± 0.006 indicating substantial agreement between the classes (Landis and Koch 1977). Sand had the highest accuracy percentage (93%), followed by coarse gravel with shell and sand (87%), sandy shell hash (83%), bryozoan-sponge reef (79%), bushy-bryozoan thickets (78%) and patchy low-relief bryozoan-sponge habitat (53%) (Table A3.1).

Table A3.1. Confusion matrix for the predictions from the model using 80% of the data as the training dataset. The top 'Actual' row indicates the ground-truthed assessment of that spot, and the 'Prediction' column indicates the percentage of time the model predicted that data point. BS-Reef = bryozoan-sponge reef, Bushy Bry = bushy-bryozoan thickets, CGSS = coarse gravel with shell and sand, PLBS = patchy low-relief bryozoan-sponge habitat, Sand = sand, SSH = sandy shell hash. Green cells indicate the accuracy for each habitat type.

		Actual					
		BS-Reef	**Bushy Bry**	**CGSS**	**PLBS**	**Sand**	**SSH**
Prediction	**BS-Reef**	79%	5%	2%	15%	1%	2%
	Bushy Bry	1%	78%	1%	2%	1%	0%
	CGSS	3%	2%	87%	17%	2%	6%
	PLBS	4%	1%	4%	53%	0%	2%
	Sand	10%	12%	4%	6%	93%	8%
	SSH	3%	2%	2%	7%	3%	83%

Variable importance

The maximum current velocity was observed to be the most important variable in predicting the habitats (Figure A3.3). In next order of importance were the bathymetry and backscatter textural statistics. These variables represent a range of geomorphological features and the overlying hydrographic environment, which either drive the patterns of species distribution or represent benthic biological features (i.e. structures) that can be detected in the environmental data layers (Brown et al. 2011; Trzcinska 2020).

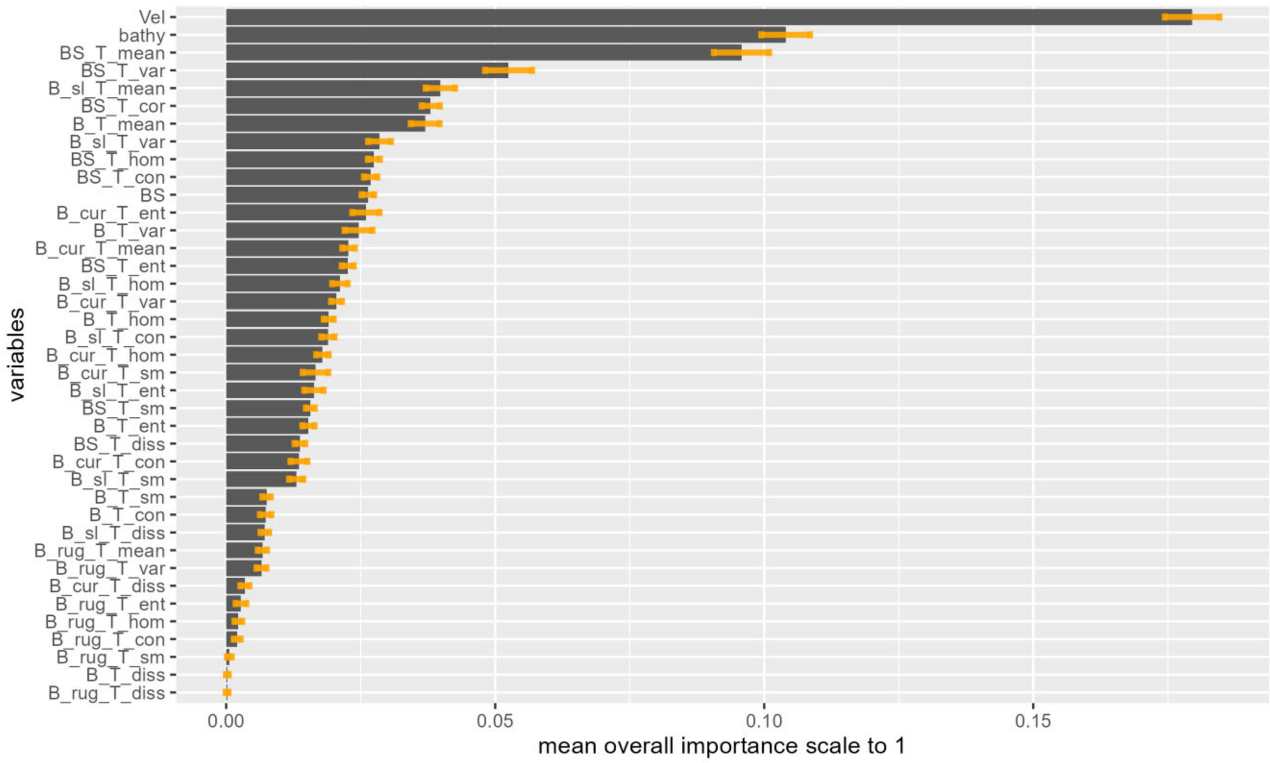

Figure A3.3. The mean importance of each input parameter into the final prediction model. Error bars indicate the variance over the 84 model runs of the uncertainty assessment. Vel = maximum current velocity, BS = backscatter, B = bathymetry, T= texture analysis, var = variance, cor = correlation, hom = homogeneity, ent = entropy, sm = second movement, diss = dissimilarity, sl = slope, cur = curvature, rug = TRI (Terrain Ruggedness Index) rugosity.

Appendix 4. Particle grain size distribution (% dry weight) and organic content (% ash-free dry weight; AFDW) of sediments from 31 sites across the Hananui proposal area

Site	Silt & Clay % (< 63 µm)	Very Fine Sand % (63 to < 125 µm)	Fine Sand % (125 to < 250 µm)	Medium Sand % (250 to < 500 µm)	Coarse Sand % (500 µm to < 1 mm)	Very Coarse Sand % (1 to < 2 mm)	Gravel % (≥ 2 mm)	AFDW %
1	4	1.1	17	19.2	5.5	10.7	42.4	2.5
5	4	0.4	27.2	36.7	16.5	6	9.3	1.36
6	3.9	0.2	7.4	39.7	18.1	10.5	20.2	1.49
8	2.7	0.2	0.7	13.2	19.3	26.7	37.2	1.99
9	4.5	0.3	23.7	54	10.1	4.2	3.2	1.54
10	5.5	0.3	4	29.3	16.2	6.9	37.8	2.5
13	3.2	0.6	42	46.7	6.1	1	0.2	1.33
14	3.7	0.2	9.3	69.2	13.2	2.8	1.5	1.41
15	4	< 0.1	0.4	36.5	51	5.9	2.2	0.96
16	4	0.9	12	23.5	17	11.6	31	2.4
17	6.4	0.2	5.1	48.9	24.6	7.4	7.4	1.84
18	3.1	0.6	14.1	23.2	14.8	14.6	29.7	2.4
19	2.8	1	39.3	32.3	4.3	3.2	17.1	1.77
20	2.9	0.7	2.6	4.1	5.8	11.6	72.2	3.1
21	3.5	0.4	18.1	60.1	12.4	3.3	2.2	1.52
22	9.4	0.3	1.7	24.1	16.6	19	28.8	1.74
24	5	0.1	3.3	30.5	14.9	6.2	40	2
25	5.6	0.4	2.5	18.6	14.6	17.7	40.6	2.3
26	4.3	0.1	19.4	49.5	21.4	4.7	0.4	1.67
27	2.8	< 0.1	6.2	73.2	13.4	3.5	0.9	1.39
29	5.5	0.4	5	23.7	25.1	16.3	24.1	2.5
30	3.5	0.1	2.2	41.5	32.5	10.8	9.3	1.17

Site	Silt & Clay % (< 63 µm)	Very Fine Sand % (63 to < 125 µm)	Fine Sand % (125 to < 250 µm)	Medium Sand % (250 to < 500 µm)	Coarse Sand % (500 µm to < 1 mm)	Very Coarse Sand % (1 to < 2 mm)	Gravel % (≥ 2 mm)	AFDW %
31	3.5	< 0.1	0.1	41.5	54.3	0.7	< 0.1	0.76
32	8.1	< 0.1	10.3	59.6	12.3	4.1	5.5	1.65
33	6.1	0.2	3.8	36.9	30.8	6.9	15.3	1.57
35	4.3	0.7	18.7	34.8	14.1	8	19.5	2.4
36	5	0.2	2.7	37.4	18.1	13.1	23.5	1.42
37	6.1	0.9	45.1	39.8	2.9	1.8	3.3	1.23
38	6.4	2.6	21.8	13	22.8	16	17.4	2.6
42	3.3	< 0.1	< 0.1	37.7	49.7	8.2	1	0.99
44	7.2	0.7	29	30.8	9.8	6.4	16.3	2.2

Appendix 5. Abundances of macrofauna in benthic grab samples from 31 sites (sites 1–22) across the Hananui proposal area

Phylum	Family	Taxon	1	5	6	8	9	10	13	14	15	16	17	18	19	20	21	22
Porifera	Unclassified	Porifera												1				
Porifera	Scyettidae	*Sycon* sp.																1
Cnidaria	Unclassified	Hydrozoa														1	1	
Cnidaria	Unclassified	Ceriantharia	2															
Nemertea	Unclassified	Nemertea				1		1		2		9			1			
Nematoda	Unclassified	Nematoda	9	1	4		5		10	9		11	1	15	7	2	8	
Sipuncula	Unclassified	Sipuncula																
Gastropoda	Unclassified	Gastropoda	3			2						1						1
Bivalvia	Unclassified	Bivalvia																2
Bivalvia	Lasaeidae	Lasaeidae														1		
Bivalvia	Ungulinidae	*Diplodonta zelandica*																
Bivalvia	Tellinidae	*Elliptotellina urinatoria*																
Bivalvia	Psammobiidae	*Gari convexa*						1										
Bivalvia	Psammobiidae	*Gari lineolata*														1		
Bivalvia	Psammobiidae	*Gari* sp.																
Bivalvia	Psammobiidae	*Gari stangeri*																1
Bivalvia	Glycymerididae	*Glycymeris modesta*			2							1						
Bivalvia	Nuculidae	*Nucula nitidula*															2	
Bivalvia	Veneridae	*Ruditapes largillierti*		1														
Bivalvia	Mactridae	*Scalpomactra scalpellum*															1	
Bivalvia	Glycymerididae	*Tucetona laticostata*																
Annelida	Unclassified	Oligochaeta	24	7	3	12	1	1	1		1	17	2	5	4	3	13	5

Phylum	Family	Taxon	1	5	6	8	9	10	13	14	15	16	17	18	19	20	21	22
Annelida	Orbiniidae	*Orbinia papillosa*	1															
Annelida	Orbiniidae	*Scoloplos* sp.			1			1		1					2	1		
Annelida	Paraonidae	Paraonidae	1	1	5			2				3			1	2		
Annelida	Paraonidae	*Aricidea* sp.																
Annelida	Spionidae	*Prionospio* sp.	1			4		2				1				5		
Annelida	Spionidae	*Prionospio tridentata*																
Annelida	Spionidae	*Spio* sp.																3
Annelida	Chaetopteridae	*Phyllochaetopterus* sp.										1						
Annelida	Capitellidae	*Barantolla lepte*										1						
Annelida	Capitellidae	*Heteromastus filiformis*																
Annelida	Capitellidae	*Heteromastus* sp.																
Annelida	Maldanidae	Maldanidae	1										2					
Annelida	Opheliidae	*Armandia maculata*		1														
Annelida	Phyllodocidae	Phyllodocidae											1					
Annelida	Piligaridae	Pilargidae														1		
Annelida	Polynoidae	Polynoidae	1															
Annelida	Sigalionidae	Sigalionidae	1		1	1				1					1		2	
Annelida	Sigalionidae	*Sigalion* sp.																
Annelida	Pisionidae	Pisionidae											1					
Annelida	Hesionidae	Hesionidae				1						1		2		1	1	
Annelida	Syllidae	Exogoninae	6	2	3			1		2		3		1	1	7	6	4
Annelida	Syllidae	Syllidae	2			4	2	1				9	1			1	4	1
Annelida	Syllidae	*Exogone* sp.							1									
Annelida	Glyceridae	Glyceridae		1					1			1					1	
Annelida	Goniadidae	Goniadidae				1												
Annelida	Eunicidae	*Lysidice* sp.																

Phylum	Family	Taxon	1	5	6	8	9	10	13	14	15	16	17	18	19	20	21	22
Annelida	Lumbrineridae	Lumbrineridae														1	1	
Annelida	Dorvilleidae	Dorvilleidae			1											1	4	
Annelida	Cirratulidae	Cirratulidae	3						3			2				3	7	3
Annelida	Flabelligeridae	Flabelligeridae	2															
Annelida	Terebellidae	Terebellidae	5	1	2								2			4	4	1
Annelida	Sabellidae	*Euchone* sp.							1									
Annelida	Sabellidae	Sabellidae													1	1		
Arthropoda	Tanaidae	Tanaidacea											1					
Arthropoda	Unclassified	Mysidacea																
Arthropoda	Unclassified	Cumacea								1		1						
Arthropoda	Munnidae	Munnidae	2									1				3		1
Arthropoda	Anthuridae	Anthuridae				2												
Arthropoda	Unclassified	Asellota										1						
Arthropoda	Unclassified	Isopoda																
Arthropoda	Aoridae	Aoridae																
Arthropoda	Caprellidae	Caprellidae		1								3		1		1		
Arthropoda	Dexaminidae	Dexaminidae														21		
Arthropoda	Haustoriidae	Haustoriidae																
Arthropoda	Lysianassidae	Lysianassidae	1									2				1		
Arthropoda	Phoxocephalidae	Phoxocephalidae	1	2	1	1	1		4	3		1		2	1	4		
Arthropoda	Stenothoidae	Stenothoidae														2		
Arthropoda	Unclassified	Amphipoda	22	4			1	1		1		2				18	11	2
Arthropoda	Hymenosomatidae	*Halicarcinus* sp.															1	
Arthropoda	Portunidae	*Nectocarcinus benetti*																1
Arthropoda	Unclassified	Brachyura															1	
Arthropoda	Cylindroleberididae	*Leuroleberis zealandica*			1													

Phylum	Family	Taxon	1	5	6	8	9	10	13	14	15	16	17	18	19	20	21	22
Arthropoda	Cylindroleberididae	*Parasterope quadrata*																
Arthropoda	Paracyprididae	*Phylctenophora zealandica*																
Arthropoda	Unclassified	Ostracoda														1		
Arthropoda	Unclassified	Copepoda																
Arthropoda	Unclassified	Pycnogonida										1					2	
Arthropoda	Pycnogonidae	Pycnogonidae																
Arthropoda	Unclassified	Acarina																1
Phoronida	Unclassified	Phoronida			2													
Bryozoa	Unclassified	Bryozoa							3							1	1	
Echinodermata	Echinoidae	*Echinocardium* spat													1			
Echinodermata	Unclassified	Asteroidea																
Echinodermata	Unclassified	Ophiuroidea															2	
Echinodermata	Chiridotidae	*Taeniogyrus dendyi*										1						
Chordata	Unclassified	Ascidiacea																
Chordata	Mullidae	Mullidae																

Appendix 5 (cont.). Abundances of macrofauna in benthic grab samples from 31 sites (sites 24–44) across the Hananui proposal area

Phylum	Family	Taxon	24	25	26	27	29	30	31	32	33	35	36	37	38	42	44
Porifera	Unclassified	Porifera															
Porifera	Scyettidae	*Sycon* sp.															
Cnidaria	Unclassified	Hydrozoa													1		
Cnidaria	Unclassified	Ceriantharia															
Nemertea	Unclassified	Nemertea			1		7				1	3		2			2
Nematoda	Unclassified	Nematoda		4	5	1	17	3	3	1	1	25	8	5	6	1	15
Sipuncula	Unclassified	Sipuncula					1										
Gastropoda	Unclassified	Gastropoda	1				1	1		1	1			1	1	1	
Bivalvia	Unclassified	Bivalvia															
Bivalvia	Lasaeidae	Lasaeidae													1		
Bivalvia	Ungulinidae	*Diplodonta zelandica*											1				
Bivalvia	Tellinidae	*Elliptotellina urinatoria*			2											1	
Bivalvia	Psammobiidae	*Gari convexa*															
Bivalvia	Psammobiidae	*Gari lineolata*												1	3		
Bivalvia	Psammobiidae	Gari sp.					1										
Bivalvia	Psammobiidae	*Gari stangeri*															
Bivalvia	Glycymerididae	*Glycymeris modesta*										1					
Bivalvia	Nuculidae	*Nucula nitidula*															
Bivalvia	Veneridae	*Ruditapes largillierti*															
Bivalvia	Mactridae	*Scalpomactra scalpellum*															
Bivalvia	Glycymerididae	*Tucetona laticostata*													2		
Annelida	Unclassified	Oligochaeta	2	5	2	2	13	2	1	1	2	7	3	1	6	1	6

Phylum	Family	Taxon	24	25	26	27	29	30	31	32	33	35	36	37	38	42	44
Annelida	Orbiniidae	*Orbinia papillosa*															
Annelida	Orbiniidae	*Scoloplos* sp.			1												
Annelida	Paraonidae	Paraonidae										9					2
Annelida	Paraonidae	*Aricidea* sp.														12	
Annelida	Spionidae	*Prionospio* sp.					1										
Annelida	Spionidae	*Prionospio tridentata*		1													
Annelida	Spionidae	*Spio* sp.		17			8					1			3		7
Annelida	Chaetopteridae	*Phyllochaetopterus* sp.			1		1					1					
Annelida	Capitellidae	*Barantolla lepte*															
Annelida	Capitellidae	*Heteromastus filiformis*															3
Annelida	Capitellidae	*Heteromastus* sp.													1		
Annelida	Maldanidae	Maldanidae															
Annelida	Opheliidae	*Armandia maculata*												1			
Annelida	Phyllodocidae	Phyllodocidae															1
Annelida	Piligaridae	Pilargidae															
Annelida	Polynoidae	Polynoidae															
Annelida	Sigalionidae	Sigalionidae		1								1					
Annelida	Sigalionidae	*Sigalion* sp.													1		
Annelida	Pisionidae	Pisionidae															
Annelida	Hesionidae	Hesionidae		1								1					1
Annelida	Syllidae	Exogoninae		5		3	5		8	1	1	4		1			12
Annelida	Syllidae	Syllidae	1	6			3	1		1	1	2		8	15	1	
Annelida	Syllidae	*Exogone* sp.													6		
Annelida	Glyceridae	Glyceridae		1		1				1				1			1
Annelida	Goniadidae	Goniadidae															
Annelida	Eunicidae	*Lysidice* sp.												1			

Phylum	Family	Taxon	24	25	26	27	29	30	31	32	33	35	36	37	38	42	44
Annelida	Lumbrineridae	Lumbrineridae													1		
Annelida	Dorvilleidae	Dorvilleidae												1			
Annelida	Cirratulidae	Cirratulidae		5								6		15	2		5
Annelida	Flabelligeridae	Flabelligeridae															
Annelida	Terebellidae	Terebellidae		3			1	1			1	4	1	2	7	3	1
Annelida	Sabellidae	*Euchone* sp.															
Annelida	Sabellidae	Sabellidae										1					
Arthropoda	Tanaidae	Tanaidacea					1										
Arthropoda	Unclassified	Mysidacea															1
Arthropoda	Unclassified	Cumacea		2			2								2		
Arthropoda	Munnidae	Munnidae					1					2					
Arthropoda	Anthuridae	Anthuridae	1														2
Arthropoda	Unclassified	Asellota															
Arthropoda	Unclassified	Isopoda													1		
Arthropoda	Aoridae	Aoridae										1			2	1	3
Arthropoda	Caprellidae	Caprellidae															
Arthropoda	Dexaminidae	Dexaminidae															
Arthropoda	Haustoriidae	Haustoriidae						1									
Arthropoda	Lysianassidae	Lysianassidae										2				1	
Arthropoda	Phoxocephalidae	Phoxocephalidae		1	3							4			5		1
Arthropoda	Stenothoidae	Stenothoidae															
Arthropoda	Unclassified	Amphipoda	9	2		1	3	1	1		1	2			8		
Arthropoda	Hymenosomatidae	*Halicarcinus* sp.															
Arthropoda	Portunidae	*Nectocarcinus benetti*															
Arthropoda	Unclassified	Brachyura															
Arthropoda	Cylindroleberididae	*Leuroleberis zealandica*										1					

Phylum	Family	Taxon	24	25	26	27	29	30	31	32	33	35	36	37	38	42	44
Arthropoda	Cylindroleberididae	*Parasterope quadrata*															1
Arthropoda	Paracyprididae	*Phylctenophora zealandica*					1										
Arthropoda	Unclassified	Ostracoda															
Arthropoda	Unclassified	Copepoda					2										
Arthropoda	Unclassified	Pycnogonida															
Arthropoda	Pycnogonidae	Pycnogonidae													6		2
Arthropoda	Unclassified	Acarina															
Phoronida	Unclassified	Phoronida															
Bryozoa	Unclassified	Bryozoa						1							1		
Echinodermata	Echinoidae	*Echinocardium* spat															
Echinodermata	Unclassified	Asteroidea													1		
Echinodermata	Unclassified	Ophiuroidea										1					1
Echinodermata	Chiridotidae	*Taeniogyrus dendyi*															1
Chordata	Unclassified	Ascidiacea													1		
Chordata	Mullidae	Mullidae													5		

Appendix 6. Sediment physical and chemical properties and community-level macrofauna variables for grab samples from 31 sites across the Hananui proposal area

Indices include: Shannon–Wiener diversity index (SWDI), Pielou's evenness index (Evenness), Margalef richness index (Richness), AMBI biotic coefficient (AMBI) and M-AMBI ecological quality ratio (M-AMBI). NR = not recorded, NC = not calculated.

Site	Depth (m)	Organic matter (% AFDW)	Redox (Eh_{NHE}, mV)	Bacterial mat	Odour	Abundance (no./core)	No. taxa (no./core)	Evenness	Richness	SWDI (*H'*)	AMBI	M-AMBI
NTS-1	35	2.5	399	no	no	88	19	0.77	4.02	2.26	3.32	0.59
NTS-5	36	1.4	347	no	no	22	11	0.87	3.24	2.09	2.89	0.52
NTS-6	38	1.5	351	no	no	26	12	0.93	3.38	2.32	2.55	0.59
NTS-8	35	2.0	351	no	no	29	10	0.81	2.67	1.86	3.38	0.44
NTS-9	32	1.5	352	no	no	10	5	0.84	1.74	1.36	3.00	0.39
NTS-10	35	2.5	362	no	no	11	9	0.98	3.34	2.15	2.45	0.56
NTS-13	29	1.3	334	no	no	24	8	0.82	2.20	1.71	2.44	0.50
NTS-14	32	1.4	351	no	no	20	8	0.82	2.34	1.70	1.63	0.52
NTS-15	34	1.0	356	no	no	1	1	NC	NC	0.00	6.00	0.01
NTS-16	38	2.4	355	no	no	74	23	0.82	5.11	2.57	2.77	0.73
NTS-17	34	1.8	351	no	no	11	8	0.97	2.92	2.02	2.69	0.52
NTS-18	31	2.4	342	no	no	27	7	0.71	1.82	1.39	2.98	0.40
NTS-19	NR	1.8	352	no	no	20	10	0.85	3.00	1.97	2.50	0.52
NTS-20	38	3.1	358	no	no	88	26	0.81	5.58	2.63	1.93	0.85
NTS-21	35	1.5	321	no	no	73	20	0.87	4.43	2.61	2.85	0.69
NTS-22	37	1.7	396	no	no	27	14	0.93	3.94	2.45	3.88	0.55

Site	Depth (m)	Organic matter (% AFDW)	Redox (Eh_{NHE}, mV)	Bacterial mat	Odour	Abundance (no./core)	No. taxa (no./core)	Evenness	Richness	SWDI (*H'*)	AMBI	M-AMBI
NTS-24	31	2.0	412	no	no	14	5	0.70	1.52	1.13	3.50	0.31
NTS-25	35	2.3	409	no	no	54	14	0.85	3.26	2.24	2.98	0.59
NTS-26	24	1.7	423	no	no	15	7	0.91	2.22	1.77	1.86	0.52
NTS-27	30	1.4	409	no	no	8	5	0.93	1.92	1.49	3.81	0.34
NTS-29	37	2.5	406	no	no	69	18	0.82	4.02	2.36	2.69	0.65
NTS-30	34	1.2	416	no	no	11	8	0.95	2.92	1.97	3.63	0.46
NTS-31	30	0.8	414	no	no	13	4	0.74	1.17	1.03	4.13	0.30
NTS-32	22	1.7	417	no	no	6	6	1.00	2.79	1.79	3.50	0.42
NTS-33	32	1.6	382	no	no	9	8	0.98	3.19	2.04	4.25	0.42
NTS-35	32	2.4	409	no	no	79	21	0.81	4.58	2.47	2.47	0.72
NTS-36	32	1.4	411	no	no	13	4	0.74	1.17	1.03	3.56	0.30
NTS-37	32	1.2	413	no	no	40	13	0.77	3.25	1.99	3.27	0.52
NTS-38	35	2.6	407	no	no	88	25	0.89	5.36	2.86	2.01	0.87
NTS-42	37	1.0	418	no	no	22	9	0.72	2.59	1.59	1.80	0.53
NTS-44	35	2.2	409	no	no	68	20	0.84	4.50	2.53	2.75	0.71

Appendix 7. Depositional modelling methods

Here, we present detailed methods regarding the depositional modelling. An initial overview of the model mechanics is provided, followed by specific notes on sinking velocities, mass calculations, resuspension thresholds, random walk calculations and particle decay. At the end of this appendix is a table containing a full list of model input parameters.

A7.1 Overview of model mechanics

Virtual particles representative of fish faeces were released continuously at hourly intervals from virtual pens for a duration of 2 years. Once released, particles are dispersed by water currents as they sink to the seabed. Their trajectories are calculated from the current velocity (taken from the hydrodynamic model) at their initial locations, their fall rate (dependent on their particle type), and a random walk component that is added to replicate turbulent diffusive processes. Particle trajectories were calculated every 30 seconds during the 731-day model run. All particles in the model were assumed to decay exponentially with a half-life of 8 days (decay constant of K = -0.0867 day $^{-1}$) to simulate the effects of macrofaunal and microbial consumption processes, as well as shear forces from ocean currents (for details, see below).

At the end of every 30-second time-step, the calculated depths of all particles were checked and compared to the water depth at their respective locations. Any particles that fell lower than the seabed were brought vertically upwards to the seabed depth at their location. Once particles land on the seabed, they do not fall further but can be resuspended by currents if the shear velocity at their location exceeds a critical threshold value determined by the benthic substrate at their location. If this condition is met, relevant particles are given a vertical 'jump' back into the water column (using a similar formula to the random walk detailed in Section A7.4). At the next model time-step, these resuspended particles are redispersed from their new depths until they resettle on the seabed.

The modelled area was divided into a grid of 50 m × 50 m squares and the total particle mass in each square was recorded at the end of each day. Mean residual solids for each peak feed scenario was calculated by averaging the mass in each square during the 3 months surrounding the peak feed in each farm.

Data collected for flux were run using a separate simulation with resuspension and decay turned off, and once these particles were counted, they were removed from the simulation. Monthly flux was calculated by taking the mean daily flux in each square during the 3 months of each peak feed scenario, and then this was converted into monthly units (mean daily flux × 365 days yr^{-1} / 12 $mo{\cdot}yr^{-1}$).

A7.2 Distribution of sinking velocities

Faecal particles were modelled using three particle types, each with a unique sinking rate (Table A7.1) based on a distribution of faecal sinking rates for Atlantic salmon (*Salmo salar*) provided in Bannister et al. (2016). Most depositional modelling exercises of proposed finfish aquaculture in Aotearoa New

Zealand have taken the approach of using a single mean sinking velocity (0.032 $m·s^{-1}$) for all faecal pellets, although depositional modelling undertaken by DHI (2020) for a proposed kingfish farm in the Hauraki Gulf / Tīkapa Moana used two representative faecal particle types (regular and slow-sinking).

The single mean value approach was inherited from Cromey et al. (2002), who created the DEPOMOD model, an early particle-tracking model developed specifically for salmon farming that became industry standard. Continued research into salmon farm waste has highlighted that a skew in the distribution of sinking velocities leads to an underestimation of deposition directly underneath the pens (Bannister et al. 2016) and that modelling slow-sinking fine particles leads to better elucidation of far-field deposition (Law et al. 2014; Bannister et al. 2016).

Bannister et al. (2016) provide a distribution of six settling velocity ranges for Atlantic salmon faecal pellets that has been adopted in recent depositional modelling studies of salmon farms overseas (Carvajalino-Fernandez et al. 2020a; Keeley et al. 2024). In this work, we adopt three different particle sinking velocities based on this distribution (Table A7.1). Fast- and medium-sinking particles are taken directly from the two fastest categories provided in Bannister et al. (2016) by adopting the midpoint value of the categories' respective velocity ranges. The sinking velocity of the slow-sinking particle type is based on a weighted average of the remaining slower-sinking particle types (0.0089 $m·s^{-1}$). Adopting three different particle types, rather than the six types detailed in Bannister et al. (2016), was necessary to constrain the computational effort of running the model.

Table A7.1. Faecal particle sinking velocities and their contribution to the total faecal mass released from pens, based on data in Bannister et al. (2016).

Faecal particle type	Sinking velocity	% of total mass
Fast-sinking	0.075 $m·s^{-1}$	66
Medium-sinking	0.0375 $m·s^{-1}$	19
Slow-sinking	0.0089 $m·s^{-1}$	15

A7.3 Determining mass efflux from pens

The mass of total faeces and wasted feed pellets expelled from the pens is a function of digestibility coefficient, feed moisture content and the percentage of wasted feed (Table A7.2). For this work, we have adopted values from Cromey et al. (2002) that have been used in other depositional modelling works for Chinook salmon (*Oncorhynchus tshawytscha*) aquaculture in Aotearoa New Zealand (Keeley et al. 2013; Elvines et al. 2021a, 2021b).

Table A7.2. Values and methods used to determine the total mass of faecal particles and wasted feed pellets released from pens in the farm.

Parameter	Value
Digestibility coefficient	85%
Water content of feed	9%
Wasted feed %	3%
% of feed excreted as faeces to seabed (dw)	13.24% ((1 – 0.09) × (1 – 0.03) × (1 – 0.85)) = 0.1324
% of feed wasted (dw)	2.73% (1 – 0.09) × 0.03 = 0.0273
% of feed excreted as fast-sinking faeces (dw)	8.74% 0.1324 × 0.66 = 0.0874
% of feed excreted as medium-sinking faeces (dw)	2.52% 0.1324 × 0.19 = 0.0252
% of feed excreted as slow-sinking faeces (dw)	1.99% 0.1324 × 0.15 = 0.0199

The digestibility coefficient (85%) provided by Cromey et al. (2002) is for Atlantic salmon, although this value agrees well with reported digestibility values for Chinook salmon fed a fish-based diet (84–88%; Hajen et al. 1993). The moisture content value (9%) is slightly higher than values provided in feed tables by Skretting (Orient Plus and Glory N feed tables, Jennie de Haan, Skretting, pers. comm., 24 April 2025) for Chinook salmon feed pellets (8%). Our adoption of this higher moisture content equates to a 0.15% underestimation of total faecal mass and a 0.03% underestimation of wasted feed pellets. We highlight that this underestimation is more than compensated for by our 7% and 8% overestimates of total feed for Stage 1 and Stage 2, respectively. Total annual feed was overestimated in our model because of difficulties in balancing variable feed inputs across farms operating asynchronously. We believe this overestimate of water content is inconsequential in the context of the overall modelling approach.

A7.4 Resuspension thresholds for different habitat types

Factors governing resuspension are complex and depend not only on particle size and density, but also substrate type (Law et al. 2016). Particle resuspension is caused by turbulent fluctuations in water currents near the seabed, quantitatively described as bottom shear stress (Carvajalino-Fernandez et al. 2020a). Bottom shear stress (τ_b with units of Pascals) is often expressed in terms of shear velocity (u_* with units of m·s^{-1}), where $\tau_b = \rho u_*^2$ and ρ is the fluid density, which for seawater is ~ 1,025 kg·m^{-3}.

Particles resuspend when the shear velocity at their location on the seabed exceeds a critical value ($u_{*\text{crit}}$), called the critical shear velocity. Above this value, particles detach from the seabed and are lifted back into suspension. Early simple approaches to model particle resuspension used single critical shear velocities for all particles in all substrates (0.0042 m·s^{-1}; Cromey et al. 2002), or separate respective values for faecal particles and wasted feed particles in all substrates (0.009 m·s^{-1} and 0.015 m·s^{-1}, respectively in Bennett et al. [2022] and Elvines et al. [2021a, 2021b]). However, research has shown that there are significant differences in the shear velocity required to resuspend particles in different substrate types (Law et al. 2016; Carvajalino-Fernandez et al. 2020a, 2020b).

Through a series of experiments in a horizontal flume, Carvajalino-Fernandez et al. (2020b) provide critical shear velocity values for the following substrates: rock slates, mud, sand and fragmented rock. Modelling here incorporates the findings of Carvajalino-Fernandez et al. (2020b) to set spatially varying critical shear velocities according to habitat type. The hydrodynamic hindcast model (Oceanum 2025) used in the depositional modelling here was parametrised according to substrate type using the categories mud, sand, gravel and reef. We applied spatially varying critical shear velocities from Carvajalino-Fernandez et al. (2020b) to hydrodynamic model cells by mapping these categories according to Table A7.3.

Table A7.3. Critical shear velocities used for different substrates.

Substrate classification in hydrodynamic model (Oceanum 2025)	Substrate classification in Carvajalino-Fernandez et al. (2020b)	Critical shear velocity
Mud	Mud	0.008 m·s^{-1}
Sand	Sand	0.0105 m·s^{-1}
Gravel	Midpoint of sand and fragmented rock	0.013 m·s^{-1}
Reef	Fragmented rock	0.0163 m·s^{-1}

No critical shear velocity values were provided in Carvajalino-Fernandez et al. (2020b) for gravel or reef substrates. The fragmented rock substrate described in Carvajalino-Fernandez et al. (2020b) was used for reef substrates because the median diameter of this substrate (200–250 mm) is of similar length scale to the bryozoan-sponge reefs within the proposal area. The critical shear velocity for gravel was estimated by taking the midpoint of sand and fragmented rock on the basis that the drag value for gravel in the hydrodynamic model was the midpoint of sand and reef (Oceanum 2025).

With no information available regarding how critical shear velocities vary for wasted feed particles in different habitats, we adopted the uniform value of 0.015 m·s^{-1} for wasted feed particles used in Elvines et al. (2021a), which is, in turn, based on Law (2019). We note that wasted feed particles account for 17% of the total mass of solid waste expelled from the pens and that their higher sinking velocity leads them to be dispersed less than faecal particles.

A7.5 Random walk and resuspension jump calculations

Initial particle advection is calculated using water velocities at the respective particle locations taken from the hydrodynamic model. A random walk dispersion component is then added to each particle trajectory to account for turbulent diffusive processes smaller than the hydrodynamic model resolution. The random walk in the horizontal direction is a function of the horizontal dispersion coefficient (A_H = 0.1 m^2·s^{-1}), while the vertical random walk is a function of the vertical diffusivity profile (κ_z) within the hydrodynamic model. Random walk distances along all three axes ($x_{\text{rand}}, y_{\text{rand}}, z_{\text{rand}}$) are calculated using methods provided in Lynch et al. (2014):

$$(x_{\text{rand}}, y_{\text{rand}}, z_{\text{rand}}) = \left(\text{randn}\sqrt{2A_{\text{H}}\Delta t}\,, \text{randn}\sqrt{2A_{\text{H}}\Delta t}\,, \frac{\partial\kappa_z}{\partial z} + \text{randn}\sqrt{2\kappa_z\Delta t}\,\right).$$

Here, randn is a normally distributed random number with a mean of 0 and variance of 1, A_{H} is the constant horizontal dispersion coefficient (0.1 $m^2 \cdot s^{-1}$), Δt is the model time-step (30 s), κ_z is the vertical diffusivity, and $\partial\kappa_z/\partial z$ is a vertical velocity associated with turbulent processes (Visser 1997).

Similarly, if resuspension conditions are met ($u_* > u_{*\text{crit}}$), particles are lifted from the seabed a distance of:

$$z_{\text{jump}} = \text{abs}(\text{randn})\sqrt{2\kappa_z\Delta t}.$$

Taking the absolute value of the random number in this formula ensures an upwards jump (negative values would result in a downwards jump).

A7.6 Particle mass decay

The mass of all particles in the model is assumed to decay exponentially with a half-life of 8 days (decay constant, *K* = -0.0867 day^{-1}) and the decay process starts as soon as the particles are released. This decay constant was approximated from Keeley et al. (2019) for a salmon farm in Norway and is based on mean measured infaunal respiration rates under the pens. Particles will also be consumed by microbial oxidation and be broken down by shear forces from ocean currents as they disperse. These additional processes depend on multiple biological and oceanographic factors, many of which are temperature dependent and seasonal. Depositional modelling undertaken in Tory Channel using this decay constant showed reasonable agreement with benthic field measurements (Elvines et al. 2021b). We note that using a different decay rate would make relating residual solids values to ecological thresholds in Elvines et al. (2021b) complicated as these thresholds were determined using this decay rate.

List of model parameters

Table A7.4. Parameters used in depositional modelling.

Model parameter	Value
Time-step	30 sec
Model start date	1 June 2018, 01:00:00
Model end date	31 May 2020, 23:00:00
Model duration	731 days
Pen geometries	
Pen depth	16 m
Pen circumference	168 m
Pen diameter	54 m
Particle release rates	
Faeces fast fraction (66% total faeces)	10 particles·pen^{-1}·hr^{-1}
Faeces medium fraction (19% total faeces)	10 particles·pen^{-1}·hr^{-1}
Faeces slow fraction (15% total faeces)	10 particles·pen^{-1}·hr^{-1}
Wasted feed pellets	10 particles·pen^{-1}·hr^{-1}
Sinking velocities	
Faeces fast fraction (66% total faeces)	0.075 m·s^{-1}
Faeces medium fraction (19% total faeces)	0.0375 m·s^{-1}
Faeces slow fraction (15% total faeces)	0.0089 m·s^{-1}
Wasted feed pellets	0.095 m·s^{-1}
Horizontal dispersion coefficient	0.1 m^{2}·s^{-1}
Vertical dispersion	Profile taken from hydrodynamic model
Critical shear velocities	
Faeces in mud	0.008 m·s^{-1}
Faeces in sand	0.0105 m·s^{-1}
Faeces in gravel	0.013 m·s^{-1}
Faeces in reef	0.0163 m·s^{-1}
Wasted feed, all substrates	0.015 m·s^{-1}
Mass and decay parameters	
Estimated feed wastage	3%
Estimated feed digestibility	85%
Estimated feed water content	9%
% of feed excreted as faeces to seabed (dw)	13.24%
% of feed wasted to seabed (dw)	2.73%
Decay rate half-life	8 days
Maximum allowed particle age	56 days (less than 1% original mass)

Appendix 8. Further information on wave resuspension

Rakiura shields the Hananui site and Foveaux Strait from the largest wave events in the region, which are predominantly from the west (Figure A8.1). The shallower waters in the strait also act to attenuate large, long-period waves. Based on a 37-year wave hindcast model, Campos et al. (2020) calculated the mean significant wave height at the site to be in the order of 1 m and the mean peak period to be in the realm of 8–10 seconds. Depths at the site range from ~ 20 m nearshore to ~ 40 m offshore.

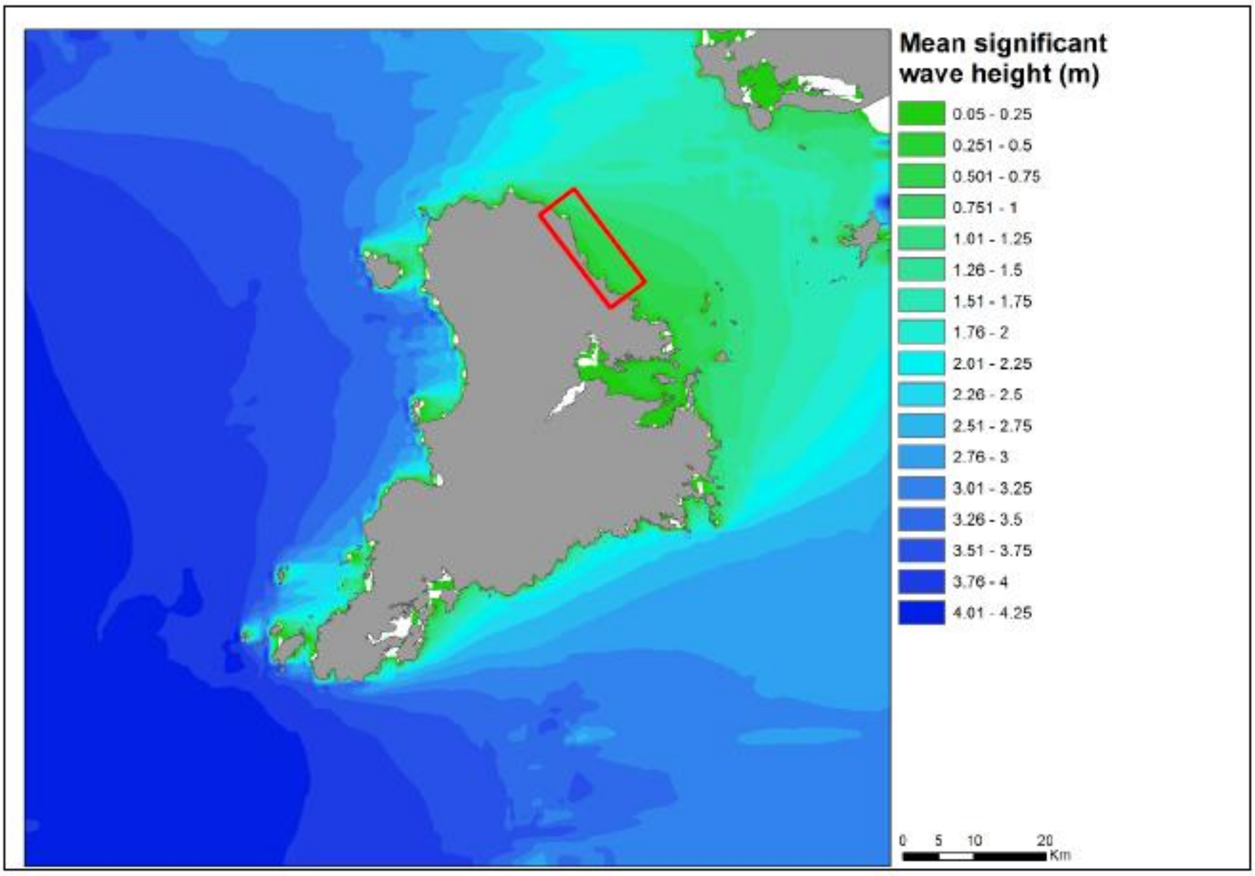

Figure A8.1. Mean significant wave height in the Rakiura region based on a 37-year wave hindcast model. Source: Campos et al. (2022).

As waves pass over sections of water, their energy causes the water to move in an orbital motion. The orbital velocity of water at the top of a crest is travelling in the opposite direction to the water at the bottom of a trough. This causes suspended particles to travel in a circular trajectory that starts and finishes in the same location. In this respect, wave orbitals will have negligible effect on particle trajectories.

However, waves can resuspend seabed particles when their near-bed orbital velocities are high, enabling further dispersion by currents. Larger waves (i.e. with a greater wave height) carry more energy and generate faster orbital velocities. These velocities decay with depth, and the rate of decay depends on the wave period: longer-period waves decay more slowly with depth than shorter-period waves. As a result, for waves of equal height, long-period waves produce higher orbital velocities at the seabed than short-period waves. The capacity of waves to resuspend sediment is therefore a function of wave height, wave period and water depth. Including waves in the model would cause particles to resuspend more frequently.

Using hourly wave orbital velocity hindcast data (1 km gridded data) for the same time period as the hydrodynamic model, we calculated the number of instances in the 2-year model period where bed shear stress exceeded critical shear stress using 'currents only' and using 'currents and waves'. The additional bed shear stress from wave orbitals was estimated using methods provided in You (1995). Figure A8.2 shows that the inclusion of waves in determining resuspension causes the percentage of time that waste is resuspended to increase by about 8% within the proposal area and the outer limit of effects zone (0.7 g·m^{-2} contour, denoted by red line in Figure A8.2). Higher increases in resuspension frequency occur on the peaks of sand waves (shallower), as well as offshore from the 0.7 g·m^{-2} solids contour, where waves are larger.

The effect of this is that particles will resuspend more frequently than suggested by the modelling and will be dispersed slightly further afield. This will likely dilute both areas of high deposition underneath and around the farms and the far-field footprint. Figure A8.2 indicates that the effect of waves is to increase resuspension everywhere.

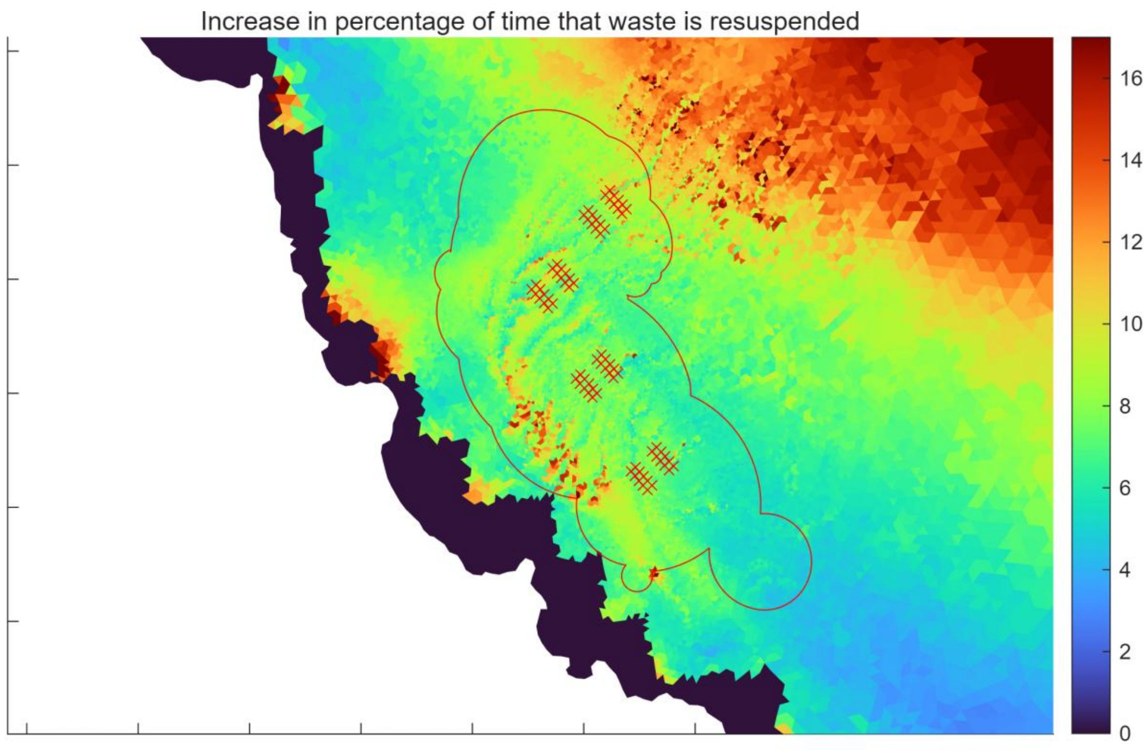

Figure A8.2. Increase in percentage of time that faecal waste is resuspended by including wave-related bed shear stress. The outer limit of effects zone (OLE) is indicated with a red line and Stage 2 pen locations are indicated by red crosses (X).

10. References

Anderson T, Morrison M, MacDiarmid A, Clark M, D'Archino R, Nelson W, Tracey D, Gordon D, Read G, Kettles H, et al. 2019. Review of New Zealand's key biogenic habitats. Wellington: National Institute of Water and Atmospheric Research. NIWA Client Report No. 2018139WN. Prepared for the Ministry for the Environment.

Angel DL, Eden N, Breitstein S, Yurman A, Katz T, Spanier E. 2002. *In situ* biofiltration: a means to limit the dispersal of effluents from marine finfish cage aquaculture. Hydrobiologia. 469(1):1–10.

[Aquenal] Aquenal Pty Ltd. 2024. Synthesis of the Storm Bay Broadscale Environmental Monitoring Program, 2019–2023. Kingston (TAS): Aquenal Pty Ltd. Report prepared for Salmon Tasmania.

Bannister RJ, Johnsen IA, Hansen PK, Kutti T, Asplin L. 2016. Near- and far-field dispersal modelling of organic waste from Atlantic salmon aquaculture in fjord systems. ICES Journal of Marine Science. 73(9):2408–2419.

Batson PB, Probert PK. 2000. Bryozoan thickets off Otago Peninsula. Wellington: Ministry of Fisheries. New Zealand Fisheries Assessment Report 2000/46.

Beentjes MP, Miller A, Kater D. 2019. Relative abundance, size and age structure, and stock status of blue cod in Foveaux Strait in 2018. Wellington: Fisheries New Zealand. New Zealand Fisheries Assessment Report 2019/13.

Bell JJ. 2008. The functional roles of marine sponges. Estuarine, Coastal and Shelf Science. 79(3):341–353.

Benedikt G, Edzer P, Gerard H. 2016. Spatio-Temporal Interpolation using gstat. The R Journal. 8(1):204–218.

Bennett H, Smeaton M, Jary M, Taylor D. 2018. Aquaculture site scoping: northern Stewart Island Stage 1b. Nelson: Cawthron Institute. Cawthron Report No. 3221. Prepared for Ministry for Primary Industries.

Bennett H, Smeaton M, McGrath E, Newcombe E. 2022. Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura. Nelson: Cawthron Institute. Cawthron Report No. 3315B. Prepared for Ngāi Tahu Seafood Resources.

Bennett H, Smeaton M, McGrath E, Newcombe E, Floerl L, Major R, Casanovas P. 2022. Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura. Nelson: Cawthron Institute. Cawthron Report No. 3315B. Prepared for Ngāi Tahu Seafood Resources.

Bergvik M, Stensås L, Handå A, Reitan KI, Strand Ø, Olsen Y. 2019. Incorporation of feed and fecal waste from salmon aquaculture in great scallops (*Pecten maximus*) co-fed by different algal concentrations. Frontiers in Marine Science. 5(2018).

Black KD, Hansen PK, Holmer M. 2008. Salmon Aquaculture Dialogue: working group report on benthic impacts and farm siting. Washington (DC): World Wildlife Fund.

Borja A, Franco J, Perez V. 2000. Guidelines for the use of AMBI (AZTI's marine biotic index) in the assessment of the benthic ecological quality. Marine Pollution Bulletin. 50:787–789.

Bradstock M, Gordon D. 1983. Coral-like bryozoan growths in Tasman Bay, and their protection to conserve commercial fish stocks. New Zealand Journal of Marine and Freshwater Research. 17(2):159–163.

Brown CJ, Smith SJ, Lawton P, Anderson JT. 2011. Benthic habitat mapping: a review of progress towards improved understanding of the spatial ecology of the seafloor using acoustic techniques. Estuarine, Coastal and Shelf Science. 92(3):502–520).

Campos C, Smeaton M, Bennett H, Mackenzie L, Scheel M, Vennell R, Newcombe E, Knight B. 2020. Assessment of water column effects associated with farming salmon offshore of northern Stewart Island / Rakiura. Nelson: Cawthron Institute. Cawthron Report No. 3326. Prepared for Ngāi Tahu Seafood Resources.

Carbines G, Jiang W, Beentjes MP. 2004. The impact of oyster dredging on the growth of blue cod, *Parapercis colias*, in Foveaux Strait, New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems. 14(5):491–504.

Carvajalino-Fernandez MA, Saevik PN, Johnsen IA, Albretsen J, Keeley NB. 2020a. Simulating particle organic matter dispersal beneath Atlantic salmon fish farms using different resuspension approaches. Marine Pollution Bulletin. 161(Pt B):111685.

Carvajalino-Fernandez MA, Keeley NB, Fer I, Law BA, Bannister RJ. 2020b. Effect of substrate type and pellet age on the resuspension of Atlantic salmon faecal material. Aquaculture Environment Interactions. 12:117–129.

Champeau O. 2013. Literature review of ecological effects of aquaculture: effects from additives. A collaboration between Ministry for Primary Industries, Cawthron Institute and National Institute for Water and Atmospheric Research Ltd. Wellington: Ministry for Primary Industries.

Chen T, He T, Benesty M, Khotilovich V, Tang Y, Cho H, Chen K, Mitchell R, Cano I, Zhou T, et al. 2022. xgboost: Extreme Gradient Boosting. R package version 1.6.0.1. (accessed 15 October 2025). https://cran.r-project.org/web/packages/xgboost/index.html

Clark D, Taylor D, Keeley N, Dunmore R, Forrest R, Goodwin E. 2011. Assessment of effects of farming salmon at Ngamahau, Queen Charlotte Sound: deposition and benthic effects. Nelson: Cawthron Institute. Cawthron Report No. 1993. Prepared for the New Zealand King Salmon Company Ltd.

Cranfield H, Carbines G, Michael K, Dunn A, Stotter D, Smith D. 2001. Promising signs of regeneration of blue cod and oyster habitat changed by dredging in Foveaux Strait, southern New Zealand. New Zealand Journal of Marine and Freshwater Research. 35(5):897–908.

Cranfield HJ, Manighetti B, Michael KP, Hill A. 2003. Effects of oyster dredging on the distribution of bryozoan biogenic reefs and associated sediments in Foveaux Strait, southern New Zealand. Continental Shelf Research. 23(14):1337–1357.

Cranfield HJ, Michael KP, Doonan IJ. 1999. Changes in the distribution of epifaunal reefs and oysters during 130 years of dredging for oysters in Foveaux Strait, southern New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems. 9(5):461–483.

Cranfield HJ, Rowden AA, Smith DJ, Gordon DP, Michael KP. 2004. Macrofaunal assemblages of benthic habitat of different complexity and the proposition of a model of biogenic reef habitat regeneration in Foveaux Strait, New Zealand. Journal of Sea Research. 52(2):109–125.

Cromey CJ, Nickell TD, Black KD. 2002. DEPOMOD – modelling the deposition and biological effects of waste solids from marine cage farms. Aquaculture. 214(1):211–239.

Cullen DJ. 1967. The submarine geology of Foveaux Strait. Wellington: Department of Scientific and Industrial Research. Bulletin 184; New Zealand Oceanographic Institute Memoir 33.

[DOC] Department of Conservation. 2019. NZCPS 2010 guidance note. Policy 11: Indigenous biological diversity (biodiversity). Wellington: Department of Conservation.

DHI. 2020. Pare Hauraki Kaimoana aquaculture development – inner Hauraki Gulf: numerical modelling, water column, benthic and surf impact assessments. Auckland: DHI. Report prepared for Pare Hauraki Kaimoana.

[DML] Discovery Marine Limited. 2022. Rakiura hydrographic survey 2022. Mount Maunganui: Discovery Marine Limited. Prepared for Ngāi Tahu Seafood.

Dunlop K, Harendza A, Bannister R, Keeley N. 2021. Spatial response of hard- and mixed-bottom benthic epifauna to organic enrichment from salmon aquaculture in northern Norway. Aquaculture Environment Interactions. 13:455–475.

Dunmore R. 2022. Reef environmental monitoring results for the New Zealand King Salmon Company Ltd salmon farms: 2021. Nelson: Cawthron Institute. Cawthron Report No. 3745. Prepared for New Zealand King Salmon Company Ltd.

Elvines DM, Hopkins GA, MacLeod CK, Ross DJ, Ericson JA, Ragg NLC, Copedo JS, White CA. 2024. Assimilation of fish farm wastes by the ecosystem engineering bivalve *Atrina zelandica*. Aquaculture Environment Interactions. 16:115–131.

Elvines D, McGrath E, Smeaton M. 2025a. Site evaluations for aquaculture settlement areas in [redacted] and [redacted]: Phase 2. Nelson: Cawthron Institute. Cawthron Report 4169A. Prepared for Ministry for Primary Industries.

Elvines D, McGrath E, Smeaton M. 2025b. Site evaluations for aquaculture settlement areas in [redacted], Nelson: Cawthron Institute. Cawthron Report 4183. Prepared for Te Ohu Kaimoana

Elvines D, Morrisey D, Smeaton M, Floerl L, Keeley N. 2021a. Assessment of seabed effects for proposed 'Blue Endeavour' (revised) open ocean salmon farm. Nelson: Cawthron Institute. Cawthron Report No. 3489. Prepared for The New Zealand King Salmon Co. Limited.

Elvines D, Smeaton M, Bennett H. 2021b. Relating VenOM depositional model outputs to ecological response; a case study using high-flow salmon farms in Tory Channel. Nelson: Cawthron Institute. Cawthron Report No. 3521. Prepared for The New Zealand King Salmon Co. Limited.

[EIANZ] Environment Institute of Australia and New Zealand. 2024. Ecological Impact Assessment. Module 1: Assigning ecological value to marine benthic habitats. Balwyn (VIC): Environment Institute of Australia and New Zealand. [accessed 30 September 2025]. https://www.eianz.org/document/item/8074

Findlay RH, Watling L. 1997. Prediction of benthic impact for salmon net-pens based on the balance of benthic oxygen supply and demand. Marine Ecology Progress Series. 155:147–157.

Fletcher L. 2015. Review of horse mussel (*Atrina zelandica*) biology and ecology with reference to Hauraki Gulf populations. Nelson: Cawthron Institute. Prepared for Ministry for Primary Industries. Cawthron Report No. 2721. 25 p.

Fletcher L, Bennett H, Elvines D, Preece M, Broekhuizen N, Ford R, Heath P, Murray C, Jorgensen E, Wade O, et al. 2022. Best management practice guidelines for salmon farms in the Marlborough Sounds. Part 1: benthic environmental quality standards and monitoring protocol (Version 1.2, August 2022). Wellington: Ministry for Primary Industries. New Zealand Aquatic Environment and Biodiversity Report No. 294.

Forrest B, Keeley N, Gillespie P, Hopkins G, Knight B, Govier D. 2007. Review of the ecological effects of marine finfish aquaculture: final report. Nelson: Cawthron Institute. Cawthron Report No. 1285. Prepared for Ministry of Fisheries.

Forrest BM, Johnston C. 2025. Hananui Aquaculture Project: marine biosecurity assessment of effects. Nelson: Salt Ecology. Salt Ecology Report 166. Prepared for Ngāi Tahu Seafood.

Freeman D, Schnabel K, Marshall B, Gordon D, Wing S, Tracey D, Hitchmough R. 2013. Conservation status of New Zealand marine invertebrates, 2013. Wellington: Department of Conservation. New Zealand Threat Classification Series 9.

Friedman J. 2001. Greedy function approximation: a gradient boosting machine. Annals of Statistics. 29(5):1189–1232.

George EM, Parrish CC. 2015. Invertebrate uptake of lipids in the vicinity of Atlantic salmon (*Salmo salar*) aquaculture sites in British Columbia. Aquaculture Research. 46(5):1044–1065.

Giles H, Baxter A, Taylor D, Elvines D, Jorgensen E, James M, Bunce M, Broekhuizen N, Wade O, Ford R, et al. 2021. Best practice guidelines for benthic and water quality monitoring of open ocean finfish culture in New Zealand. Wellington: Fisheries New Zealand. New Zealand Aquatic Environment and Biodiversity Report No. 278.

Hajen WE, Higgs DA, Beames RM, Dosanjh, BS. 1993. Digestibility of various feedstuffs by post-juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in sea water. 2. Measurement of digestibility. Aquaculture. 112(4):333–348.

Hall-Spencer J, White N, Gillespie E, Gillham K, Foggo A. 2006. Impact of fish farms on maerl beds in strongly tidal areas. Marine Ecology Progress Series. 326:1–9.

Hamoutene D. 2014. Sediment sulphides and redox potential associated with spatial coverage of *Beggiatoa* spp. at finfish aquaculture sites in Newfoundland, Canada. ICES Journal of Marine Science. 71(5):1153–1157.

Haugland BT, Armitage CS, Kutti T, Husa V, Skogen MD, Bekkby T, Carvajalino-Fernández MA, Bannister RJ, White CA, Norderhaug KM, et al. 2021. Large-scale salmon farming in Norway impacts the epiphytic community of *Laminaria hyperborea*. Aquaculture Environment Interactions. 13:81–100.

Hewitt JE, Pilditch CA. 2004. Short-term feeding responses of *Atrina zelandica* to suspended sediment concentrations: effects of environmental history and physiological state. Journal of Experimental Marine Biology and Ecology. 306:95–112.

Hill NAO, Michael KP, Frazer A, Leslie S. 2010. The utility and risk of local ecological knowledge in developing stakeholder-driven fisheries management: the Foveaux Strait dredge oyster fishery, New Zealand. Ocean & Coastal Management. 53(11):659–668.

Holmer M, Wildish D, Hargrave B. 2005. Organic enrichment from marine finfish aquaculture and effects on sediment biogeochemical processes. In: Hargrave B, editor. Environmental effects of marine finfish aquaculture. Berlin, Heidelberg: Springer. p. 181–206

Huang YC, Hsieh HJ, Huang SC, Meng PJ, Chen YS, Keshavmurthy S, Nozawa Y, Chen CA. 2011. Nutrient enrichment caused by marine cage culture and its influence on subtropical coral communities in turbid waters. Marine Ecology Progress Series. 423:83–93.

Hurlbert SH. 1971. The nonconcept of species diversity: a critique and alternative parameters. Ecology. 52(4):577–586.

Hyland J, Balthis L, Karakassis I, Magni P, Petrov A, Shine J, Vestergaard O, Warwick R. 2005. Organic carbon content of sediments as an indicator of stress in the marine benthos. Marine Ecology Progress Series. 295:91–103.

Israel D, Gallo C, Angel DL. 2017. Benthic artificial reefs as a means to reduce the environmental effects of cod mariculture in Skutulsfjörður, Iceland. Marine Biodiversity. 47(2):405–411.

James M, Hartstein N, Giles H. 2018. Assessment of ecological effects of expanding salmon farming in Big Glory Bay, Stewart Island – Part 1 Description of aquatic ecology. Whangamatā: Aquatic Environmental Sciences Ltd. Prepared for Sanford Ltd.

Jones E, Morrison M, Davey N, Mills S, Pallentin A, George S, Hartill B, Sutton C. 2016. Biogenic habitats on New Zealand's continental shelf. Part 1: Local ecological knowledge. Wellington: Fisheries New Zealand. New Zealand Aquatic Environment and Biodiversity Report No. 202.

Keeley N. 2013. Literature review of ecological effects of aquaculture: benthic effects. A collaboration between Ministry for Primary Industries, Cawthron Institute and National Institute for Water and Atmospheric Research Ltd. Wellington: Ministry for Primary Industries.

Keeley N, Cromey A, Goodwin E, Gibbs M, Macleod C. 2013. Predictive depositional modelling (DEPOMOD) of the interactive effect of current flow and resuspension on ecological impacts beneath salmon farms. Aquaculture Environment Interactions. 3(3):275–291.

Keeley N, Macleod C, Hopkins G, Forrest B. 2014. Spatial and temporal dynamics in macrobenthos during recovery from salmon farm induced organic enrichment: when is recovery complete? Marine Pollution Bulletin. 80:250–262.

Keeley N, Saevik P, Woodcock S, Bannister R. 2024. Tackling the elephant in the room – large-scale salmon farming and the potential for far-field ecosystem effects. Marine Pollution Bulletin. 209 (Pt A):117056.

Keeley N, Valdemarsen T, Woodcock S, Holmer M, Husa V, Bannister R. 2019. Resilience of dynamic coastal benthic ecosystems in response to large-scale finfish farming. Aquaculture Environment Interactions. 11:161–179.

Kettles H, Smith F, Shears N. 2017. Subtidal reef and rockwall communities of the greater Foveaux Strait region, Southland, New Zealand. Science for Conservation 329. Wellington: Department of Conservation.

Kuhn M. 2022. caret: Classification and Regression Training. R package version 6.0-92. [accessed 15 October 2025). https://CRAN.R-project.org/package=caret

Koçak F. 2008. Bryozoan assemblages at some marinas in the Aegean Sea. Marine Biodiversity Records. 1:e45.

Kupriyanova EK, Nishi E, Hove HA, Rzhavsky AV. 2001. Life-history patterns in serpulimorph polychaetes: ecological and evolutionary perspectives. Oceanography and Marine Biology: An Annual Review. 39:1–101.

Kutti T, Hansen PK, Ervik A, Høisæter T, Johannessen P. 2007. Effects of organic effluents from a salmon farm on a fjord system. II. Temporal and spatial patterns in infauna community composition. Aquaculture. 262(2–4):355–366.

Landis JR, Koch GG. 1977. The measurement of observer agreement for categorical data. Biometrics. 33(1):159–174.

Laroche O, Meier S, Mjøs SA, Keeley N. 2021. Effects of fish farm activities on the sponge *Weberella bursa*, and its associated microbiota. Ecological Indicators. 129:107879.

Laroche O, Meier S, Mjøs SA, Keeley N. 2022. Suspension-feeding benthic species' physiological and microbiome response to salmon farming and associated environmental changes. Frontiers in Marine Science. 9:841806.

Law BA. 2019. Quantifying transport of aquaculture particulate wastes [PhD thesis]. Halifax (NS): Dalhousie University.

Law BA, Hill PS, Maier L, Milligan TG, Paige F. 2014. Size, settling velocity and density of small suspended particles at an active salmon aquaculture site. Aquaculture Environment Interactions. 6:29–42.

Law BA, Hill PS, Milligan TG, Zions V. 2016. Erodibility of aquaculture waste from different bottom substrates. Aquaculture Environment Interactions. 8:575–584.

Lohrer AM, Thrush SF, Lundquist CJ, Vopel K, Hewitt JE, Nicholls PE. 2006. Deposition of terrigenous sediment on subtidal marine macrobenthos: response of two contrasting community types. Marine Ecology Progress Series. 307:115–125.

Lynch DR, Greenberg DA, Bilgili A, McGillicuddy Jr D J, Manning JP, Aretxabaleta AL. 2014. Particles in the coastal ocean: theory and applications. New York (NY): Cambridge University Press

MacDiarmid A, Bowden D, Cummings V, Morrison M, Jones E, Kelly M, Neil H, Nelson W, Rowden A. 2013. Sensitive marine benthic habitats defined. Wellington: National Institute of Water and Atmospheric Research. NIWA Client Report No. WLG2013-18. Prepared for Ministry for the Environment.

Macleod CK, Moltschaniwskyj NA, Crawford CM. 2006. Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. Marine Pollution Bulletin. 52:1458–1466.

Macleod CK, Moltschaniwskyj NA, Crawford CM, Forbes SE. 2007. Biological recovery from organic enrichment: some systems cope better than others. Marine Ecology Progress Series. 342:41–53.

Margalef R. 1958. Information theory in ecology. International Journal of General Systems. 3:36–71.

Martinez-Garcia E, Carlsson M, Sanchez-Jerez P, Sánchez-Lizaso JL, Sanz-Lázaro C, Holmer M. 2015. Effect of sediment grain size and bioturbation on decomposition of organic matter from aquaculture. Biogeochemistry. 125:133–148.

McLachlan A, Cockcroft AC, Malan DE. 1984. Benthic faunal response to a high energy gradient. Marine Ecology Progress Series. 16(1):51–63.

McMullin RM, McGrath EC. 2023. Candidate species selection for the development of environmental health measures for open ocean salmon aquaculture. Wellington: Ministry for Primary Industries. New Zealand Aquatic Environment and Biodiversity Report No. 317.

McMullin R, McGrath E, Bennett H. Copedo J, Ericson J, Delorme N, Miller M, Biessy L, Pearman J. 2025. Environmental health measures for open ocean aquaculture – physiological responses of epifauna to enrichment under laboratory and field studies. Wellington: Fisheries New Zealand. New Zealand Aquatic Environment and Biodiversity Report No. 353.

Michael KP. 2007. Summary of information in support of the Foveaux Strait Oyster Fishery Plan: the Foveaux Strait ecosystem and effects of oyster dredging. Final research report for the Ministry of Fisheries for Project ZBD200504. Located at: Ministry of Fisheries, Wellington.

Michael KP. 2019. The potential effects of salmon and finfish aquaculture on wild oysters (*Ostrea chilensis*) in Foveaux Strait. Prepared for Ngāi Tahu Seafood Resources. Hamilton: National Institute of Water and Atmospheric Research. NIWA Client Report No. 2019085WN.

Michael KP, Forman J, Hulston D, Sutherland J. 2015. A survey of the Foveaux Strait oyster (*Ostrea chilensis*) population (OYU5) commercial fishery areas and the status of bonamia (*Bonamia exitiosa*) in February 2015. Wellington: Ministry for Primary Industries. New Zealand Fisheries Assessment Report 2015/73.

Millar R. 2025. Annual environmental monitoring at the Beatrix Bay finfish farm: 2025. Nelson: Cawthron Institute. Cawthron Report 4155. Prepared for Ngāi Tahu Seafood Resources Limited.

Morrisey D, Anderson T, Broekhuizen N, Stenton-Dozey J, Brown S, Plew D. 2015. Baseline monitoring report for new salmon farms, Marlborough Sounds. Nelson: National Institute of Water and Atmospheric Research. NIWA Client Report No. NEL1014-020. Prepared for New Zealand King Salmon.

Muxika I, Borja A, Bald J. 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status according to the European Water Framework Directive. Marine Pollution Bulletin. 55:16–29.

[NTS] Ngāi Tahu Seafood Resources. 2025. Ngāi Tahu Seafood Biosecurity Management Plan: Hananui Aquaculture Project. Christchurch: Ngāi Tahu Seafood Resources.

Oceanum. 2025. Oceanographic modeling for the proposed Hananui Aquaculture Project in Te Ara a Kiwa. Report 1 – hydrodynamical model. Raglan: Oceanum. Prepared for SLR Consulting Ltd.

Oh ES, Edgar GJ, Kirkpatrick JB, Stuart-Smith RD, Barrett NS. 2015. Broad-scale impacts of salmon farms on temperate macroalgal assemblages on rocky reefs. Marine Pollution Bulletin. 98(1–2):201–209.

Papageorgiou N, Kalantzi I, Karakassis I. 2009. Effects of fish farming on the biological and geochemical properties of muddy and sandy sediments in the Mediterranean Sea. Marine Environmental Research. 69(5):326–336.

Pearson TH, Rosenberg R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology Annual Review. 16:229–311.

Pebesma EJ. 2004. Multivariable geostatistics in S: the gstat package. Computers & Geosciences. 30:683–691.

Pielou EC. 1966. The measurement of diversity in different types of biological collections. Journal of Theoretical Biology. 13:131–144.

Plew DR. 2019. Investigating benthic impacts at salmon farms using eddy covariance measurements of benthic oxygen fluxes. Aquaculture Environment Interactions. 11:337–357.

Roper-Lindsay J, Fuller SA, Hooson S, Sanders MD, Ussher GT. 2018. Ecological impact assessment. EIANZ guidelines for use in New Zealand: terrestrial and freshwater ecosystems. 2nd ed. Melbourne (Vic): Environment Institute of Australia and New Zealand.

Rosenberg R, Blomqvist M, Nilsson HC, Cederwall H, Dimming A. 2004. Marine quality assessment by use of benthic species-abundance distributions: a proposed new protocol within the European Union Water Framework Directive. Marine Pollution Bulletin. 49(9):728–739.

Sanderson JC, Dring MJ, Davidson K, Kelly MS. 2012. Culture, yield and bioremediation potential of *Palmaria palmata* (Linnaeus) Weber & Mohr and *Saccharina latissima* (Linnaeus) CE Lane, C. Mayes, Druehl & GW Saunders adjacent to fish farm cages in northwest Scotland. Aquaculture. 354:128–135.

Sanz-Lázaro C, Marín A. 2008. Assessment of finfish aquaculture impact on the benthic communities in the Mediterranean Sea. Dynamic Biochemistry, Process Biotechnology and Molecular Biology. 2(S1):21–32.

Smith A. 2021. Expert evidence of Abigail Smith. Presented to Environment Southland Regional Council in the matter of Resource Consent Application for Te Ara a Kiwa Aquaculture Project. 19 August 2021.

Smith AM, McGourty CR, Kregting L, Elliot A. 2005. Subtidal *Galeolaria hystrix* (Polychaeta: Serpulidae) reefs in Paterson Inlet, Stewart Island, New Zealand. New Zealand Journal of Marine and Freshwater Research. 39(6):1297–1304.

Sneddon R, Tremblay L. 2011. The New Zealand King Salmon Company Limited: assessment of environmental effects – copper and zinc. Cawthron Report No. 1984. Nelson: Cawthron Institute. Prepared for The New Zealand King Salmon Company Ltd.

Sutherland TF, Sterling AM, Ou M. 2018. Influence of salmonid aquaculture activities on a rock-cliff epifaunal community in Jervis Inlet, British Columbia. Marine Pollution Bulletin. 127:297–309.

Taylor DI, Jary M. 2018. Aquaculture site scoping: northern Stewart Island stage 1a. Nelson: Cawthron Institute. Cawthron Report No. 3163. Prepared for Ministry for Primary Industries.

Trzcinska K, Janowski L, Nowak J, Rucinska-Zjadacz M, Kruss A, von Deimling JS, Pocwiardowski P, Tegowski J. 2020. Spectral features of dual-frequency multibeam echosounder data for benthic habitat mapping. Marine Geology. 427:106239.

Vennell R, Scheel M, Weppe S, Knight B, Smeaton M. 2021. Fast Lagrangian particle tracking in unstructured ocean model grids. Ocean Dynamics. 71:423–437.

Visser AW. 1997. Using random walk models to simulate the vertical distribution of particles in a turbulent water column. Marine Ecology Progress Series. 15:175–281

Weitzman J, Steeves L, Bradford J, Filgueira R. 2019. Far-field and near-field effects of marine aquaculture. In: Sheppard C, editor. World seas: an environmental evaluation. Vol. III, Ecological issues and environmental impacts. London: Academic Press. p. 197–220.

White CA, Dworjanyn SA, Nichols PD, Mos B, Dempster T. 2016. Future aquafeeds may compromise reproductive fitness in a marine invertebrate. Marine Environmental Research. 122:67–75.

Wildish DJ, Pohle GW. 2005. Benthic macrofaunal changes resulting from finfish mariculture. In: Hargrave B, editor. Environmental effects of marine finfish aquaculture. Berlin, Heidelberg: Springer. p. 275–304.

Wilson P. 2025. Water column assessment: Hananui Aquaculture Project. Richmond: SLR Consulting. Prepared for Ngāi Tahu Seafood.

Woodcock SH, Strohmeier T, Strand Ø, Olsen SA, Bannister RJ. 2018. Mobile epibenthic fauna consume organic waste from coastal fin-fish aquaculture. Marine Environmental Research. 137:16–23.

Xu W, Cheng H, Zheng S, Hu H. 2021. Predicted mapping of seabed sediments based on MBES backscatter and bathymetric data: a case study in Joseph Bonaparte Gulf, Australia, using random forest decision tree. Journal of Marine Science and Engineering. 9:947.

You Z. 1995. Technical note: Increase of current bottom shear stress due to waves. Coastal Engineering. 26. pp 291 – 295.

Zhang J, Zhu Z, Mo WY, Liu SM, Wang DR, Zhang GS. 2018. Hypoxia and nutrient dynamics affected by marine aquaculture in a monsoon-regulated tropical coastal lagoon. Environmental Monitoring and Assessment. 190(11):656.

Zhulay I, Reiss K, Reiss H. 2015. Effects of aquaculture fallowing on the recovery of macrofauna communities. Marine Pollution Bulletin. 97(1–2):381–390.

Zvoleff A. 2020. glcm: Calculate Textures from Grey-Level Co-Occurrence Matrices (GLCMs). R package version 1.6.5. (accessed 15 October 2025). https://CRAN.R-project.org/package=glcm

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