UNIVERSITY of TASMANIA

IMAS

Institute for Marine and Antarctic Studies

Aquaculture Environment Review Series

Ecological effects of finfish aquaculture on soft sediments

Ross DJ, Elvines DM, Brasier M, Bush F & White CA

May 2025

Institute for Marine and Antarctic Studies,
University of Tasmania,
Private Bag 49, Hobart TAS 7001

Enquires should be directed to:

Dr Camille White
Institute for Marine and Antarctic Studies
University of Tasmania
Private Bag 49, Hobart, Tasmania, 7001, Australia

Camille.White@utas.edu.au

Ph: (03) 6226 8377

Front cover photo: Adam Davey

Contents

1 Introduction

This is the first in a series of independent literature reviews about marine farming interactions with the environment.

The Department of Natural Resources and Environment Tasmania (NRE Tas) commissioned the University of Tasmania's Institute for Marine and Antarctic Studies (IMAS) to perform these reviews to support the planning and regulation of marine farming in Tasmania, and to provide a reference source for people with an interest in Tasmanian aquaculture and the marine environment. It is intended to cover all forms of existing or proposed marine-based aquaculture within Tasmanian state waters and will be regularly updated and revised to consider new scientific information.
This review will focus on the interaction between soft sediment habitats and finfish aquaculture.

Soft sediment habitats cover large areas of estuarine and oceanic seabed environments (Barros 2016), both globally and around Tasmania's coastline. Soft sediments are found in estuaries, harbours, and shallow coastal zones which fringe rocky reef habitats, and extend into deeper waters. They are the dominant seafloor type in southeast Tasmanian coastal areas (Jordan et al. 2003), including in finfish aquaculture areas (Figure 1).

Soft sediment habitats are diverse and productive (Edgar et al. 2009), but generally have low structural complexity compared to other benthic habitats (Figure 2; Byers & Grabowski 2014). Most soft sediment habitats are unvegetated, but shallower soft sediment areas may contain vegetation including seagrasses and low-lying algae which contribute to primary production. Other biogenic structures (shellfish beds, sponge gardens, maerl beds) may also feature strongly in deeper soft sediment areas (Byers & Grabowski 2014, Strain et al. 2023). These 'structured' soft sediment habitats are not included in this review, due to their different biological assemblages and ecosystem functioning compared to unstructured soft sediments.

Animals inhabiting unstructured soft sediment habitats are typically small invertebrates such as polychaetes, crustaceans, echinoderms and molluscs that live within the sediment (infauna) and, to a lesser extent, on the sediment (epifauna). The feeding, burrowing and bio-irrigation activity of these animals, as well as microbial and meiofaunal activity, are important for system function and the associated ecosystem services soft sediment systems provide. These include nutrient cycling, organic matter remineralisation, oxygenation and carbon sequestration (Troell et al. 2005, Burdige 2006, Byers & Grabowski 2014; Figure 3). Arguably the most important function of these systems in the context of finfish aquaculture interactions is their ability to process organic matter.

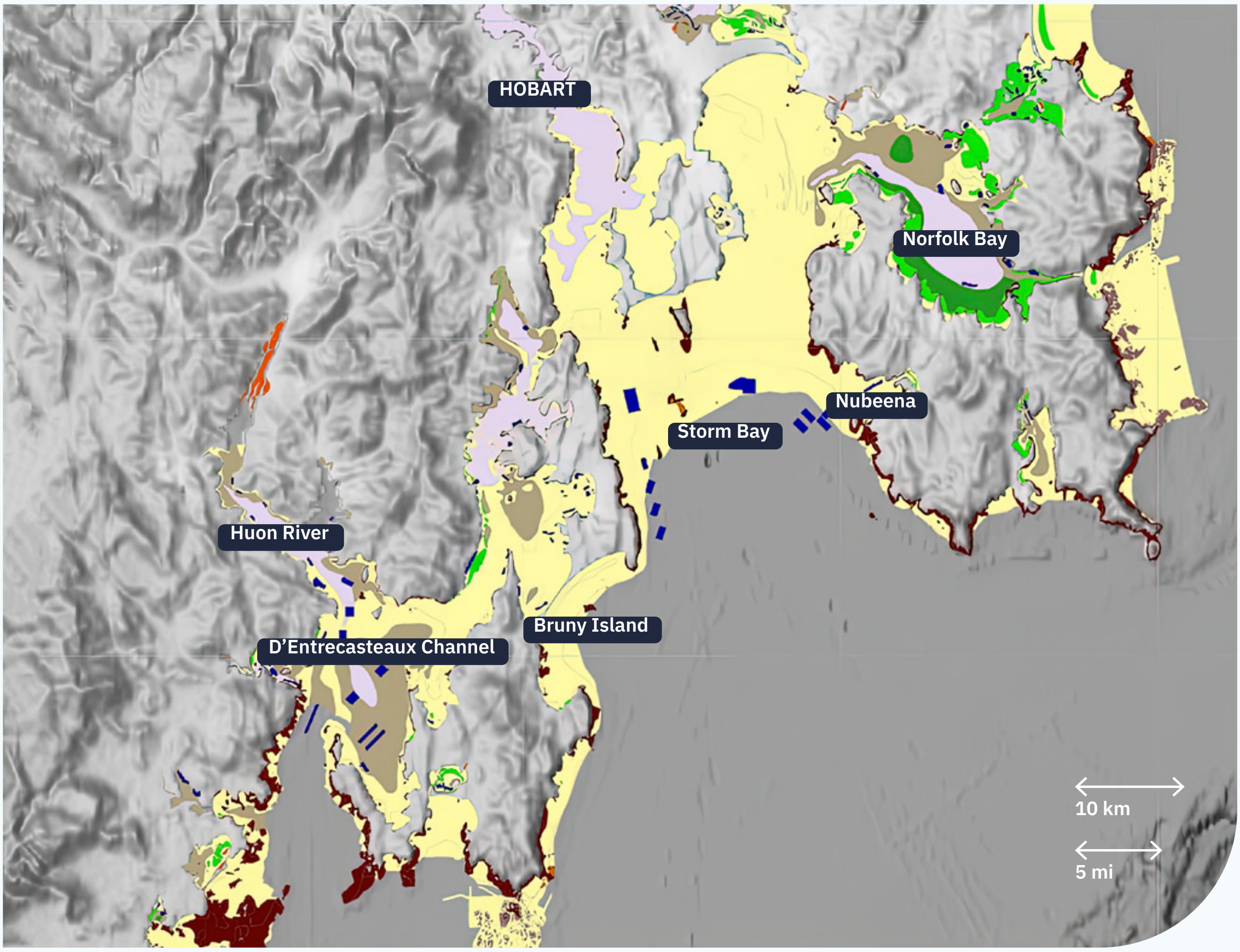
The composition of biological assemblages living in soft sediment habitats varies according to the prevailing environmental conditions, which are reflected in the local physicochemical characteristics of the sediment (Snelgrove & Butman 1995, Morrisey et al. 1998, Edgar et al. 2005, Anderson 2008, Macdonald et al. 2012). For example, silt, mud and very fine sands tend to dominate more sheltered and low-energy coastal areas and have a higher organic content, while in higher energy areas sediments have a lower organic content and feature coarser sands, possibly containing some gravel (Snelgrove & Butman 1995). Generally, suspension feeding animals are more abundant in high energy areas, and deposit feeders in lower energy, depositional areas (Snelgrove 1999, Byers & Grabowski 2014).

These patterns are evidenced in the finfish growing regions across Tasmania. In Storm Bay, coarser sediments, mostly fine sands, dominate with high species richness and diversity in benthic communities. Crustaceans are the dominant taxa followed by polychaete worms, molluscs and echinoderms (Strain et al. 2023). In more sheltered areas, finer sediments and higher organic content favours polychaete worms as the dominant taxa (Figure 1; Ross et al. 2021).

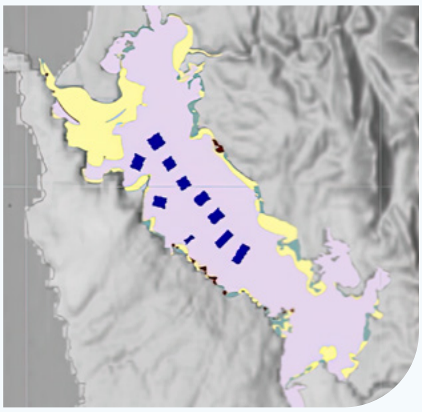
In Macquarie Harbour (west coast Tasmania) sediments are finer still, with assemblages dominated by surface-deposit-feeding polychaete worms with small numbers of crustaceans, molluscs and echinoderms (Ross et al. 2021). Benthic communities in Macquarie Harbour are naturally low in productivity due to the dark tannin-rich freshwater layer that limits light, and subsequently photosynthesis (Talman et al. 1996, Edgar et al. 1999). Such differences in benthic ecology and associated environmental factors can affect resilience of soft sediments to anthropogenic stressors including aquaculture (Bolam et al. 2002, Macleod et al. 2007, Ross et al. 2025), and lead to system-specific patterns in benthic response to organic enrichment from farming.

Figure 1: Seafloor substrate classification and marine farm leases (all types, including finfish)

South East



West coast in Macquarie Harbour

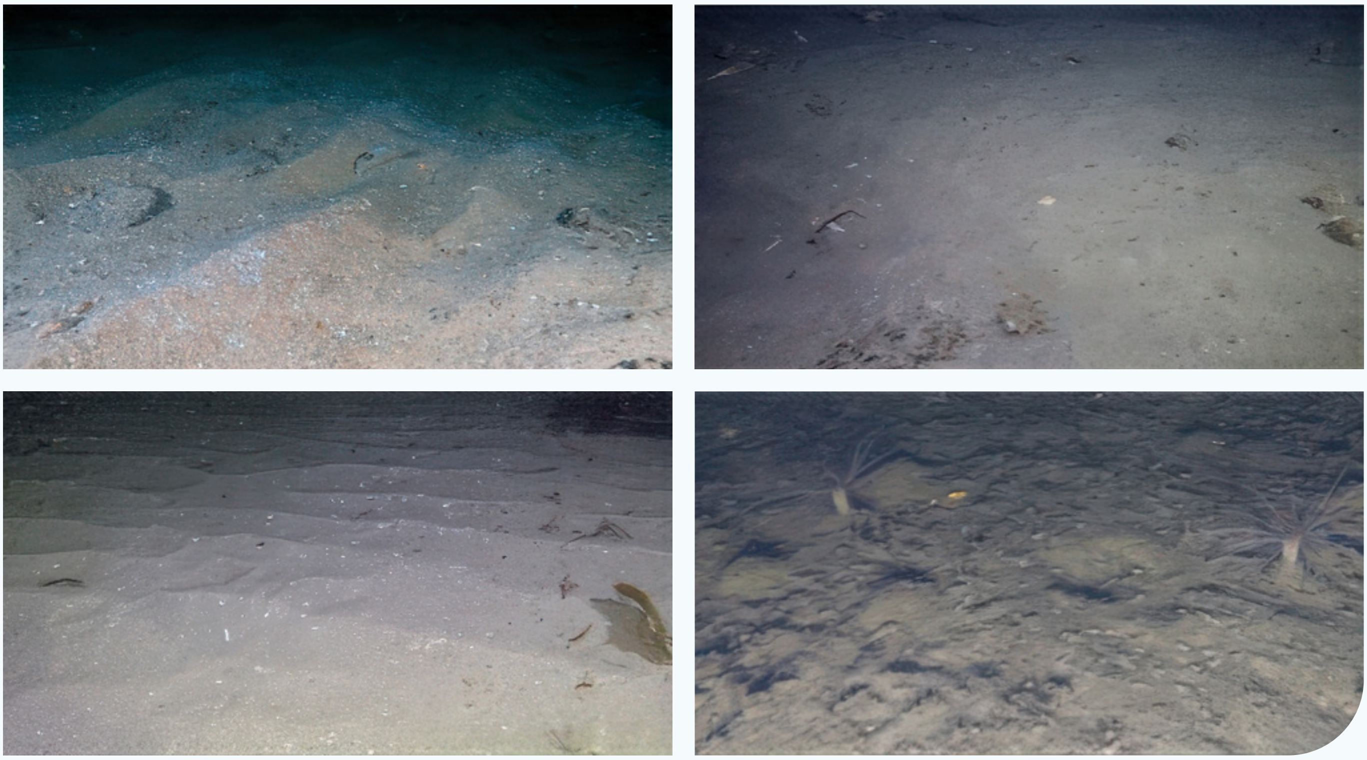


- Hard Substrata
- Consolidated Hard Substrata
- Megaclast
- Pavement
- Unconsolidated Hard Substrata
- Cobble
- Vegetation
- Macrophytes
- Macroalgae
- Seagrass
- Soft Substrata
- Coarse Sediments
- Pebble
- Sand
- Mixed Coarse Sediments
- Fine Sediments
- Silt
- Mixed Fine Sediments
- Mixed Soft Substrata
- Mixed Hard/Soft Substrata
- Marine farming leases (all types)

Classifications are from Lucieer et al. (2025; Seamap Australia National Benthic Habitat Layer[1]).

1 *In Tasmania's Marine Atlas*; https://tasmarineatlas.org/map/#01ed3a67-d17c-4815-8c2c-b6167048f6d7

Figure 2: Examples of soft sediment habitats from within Tasmania

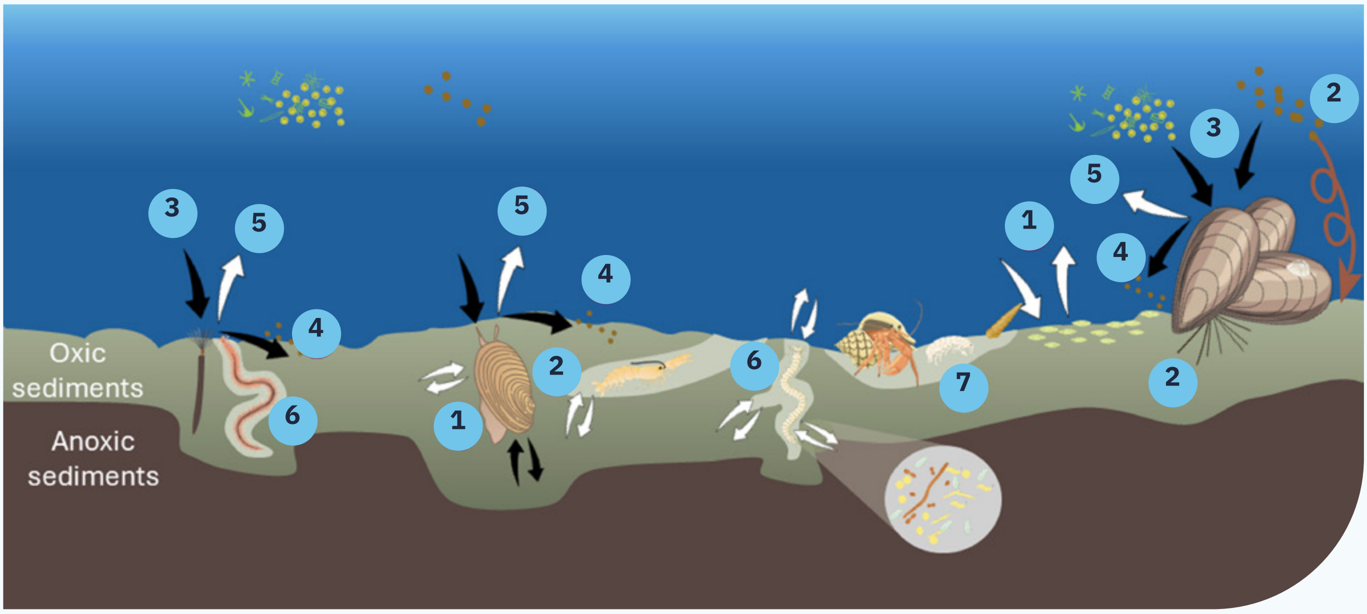


Soft sediments are typically a depositional habitat, specifically adapted to breaking down organic material. They cover vast areas of the seafloor and typically have fewer species associations at higher trophic levels compared with rocky reefs and other complex benthic habitats (Byers & Grabowski 2014). Consequently, unstructured soft sediments are the primary habitat over which finfish aquaculture developments are sited (Figure 1; Woods et al. 2004), and ecological interactions with fish farming are correspondingly well-described both in Tasmania (Ye et al. 1991, Macleod & Forbes 2004, Macleod et al. 2004a, Macleod et al. 2004b, Edgar et al. 2005) and overseas (Karakassis 2001, Tomassetti & Porrello 2005, Kalantzi & Karakassis 2006, Keeley 2013). However, there has been relatively less experience with fish farming over soft sediment habitats in more dispersive and dynamic environments where industry is currently focusing finfish aquaculture expansion (Morro et al. 2022).

The key concern regarding environmental sustainability of coastal fish farms is nutrient enrichment from the discharge of dissolved and particulate organic wastes. Importantly, the interactions of particulate wastes with benthic soft sediment habitats can be actively managed to ensure their important system functions, and their ability to recover from impacts, are maintained (Hansen et al. 2001, Read & Fernandes 2003, Bannister et al. 2014, Fletcher et al. 2022).

This review describes the present state of knowledge about potential interactions between finfish aquaculture and unstructured soft sediments, as well as their monitoring and management, with an emphasis on the Tasmanian context.

Figure 3: Adaption of Ehrnsten et al. (2020), showing the major processes controlling benthic biogeochemical fluxes



- Solute fluxes
- Particulate fluxes
- 1 Nutrient and CO2 uptake and oxygen release by primary producers
- 2 Enhanced sedimentation and sediment stabilization
- 3 Food uptake
- 4 Egestion/deposition of faeces
- 5 Nutrient excretion and respiration
- 6 Bioturbation, including bioirrigation
- 7 Mixing of sediments

2 Effects on soft sediment habitats

Finfish aquaculture can interact with soft sediments through numerous pathways. The current state of knowledge of these interactions, along with their associated effects are discussed in this section.

The major pathway of interaction between soft sediments and open-cage finfish aquaculture is the discharge of organic matter which can build up on the seabed (Kalantzi & Karakassis 2006; Figure 4). Deposition comes mainly in the form of uneaten feed and faecal material (Cromey et al. 2002, Wang et al. 2012, Schumann & Brinker 2020), but also from biofouling organisms that drop to the seafloor after being dislodged naturally, or mechanically through *in situ* cleaning (Bannister et al. 2019).

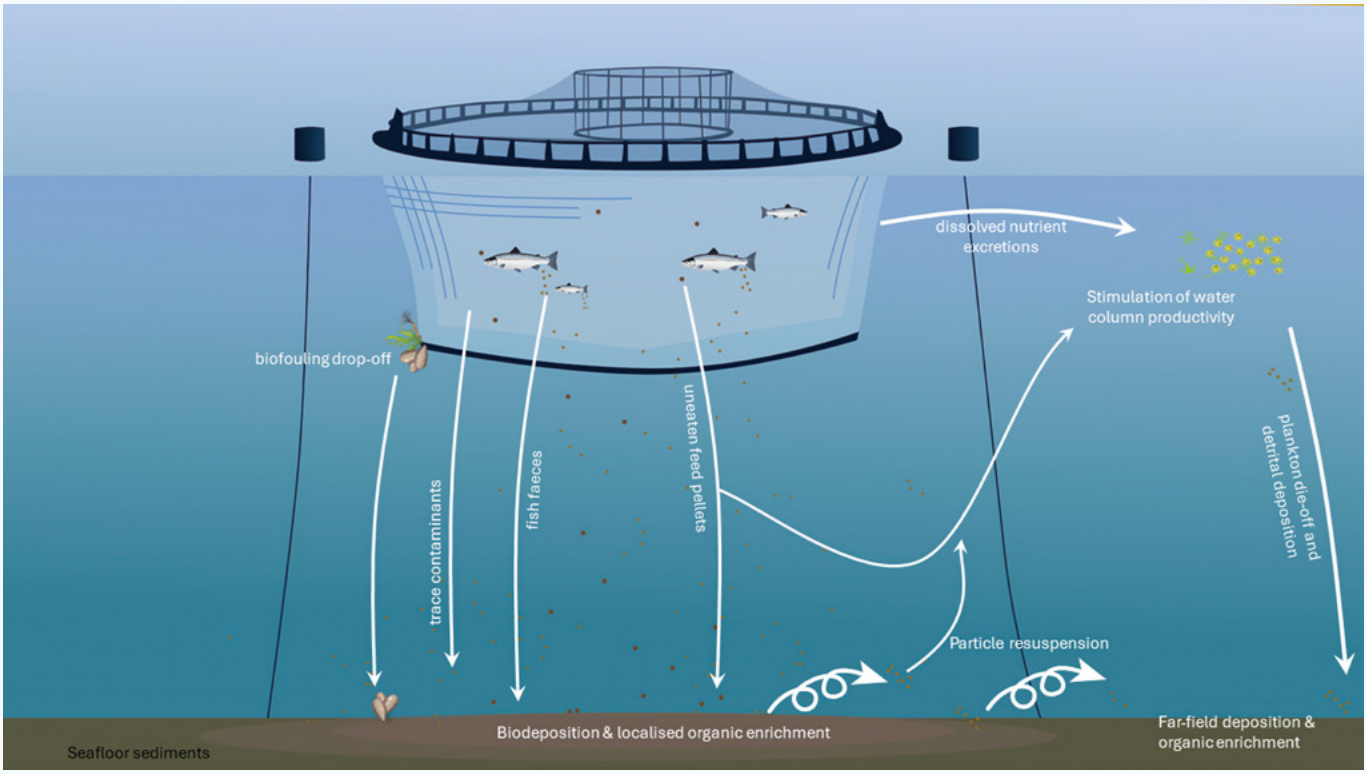
Deposition of organic material can lead to enrichment of soft sediments, ultimately leading to highly modified benthic community structure and function (Sweetman et al. 2014, Keeley et al. 2019). While effects are typically localised, there is the potential for more subtle enrichment to occur on a broader scale, depending on the receiving environment (Wang et al. 2012, Law et al. 2014, Bannister et al. 2016, Broch et al. 2017, Keeley et al. 2019). Finfish aquaculture is also a potential source of soft sediment contamination from feed additives, the use of therapeutants, and anti-foul coatings from farm infrastructure (Wu 1995, Loucks et al. 2012, Hamoutene et al. 2018), which are also considered in more detail below.

2.1 Organic enrichment

Deposits of fish faeces and uneaten feed pellets from finfish aquaculture stimulate microbial activity, which in turn alters both the chemistry and ecology of the sediments. The increase in oxygen consumption during the breakdown of excess organic matter can progressively lead to an oxygen deficit and anoxic conditions in the sediments (Figure 5a). This is followed by a transition to anaerobic decomposition processes including sulphate reduction and methanogenesis (Wu 1995, Kalantzi & Karakassis 2006, Cranford et al. 2020).

These processes result in a lowered redox (reduction-oxidation) potential and increased sediment free-sulphide concentrations (Valdemarsen et al. 2012). In highly over-enriched conditions, sediments can become completely anoxic, evident as a strong “rotten egg gas” smell of hydrogen sulphide and a black colour throughout the entire sediment profile (Figure 5a). Sediment out-gassing often also occurs in these conditions, evident as gas bubbles emerging from the sediment surface.

Figure 4: Conceptual diagram of the potential deposition mechanisms from salmon farm aquaculture

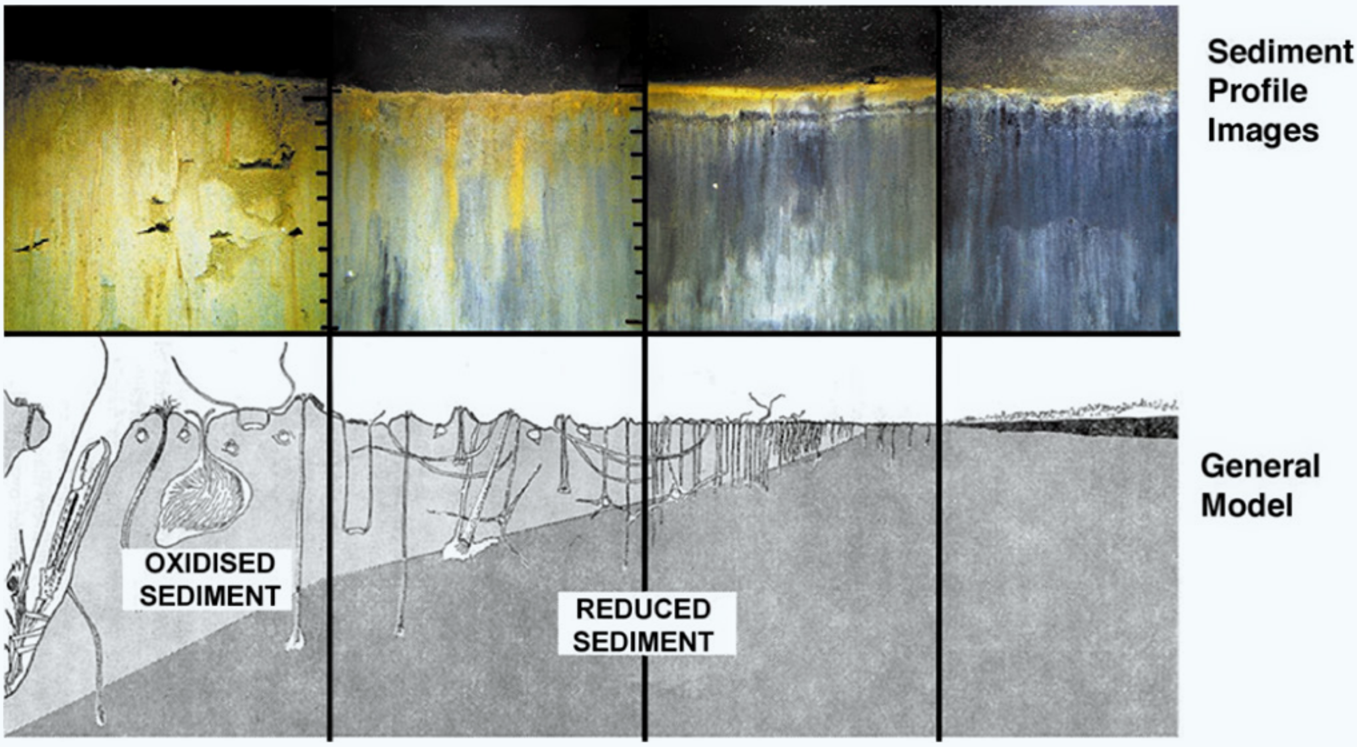


Reduced oxygen and build-up of free sulphides can adversely affect the health and functioning of soft sediment fauna, which display a range of tolerances (Hargrave et al. 2008). Faunal communities become progressively modified in response to increasing organic enrichment, with species diversity beginning to decline as sensitive species are displaced, and opportunistic taxa tolerant of low oxygen taking advantage of the readily available organic matter. Pearson and Rosenberg (1978) produced the classical model describing the series of macrobenthic successional changes in response to a gradient of increasing organic enrichment (Figure 5b), and this has been supported by many studies of the benthic impacts from aquaculture-derived organic enrichment (Holmer & Kristensen 1992, Wildish et al. 1999, Keeley 2013).

Anoxic sediments that occur in excessively over-enriched conditions are often virtually devoid of all fauna, and organic matter cycling is driven entirely by bacteria (Pearson & Rosenberg 1978, Keeley et al. 2012a). One well-known example is the mat-forming filamentous bacteria *Beggiatoa* spp., which is visible as white/cream patches on the seafloor at higher levels of enrichment (Fenchel & Bernard 1995; Figure 6). As deposition is typically greater in proximity to the farm, a gradient in benthic response is generally seen, with impact decreasing with distance from the farm (Keeley et al. 2013a, Weitzman et al. 2019, Ross et al. 2021).

Figure 5: Generalised model of benthic succession in response to organic enrichment

a) with diverse benthic communities in well oxidised sediment on a gradient through to bacteria dominated communities in fully reduced sediment, adapted from Rosenberg (2001)



b) with the classic Pearson-Rosenberg species abundance biomass response to increasing organic input (Pearson & Rosenberg 1978)

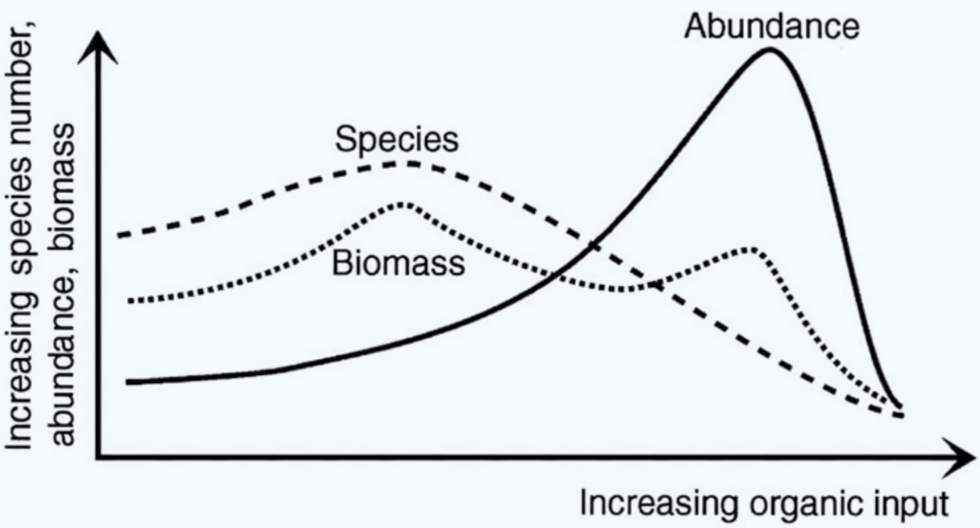
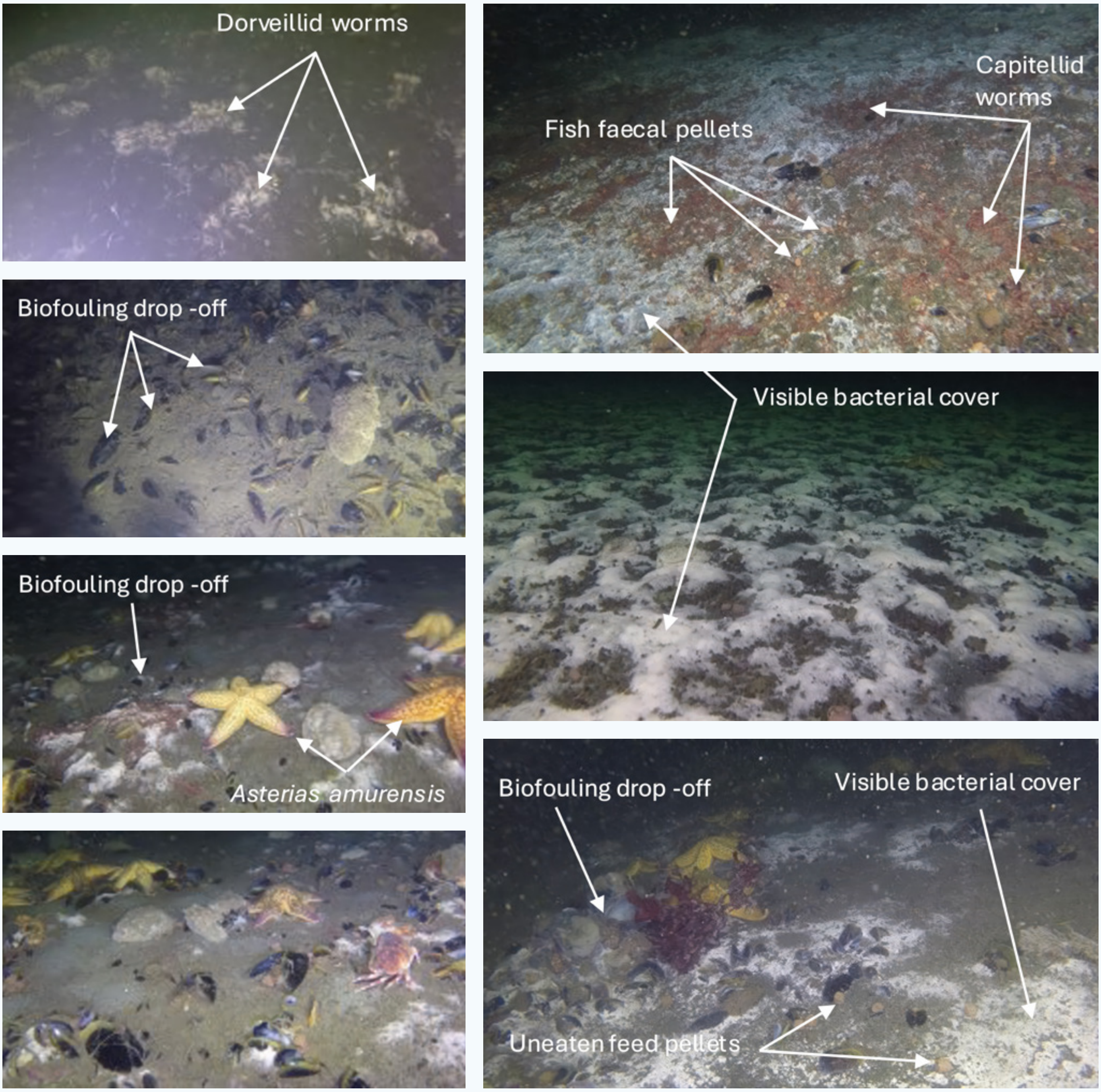


Figure 6: Generalised representation of modified soft sediment habitats due to deposition from finfish aquaculture



Common visual indicators associated with enrichment are annotated. *Image credit: IMAS*

At intermediary levels of organic enrichment, a more diverse range of species persists (Ross et al. 2021). These communities include scavengers, deposit feeders and suspension feeders which are able to rapidly exploit the enhanced detrital food supply (Ross et al. 2021), and include some of the sensitive species displaced by the low redox conditions in highly impacted sediments. Where there are only mild levels of organic enrichment, communities can respond to the more abundant food supply without associated deteriorations in sediment chemistry. Proliferation or attraction of some organisms may enhance overall species richness or biomass, including that of epifauna (White et al. 2018, McMullin et al. 2021, Ross et al. 2025), although larger and longer-lived species associated with pristine conditions may be displaced from these communities (Ross et al. 2025).

Mobile species may be attracted to deposition, including biofouling drop-off, from fish farms (Fernandez-Jover et al. 2011, Sanz-Lázaro & Marín 2011, White et al. 2018, Keeley et al. 2020). For example, Crawford et al. (2001) reported high densities of the native starfish *Coscinasterias muricata* scavenging on biofouling drop off in the vicinity of fish farms on remotely operated vehicle (ROV) surveys in the late 1990s. More recent observations have shown the side gill slug *Pleurobranchaea maculata*, New Zealand piecrust crab *Metacarcinus novaezelandiae* and northern Pacific seastar *Asterias amurensis* are also attracted to the organic deposits, the latter two being introduced species (Ross et al. 2021, Strain et al. 2023). Organisms from biofouling communities may also recruit to, or incorporate into, sediments and change the benthic community composition or habitat structure (e.g. mussel shells; Figure 6), particularly following a cleaning event (see Section 3.2.2).

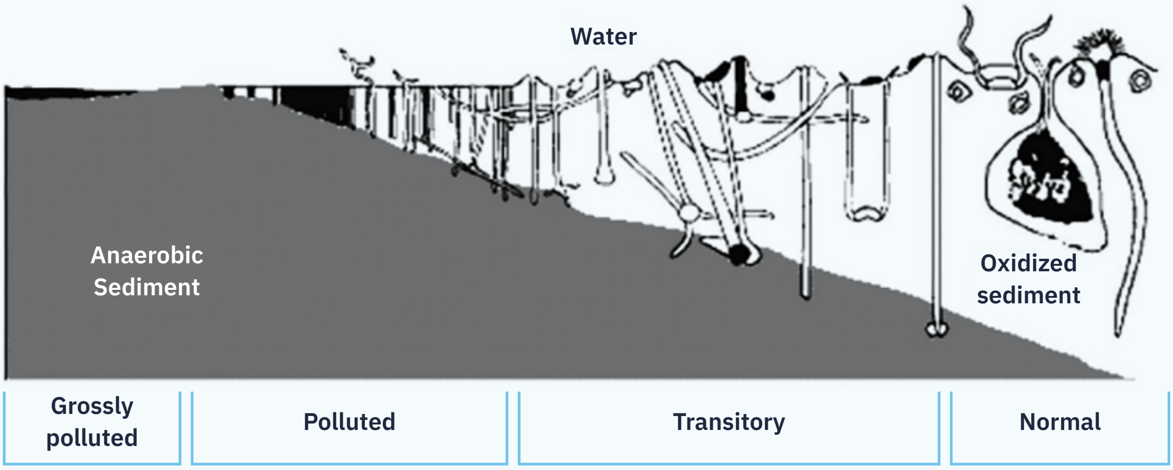
Biofouling species which feature in the sediments below fish farms include mussels (e.g. *Mytilus galloprovincialis*), ascidians (e.g. *Ascidiella aspersa*), amphipods from the genus *Jassa*, and caprellids from the genus *Caprella*. This colonisation by biofouling species on the seafloor environment also represents a potential pathway for the spread of invasive or unwanted marine species.

Aggregations of mobile deposit feeders may also enhance organic matter assimilative capacity of the benthic community and may actively transport assimilated organic carbon beyond the area of impact (Keeley et al. 2020). Localised aggregations attracted to the farm are often precluded from advanced stages of organic enrichment under cages, so aggregations generally occur in a 'halo' around the farm where sediment oxygen levels are more tolerable, yet food supply is enhanced.

Consistency with the classical benthic succession model has been demonstrated in Tasmanian systems (Ye et al. 1991, Macleod & Forbes 2004, Edgar et al. 2005, Macleod et al. 2007), with the most enriched and impacted communities typically characterised by the enrichment-tolerant opportunist, *Capitella capitata*, a small deposit feeding polychaete (Pearson & Rosenberg 1978). However, specific indicators of the various response stages can vary locally (Figure 7). In Macquarie Harbour, highly enriched farm sediments are dominated by opportunistic dorvilleid polychaetes *Schistomeringos loveni* and *Ophryotrocha shieldsi* (Figure 6) rather than capitellids. In southern Tasmanian farming regions, *Capitella* spp. is the dominant opportunist at high enrichment levels (Figure 6), as with the classical model, but additional species are also noted as important indicators, and these vary according to hydrodynamic regime.

At high energy sites in south Tasmania, *Mediomastus* sp. also prevails at high enrichment, while *Malacoceros tripartitus* and *Nebalia* sp. prevail under highly enriched conditions at lower energy sites (Ross et al. 2025). The sediment chemistry response at the different severities of organic enrichment can also vary across sites, as can its relationship with overall macrofaunal health (Keeley et al. 2013a, Ross et al. 2021, Sanz-Lazaro et al. 2021). These regional or site-level differences are important to note in the context of managing the interaction between finfish aquaculture and soft sediments.

Figure 7: Macleod et al. (2007) adaption of Pearson & Rosenberg (1978) succession model showing the local indicator taxa found in response to finfish aquaculture-derived enrichment at study leases in Tasmania



Dominant genera in Pearson & Rosenberg model (1978)

No Fauna	*Capitella* *Scolelepis*	*Labidoplax* *Corbula* *Goniada* *Thyasira* *Pholoe*	*Chaetozone* *Anatides* *Pectinaria* *Myriochele* *Ophiodromus*	*Nucula* *Amphiura* *Terebellides* *Rhodine* *Echinocardium* *Nephrops*

Dominant taxa in south-east Tasmania

Creeses Mistake

N/A	*Capitella capitata*	*Spionidae sp. 2* *Neanthes cricognatha*	*Apseudes sp. 2* *Ampelisca sp.*

Stringers Cove

N/A	*Capitella capitata*	*Nassarius nigellus* *Theora fragilis* *Corbula gibba* *(Echinocardium cordatum)*	*Amphiura elandiformus* *Mediomastus australiensis* *(Lysilla jennacubinae)* *(Thyasira adelaideana)*

The volume and rate of depositional material onto soft sediments will also influence the assimilatory capacity of the habitat. At high-flow sites, benthic communities can maintain greater species diversity under highly enriched conditions, while deposit feeding polychaetes can reach extremely high total abundances with higher resilience to macrofaunal collapse compared to sites with low current flow (Keeley et al. 2013a).

In Tasmania, Macleod indicated that biological communities in higher flow areas appeared to have a lower assimilative capacity relative to those in more sheltered areas, as their predominant functional traits were not well-adapted to organic matter processing (Macleod et al. 2006a, Macleod et al. 2007). However, recent work at several exposed and sandier farming locations across Storm Bay and the southern D'Entrecasteaux Channel found benthic communities adapted to organic matter processing, including high abundances of *Capitella* spp. at sites in close proximity to cages (Ross et al. 2021, Strain et al. 2023).

In Macquarie Harbour, the naturally low oxygen conditions in bottom waters and benthic communities lacking in diversity and biomass ('depauperate') combine to limit the assimilative capacity of benthic sediments. This predisposes the system to becoming more readily overwhelmed by additions of farm-derived organic matter than south-east Tasmania farming areas (Ross et al. 2021). Differences in assimilative capacity have implications for both managing and monitoring fish farm organic enrichment.

2.1.1 Factors affecting scale and intensity of organic enrichment

The most significant benthic responses are typically evident directly beneath and adjacent to the cages where most particulate waste is deposited, with impacts decreasing further away from the farm. However, the intensity and spatial extent of the depositional footprint and associated benthic response is site specific. As well as operational farming practices which dictate the amount and timing of particulate loads, the benthic response can be influenced by local and regional environmental (current regimes, bottom type) and ecological (species and community specific) factors.

Farm practices

The intensity and timing of feed inputs have a major influence on soft sediment effects. As with most other countries, most finfish farms in Tasmania utilise a single year class production model, where a single cohort of fish are reared. In this situation, sediment effects change with seasonal production and feed inputs, intensifying as feed inputs (and subsequent deposition) increase with fish growth (Kutti et al. 2007a, Keeley et al. 2019). Where multiple year-class production models are used, feed inputs are more consistent through time due to regular re-stocking and harvesting of fish (Fletcher et al. 2022), so there is likely to be a more continuous deposition of farm particulates with less opportunity for the sediments to recover.

Another key aspect of farm operations that influences sediment enrichment is the type of feeding system and its efficiency. Minimising feed wastage is good practice economically, environmentally, and with regard to fish health. In Tasmania, there have been major advances in the feeding systems across all growing regions in the past decade.

Site characteristics

The physical characteristics of the site will also influence the magnitude and spatial extent of soft sediment deposition. Water currents are an important factor controlling the footprint of organic enrichment, with stronger water currents spreading the particulate matter over a larger area. This reduces the intensity of enrichment directly beneath the farm but increases the size of the depositional footprint (Yokoyama et al. 2007, Chary et al. 2021).

Strong currents near the seabed may also resuspend settled organic matter, spreading it over a larger area and reducing the intensity of enrichment close to the farm. Increased water movement also promotes oxygenation of the sediment-water interface, increasing the efficiency of organic matter mineralisation and reducing effects on the biological community (Bravo & Grant 2018, Sanz-Lazaro et al. 2021).

Deeper sites are also likely to produce larger and more diffuse depositional footprints (Kutti et al. 2007b, Woodcock et al. 2018, Keeley et al. 2019). The shape of the footprint can also be distorted according to the local hydrodynamics (Keeley et al. 2013b, Broch et al. 2017), potentially leading to patchy or irregularly shaped footprints of organic enrichment in dynamic areas (Broch et al. 2017).

Mild enrichment can be detected up to 100–1,000 m from a finfish aquaculture site, depending on the site's dispersive characteristics (Keeley et al. 2013b, 2019). In Tasmania, the greatest enrichment and most significant benthic impacts have been shown to be largely restricted to within 35 m from the cage, but with milder enrichment and more subtle effects evident out to 150 m at low flow, depositional sites (Edgar et al. 2010) and 200-500m at deep or more dispersive sites (Ross et al. 2021).

Based on research performed overseas, it is understood that significant amounts of particulate material can be transported beyond the primary area of deposition at dispersive sites (Bannister et al. 2016, Keeley et al. 2019), with empirical research showing diffuse deposition of farm-derived organic matter up to 1,100 m from the farm (Woodcock et al. 2019). Far-field deposition of particulate wastes may lead to effects in intensively farmed regions where multiple farms contribute to far field deposition, and/or if particulate wastes are entrained into depositional areas such as when bathymetry and hydrodynamics are spatially variable, including in areas with strong particle transport characteristics (Bannister et al. 2016, Broch et al. 2017, White et al. 2017, Keeley et al. 2019).

Dissolved nutrients directly excreted from farmed fish or released from farm derived organic matter can also stimulate primary production in the water column, leading to increased organic matter as an indirect source for diffuse organic enrichment at a broader scale (Robinson et al. 2005, Sarà et al. 2011, Weitzman et al. 2019). The spatial extent of these effects depends on the system dynamics but can be several kilometres (see *Review 2: Ecological effects of finfish aquaculture on the water column*). Although there is potential for organic enrichment via enhanced water column productivity, the broadscale nature of this enrichment pathway is difficult to measure and identify given it is likely to be subtle and influenced by other natural and anthropogenic sources of nutrients (Sarà et al. 2011, Husa et al. 2014, Price et al. 2015).

2.1.2 Seabed recovery

Sediment recovery is also influenced by the biological and physical characteristics of the site (Macleod et al. 2006b). Assimilation and processing of the organic matter by microbes and macrofauna are key biological processes determining recovery, with bioirrigation and bioturbation (the process of sediment and interstitial water cycling facilitated by sediment infauna) enabling sediment reoxygenation and mixing.

Physical processes are also important, with water movement reoxygenating sediments by eroding accumulated organic matter from surface sediments, causing mixing and export. Generally the more severe the impact, the longer the recovery time (Keeley et al. 2015, Hale et al. 2023, Wang & Olsen 2024), although local biological and environmental factors will often combine to influence this. For example, two farming sites in southern Tasmania with a comparable farming history but different native biological assemblages, showed different recovery responses (Macleod et al. 2007).

The more exposed site with fauna adapted to dynamic sediment conditions and lower natural organic loading was slower to recover. Residual sediment chemistry conditions and slower recruitment strategies of native fauna at this location inhibited early colonisation of impacted areas. On the other hand, at the farm located in a more sheltered depositional area, benthic communities appeared to be better equipped to recolonise recovering sediments due to their greater tolerance to organically enriched conditions, and more opportunistic reproductive strategies (Macleod et al. 2007). Consequently, sediments with naturally higher organic carbon content may contain species better adapted to rapid recolonisation and remediation of recovering sediments.

The point at which the recovery is fully concluded is subjective, leading to difficulties in comparing recovery timeframes across the literature (Macleod et al. 2004b, Crawford et al. 2008, Keeley et al. 2014). Substantial benthic recovery can occur within a few months of destocking (Macleod et al. 2004b, Keeley et al. 2019), and near complete recovery has been observed at six to seven months (Brooks et al. 2003, Keeley et al. 2019). By contrast, heavily farmed sites with evidence of excess organic matter accumulations in the sediments have shown lasting effects beyond four to seven years after farming (Brooks et al. 2004, Sim et al. 2024).

Sediment chemistry is first to improve (Chang & Page 2011, Keeley et al. 2015), with signs of recovery within a few months, and full recovery within two to three years (Macleod et al. 2004b). Faunal communities typically recover at slower timescales (Brooks et al. 2004, Macleod et al. 2004b). In Tasmania and New Zealand, macrofaunal recovery was shown to commence within 3–12 months and take several years to ‘fully recover’ (Macleod et al. 2007, Keeley et al. 2014, Keeley et al. 2019). However, residual effects in benthic community structure have still been observed ten years on, although these were not considered to be functionally significant (Macleod et al. 2014).

2.2 Sediment contamination

There are additives in feed, primarily metals or therapeutants, that have the potential to accumulate in the sediments around the farm. Species that are sedentary are at greatest risk of higher exposure due to their limited mobility (Clement et al. 2010) and juvenile life stages are likely to be the most sensitive (Macleod & Eriksen 2009).

At toxic concentrations, local species diversity may be reduced, while for metals, lower concentrations may be a concern due to the potential for bioaccumulation by fauna (Macleod & Eriksen 2009). The ecological risk associated with residues of contaminants in sediments depends on their bioavailability (potential to cause biological effect; Macleod & Eriksen 2009).

2.2.1 Metals

Elevated levels of metals have been found in sediments beneath fish farms, including copper, zinc, cadmium and iron (Rožič et al. 2012, Weitzman et al. 2019, Kalantzi et al. 2021). Of these, copper and zinc are considered to present the greatest risk to soft sediment communities (Brooks & Mahnken 2003, Dean et al. 2007). Their bioavailability is controlled by sediment geochemistry.

Although bioavailability is generally lower under anoxic conditions where the highest concentrations are normally seen, changing geochemistry in recovering sediments may increase metal bioavailability (Macleod & Eriksen 2009). Measured concentrations of metal contaminants in sediments around fish farms are often found to be below guideline levels where chronic biological effects may become apparent (Rožič et al. 2012, Bennett & Elvines 2018, Aslam et al. 2020, Fletcher et al. 2024).

While feed contains copper, most sediment inputs have historically been from copper-based anti-foul coatings used on farming structures (Macleod & Eriksen 2009, Burridge et al. 2010). In Tasmania, elevated copper concentrations have been observed in farm sediments, with the highest concentrations typically reported at sites with higher organic enrichment (Macleod & Eriksen 2009). In some cases, concentrations have exceeded recommended guideline levels (Macleod & Eriksen 2009). However, in 2010 the use of copper-based antifoul coatings on farm nets was voluntarily phased out in Tasmania, with a transition to *in situ* net cleaners for biofouling management (DPIPWE 2016, Bannister et al. 2019). The use of copper nets remains common practice in many other countries where finfish aquaculture is an industry.

The source of zinc contamination in sediments is primarily from uneaten feed and faecal material, so concentrations vary with these inputs (Brooks & Mahnken 2003, Dean et al. 2007, Clement et al. 2010). In New Zealand, zinc accumulations in sediments are considered a concern because bioavailable[2] concentrations regularly exceed protective guideline values beneath a number of farms at depositional sites (Elvines et al. 2016, Fletcher et al. 2018, SLR 2023). There is also occasional and irregular exceedance of these guidelines at dispersive sites (McGrath et al. 2019, McGrath et al. 2020). No recent data is available on zinc sediment levels beneath active Tasmanian fish farms, but limited historical measurements shows zinc concentrations can be elevated at active leases, with some measurements exceeding protective guidelines (Macleod & Eriksen 2009).

2 Dilute-acid extractable zinc, used a surrogate for bioavailability.

2.2.2 Therapeutants

Therapeutants include antibiotics and parasiticides, among other chemicals used to promote fish health in the farm environment. These are normally administered via medicated feed, although bathing and injection can also be used (Leal et al. 2019). These therapeutants can ultimately be released into the receiving environment, including contaminating sediments and biota (Hamoutene et al. 2018, Leal et al. 2019, Kalantzi et al. 2021, Aquenal 2023). Antibiotics are the most commonly used therapeutant in Tasmanian aquaculture, although other types of therapeutants can be prescribed under veterinary advice (AVM 2020, DNRET 2023a).

Globally, parasiticides are commonly used to treat lice infestations (Hamoutene et al. 2018). However, sea lice parasitism is not a concern for the salmon industry in Tasmania (Nowak et al. 2011), so this parasiticide has not been used. Amoebic gill disease (AGD) is the most prevalent salmonid parasite in Tasmania, but treatment for this issue is freshwater bathing rather than therapeutant remedies (Nowak et al. 2011, Wynne et al. 2020).

The primary concerns regarding antibiotics and parasiticides are environmental persistence, effects on non-target organisms and, for antibiotics, the development of resistance through selective pressure on microbiota communities (Macleod & Eriksen 2009, Leal et al. 2019, Da Silva et al. 2022). For example, oxytetracycline (OTC) is a commonly used antibiotic in aquaculture and is reported to inhibit growth of nitrifying bacteria (Leal et al. 2019).

In Macquarie Harbour, antibiotic resistance has been inferred in bacterioplankton communities in bottom waters beneath fish farm lease areas (Da Silva et al. 2022). In general, the highest concentrations of antibiotics in sediments are found in surface sediments underneath the cages (Alday et al. 2006, Aquenal 2023), but residues have been detected 30–100 m from the cages depending on the site (Capone et al. 1996, Petuna 2022).

The environmental fate and persistence of antibiotics on the seabed depends on the amount and type administered, current flow and oxygen levels. As a guide, Alday et al. (2006) suggests the half-life[3] should be around one month for well-managed farms, but traces have been found in flocculent material at least two years later at a site in Canada (Hamoutene et al. 2018). With persistence, there is an increased risk of antimicrobial resistance (Miranda et al. 2018), although toxicity also reduces over time (Leal et al. 2019). This is more likely to occur if antibiotics are over administered (Deng et al. 2025). Some antibiotics are acutely toxic to aquatic invertebrates (Wollenberger et al. 2000) and have been found in benthic species at higher trophic levels (Capone et al. 1996).

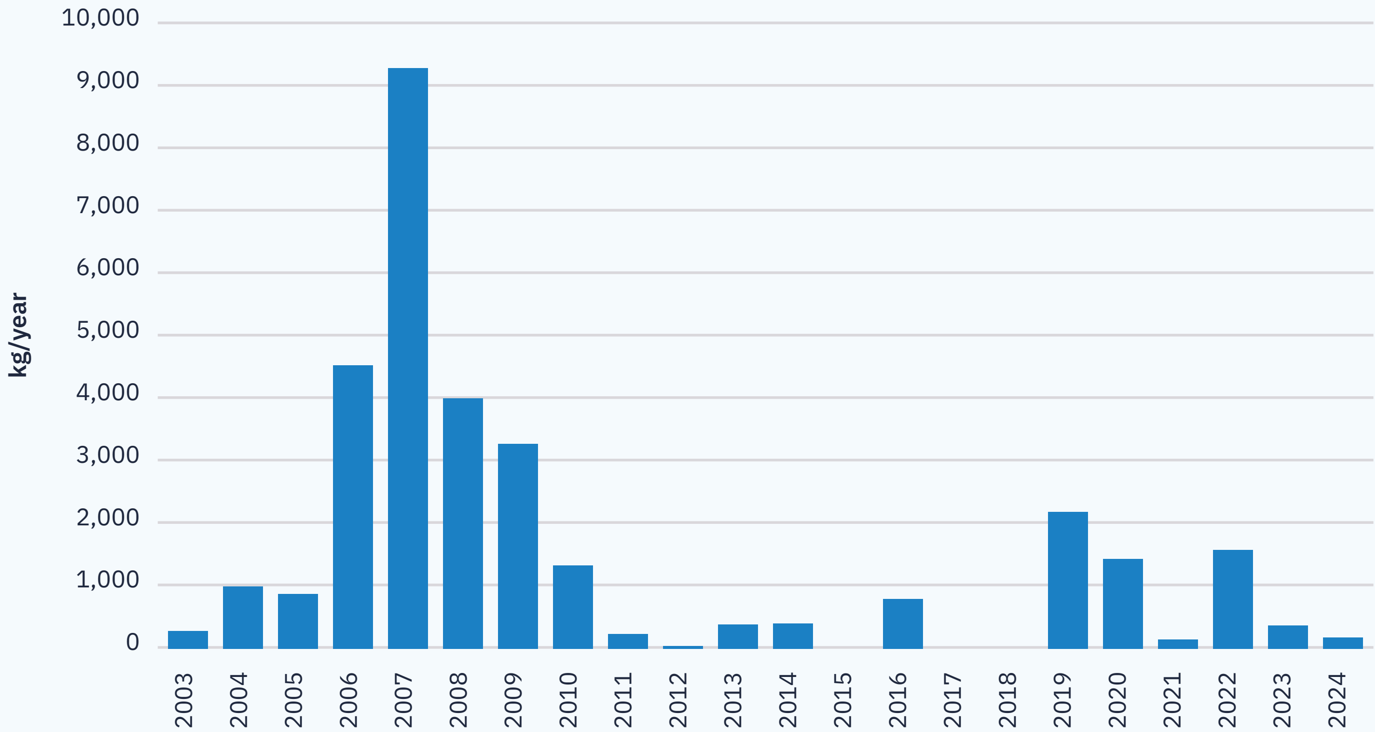
Linking sediment therapeutant levels to benthic effects can be challenging, and although adverse benthic effects have been inferred in several studies, it is not clear whether these were direct, or indirect via biogeochemical or microbiological changes (Capone et al. 1996, Gray & Herwig 1996).

3 Time required for the concentration or amount of drug to be reduced by one-half.

Sediment testing for antibiotic residues associated with applications in Tasmania has shown variable results. Trace levels of the OTC antibiotic were found in surficial sediments beneath cages up to 55 and 112 days post-treatment, respectively (Petuna 2022, Aquenal 2023), while at another farm, sediments showed no detectable residues following Trimethoprim use (Aquenal 2022). Both antibiotics were detected in the flesh of wild fish tested in proximity to the lease following application, but were absent in all but one fish captured two to three months later.

While many countries are reducing the use of antibiotics (Macleod & Eriksen 2009, Champeau 2013, Kalantzi et al. 2021), antibiotic usage and residues in sediments are common in other countries (Lake Volta, Ghana; Magna et al. [2024], Bohai Sea, China; Han et al. [2021]). The use of vaccines in Tasmania has reduced the need for antibiotics, with their administration declining significantly since 2006 – 2009 (Figure 8). However, they remain an important tool for managing fish health and welfare when dealing with certain bacterial infections.

Figure 8: Administered amount of antibiotics (active medication in kg/year) between 2003 – 2024 in Tasmania



Source: Management of therapeutants (including antibiotics) | EPA Tasmania.

Summary of potential interactions between finfish farming and soft sediments and key management practices to mitigate these interactions

Potential ecological interactions

The ecological interactions between finfish aquaculture and soft sediment ecosystems arise primarily from deposition of uneaten feed, faecal material and biofouling. The main concern is organic enrichment of sediment systems, which can cause significant modification to physicochemical sediment conditions and benthic community structure and function. Sediment contamination can also occur, mainly from additives in the feed used.

Organic enrichment

Decomposition of farm deposits on the seafloor can reduce sediment oxygen levels and increase free sulphide levels, which can be toxic to organisms.

- Moderate organic enrichment typically includes reduced total abundances and loss of species richness, due to displacement of species sensitive to smothering and sediment chemistry changes. Community structure and function begin to favour deposit feeding opportunists, and mobile epifaunal scavengers – and deposit feeders may also be attracted to the enhanced detrital food supply.
- Under heavily enriched organic conditions, communities have low species richness and are dominated by one or several species of opportunistic deposit feeding worms. Although highly modified, these are functional communities which can process organic matter deposits. The dominant species can vary according to site type, as does their capacity for organic matter waste processing.
- Excessive organic enrichment occurs when physical and biological assimilative mechanisms are overwhelmed, leading to accumulation of organic material on the sediments and eventual loss of biological communities beneath the cages. These situations are uncommon and can be avoided with proper environmental management. Some sites are more resilient to the effects of organic enrichment than others.

Other effects

- Deposited sediment can also contain metals (primarily copper and zinc) and therapeutants (such as antibiotics) from feed and on-farm applications.
- Local accumulation may occur in sediments (and food webs), and if allowed to accumulate to high enough levels, may lead to ecological effects. Accumulations have potential to persist for years.

Summary of potential interactions between finfish farming and soft sediments and key management practices to mitigate these interactions (continued)

Spatial and temporal scales

- The response of sediments typically follows a spatial gradient of effect corresponding to the level of feed input to the receiving environment and hydrodynamics, with effects most pronounced beneath the farm and becoming less severe with increasing distance. In Tasmania, more subtle effects of enrichment have been detected out to 150m at low flow deposition sites and at 200–500m at high flow and deeper sites. There is also the potential for organic enrichment at the broader scale via far field transport and deposition of particulate waste, and through increased organic matter deposition via the stimulation of primary production from dissolved nutrients inputs that are dispersed more widely.
- When deposition ceases following destocking, substantial recovery can occur within a few months. Sediment chemistry is the first to improve, and biological recovery commences within 3–12 months. However, it can take years for complete functional recovery of the community. Rates of recovery depend on the physical and biological characteristics of the site, and the level of impact.

Management and mitigation

Management actions to mitigate the effects of organic enrichment include:

- appropriate site selection during the planning process
- system and/or site-specific limits on farming via limits on feed inputs or biomass
- baseline environmental monitoring before farming commences
- local (lease) and broad scale environmental monitoring to inform farm management (e.g. fallowing requirements).

Knowledge gaps

While the response of soft sediments to organic enrichment are well understood there are some uncertainties and areas for refinement that will further improve management. Notably, an improved understanding of the fate of particulate wastes at more dispersive and exposed farming locations, and more validation of benthic health indices and the application of eDNA techniques in Tasmania farming environments.

3 Mitigation and management of impacts

Mitigation and management of finfish aquaculture and soft sediment interactions is largely focused around the management of organic enrichment onto the seafloor.

As interactions between soft sediments and coastal finfish aquaculture are generally well understood, subsequent management and mitigation techniques are well-established. This section is primarily Tasmanian focused, with international regulatory frameworks provided as a point of comparison (EPA 2022a).

Finfish farming in Tasmania primarily occurs in coastal areas encompassed by a marine farming development plan and within a marine farming lease area (DPIWE 2002, DPIPWE 2018). A marine farming licence is required to operate within a lease area, and an environmental licence is also required to farm finfish. Environmental licences require ongoing environmental monitoring and compliance against environmental conditions.

Key mitigation and management aspects include site selection, limits on feed inputs and performance-based management which includes limits of acceptable change to the environment (DNRET 2023a, see 4.2) and environmental monitoring which feeds back into management actions as necessary (Wu 1995, EPA 2022a, Fletcher et al. 2022).

3.1 Planning frameworks, site selection and production constraints

Site selection and production constraints are set for the farming area using site-specific environmental information (e.g. baseline survey data; Thompson et al. 2005 Crawford et al. 2006) and, where available, predictive modelling tools (e.g. hydrodynamic, biogeochemical, particle tracking and emulator models; Wild-Allen et al. 2005, Wild-Allen 2008, Bannister et al. 2016, Condie et al. 2017, Morro et al. 2022, MPI 2023).

Specific considerations for siting and production decision making include:

1. Site characteristics that determine the dispersion of dissolved and particulate wastes e.g. depth and hydrodynamics.
2. Proximity and connectivity to protected or sensitive/valuable habitats.
3. Characteristics of sediment biological communities e.g. known capacity to adapt to, and recover from, organic enrichment, presence of threatened species.
4. Presence of other organic enrichment sources (anthropogenic and natural) and the likelihood of cumulative or synergistic effects.

During the planning stages of a new farm development, careful consideration of farm biomass and feed inputs is made to establish the carrying capacity of the receiving system for both dissolved and particulate loadings from finfish aquaculture (Wu 1995, Stigebrandt et al. 2004, Sim-Smith & Forsythe 2013, Hadley et al. 2017, Chary et al. 2021). At the lease level, finfish production must also be set so the benthic receiving environment can assimilate localised deposition (Macleod et al. 2006b, Ross et al. 2013, Bannister et al. 2014, MPI 2023), which is assessed through both local understanding of benthic responses and numerical modelling of feed loadings/organic waste outputs (Cromey et al. 2002, Broch et al. 2017, Huon Aquaculture Company 2018). Predicted deposition rates may also be linked to pre-defined numerical relationships with benthic response parameters (Chamberlain & Stucchi 2007, Keeley et al. 2013a, Elvines et al. 2021a).

With the expansion of finfish farming into more dynamic areas, predictive models adequate for assessing deposition in more sheltered locations have been found to be limited for assessing the complexities of particle transport/settlement in more dynamic environments (Cromey et al. 2002, Fox 2022). In these areas, predicting particle waste transport must take account of particle resuspension, as well as high variability in water currents, bathymetry, seabed roughness and cohesiveness, along with increased mechanical abrasion (Law et al. 2016, Broch et al. 2017). The extent to which these factors influence particle resuspension and how they are best built into particle transport models is still an active area of research (Adams et al. 2020, Carvajalino-Fernandez et al. 2020a, Carvajalino-Fernandez et al. 2020b, Elvines et al. 2021b, Ross et al. 2021, Hadley et al. 2025).

The associated benthic community response in dynamic or previously unfarmed areas may also be more difficult to predict due to site-specific ecological differences described earlier. Recently, a lease scale model within a Relocatable Ocean Coastal Model (RECOM) for predicting depositional patterns at the more exposed Storm Bay farming areas has been developed (Hadley et al. 2025). The model has performed reasonably well against field measurements and the predictions of NewDEPOMOD, an updated version of the widely used DEPOMOD software, has been developed to better account for spatially variable currents and the challenges of modelling deposition in more exposed locations.

3.2 Environmental performance frameworks

3.2.1 Environmental requirements

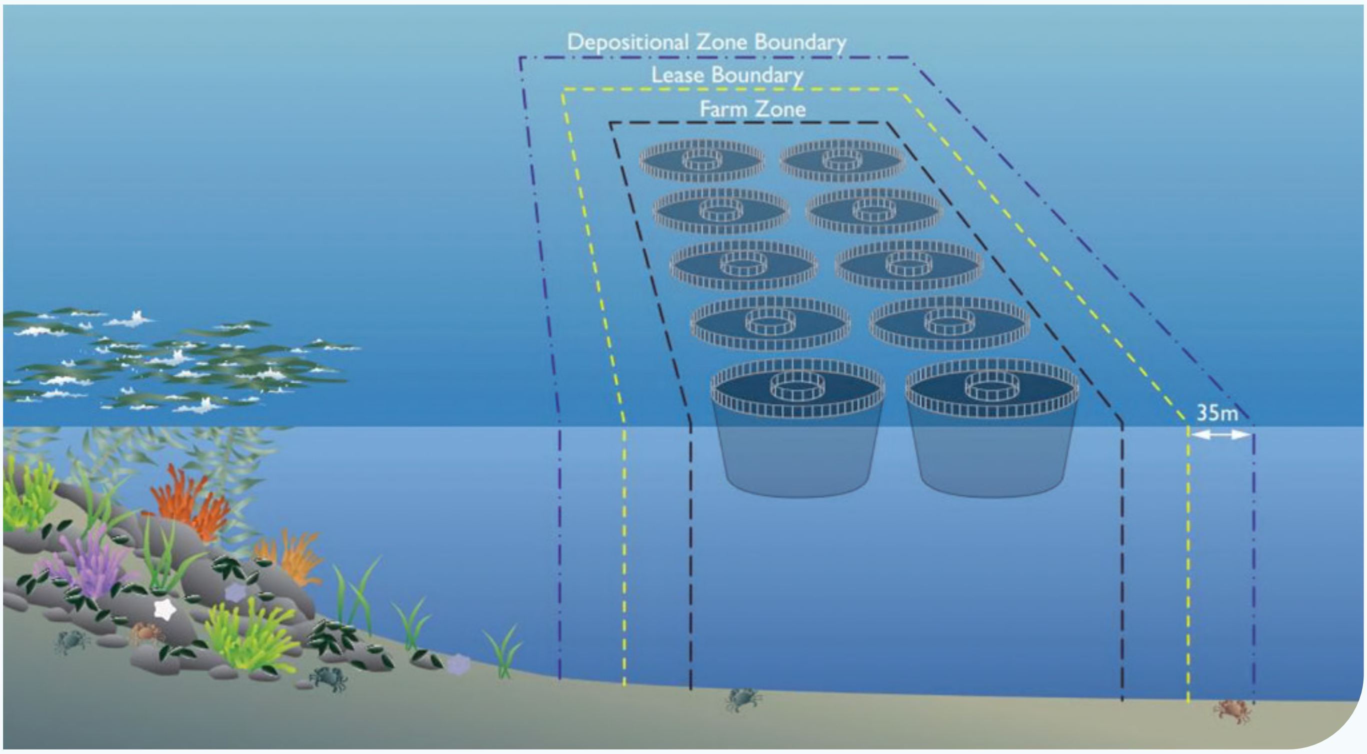
After farm installation, environmental monitoring helps to validate model predictions and routinely check that effects remain within acceptable levels. In Tasmania, a set of local (lease) scale environmental conditions are set in the environmental licence conditions (see page 25). If exceeded, operational/mitigative or additional monitoring actions need to be taken by the lease holder (EPA 2022a). Similar frameworks are used in Norway (Maroni 2000, Hansen et al. 2001, Holmer et al. 2008), Canada (Backman et al. 2009, NSDFA 2011, Chang et al. 2014) and New Zealand (Fletcher et al. 2022). Standards are usually set for the intensity of the impact, and its spatial extent.

The general intent of management is to not exceed the assimilative and processing capacity of the macrofaunal communities (Yokoyama et al. 2007, Fletcher et al. 2022) and to avoid progressive deterioration of the seabed with successive grow-out cycles (Ross et al. 2013, Fisheries and Oceans Canada 2017, EPA 2022a). Assimilative capacity of the sediments can be monitored through a range of indicators, which vary between countries.

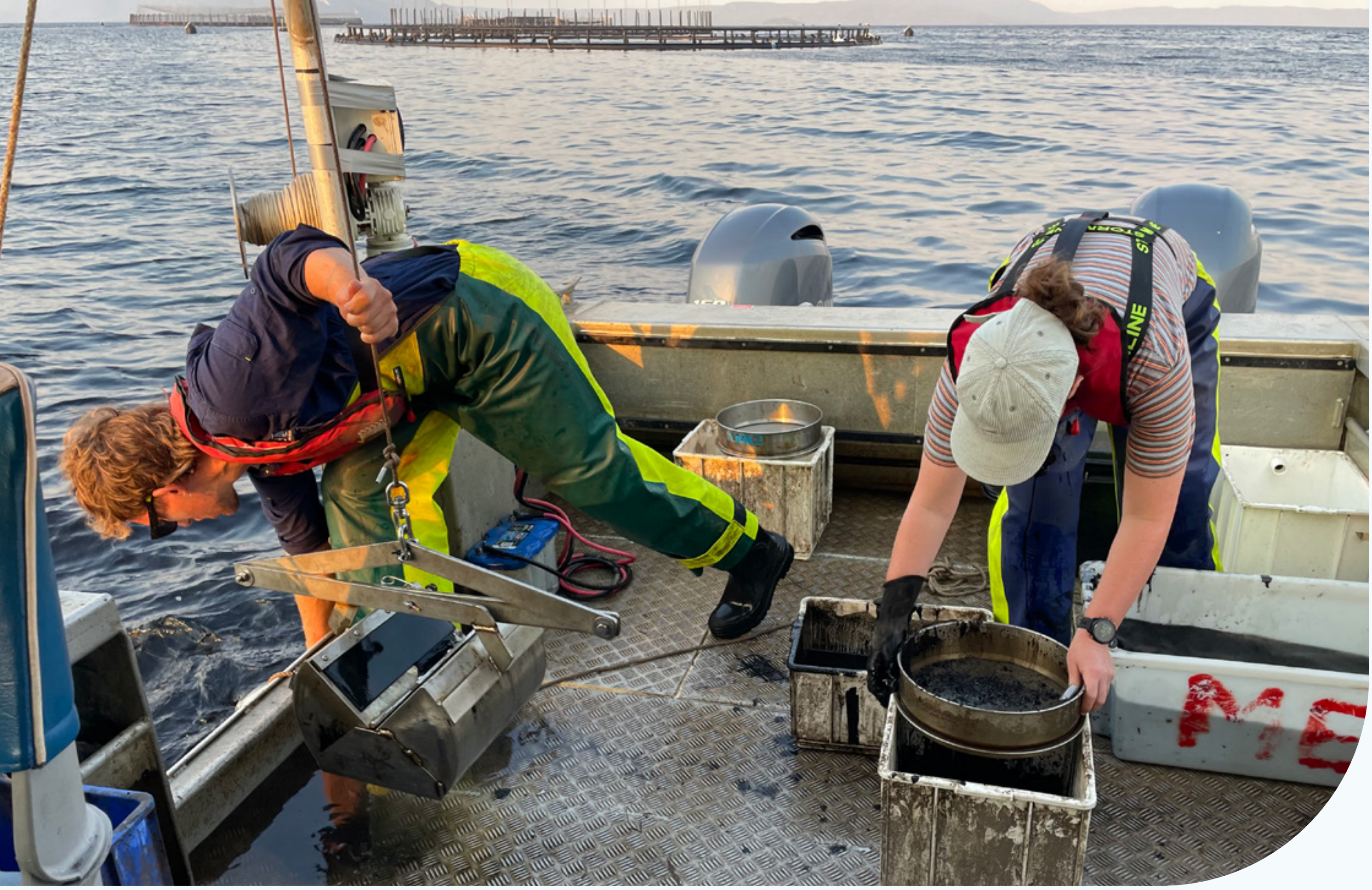
In Tasmania, environmental performance is assessed through both qualitative indicators that are monitored to inform the stocking and fallowing of cages, and a set of more protective qualitative and quantitative criteria that apply at and beyond 35 m from the lease boundary. The assessment framework of the *Environmental Standards for Tasmanian Marine Finfish Farming 2023* (Environmental Standards) is underpinned by the concept of two management zones – the Farm Zone[4] and the Depositional Zone, as illustrated in Figure 9 (DNRET 2023b). It is important to note the existing environmental conditions, indicators and thresholds will be updated through the development of Technical Standards.

It is becoming increasingly well established that the response of indicators can vary across different environments, so the choice of indicators and the thresholds used need to be locally relevant for different growing regions and environments (Giles et al. 2021, Ross et al. 2021, Strain et al. 2023). As described earlier, Macquarie Harbour provides a very clear example of this, where the response of benthic communities, including the key indicator species, differs from that observed in other growing areas (Ross et al. 2016). Although environmental licence conditions have been varied somewhat to help accommodate this, the Environmental Standards (DNRET 2023a), and development of its supporting Technical Standards for Marine Finfish Farming (see *Aquaculture Environment Review Series – General Introduction*) will enhance regulation and management of the marine finfish farming industry in Tasmania.

Figure 9: Cross section showing management zones of a marine lease in Tasmania, from DNRET (2023b)



4 Note, the 'farm zone' extents to the outer edge of the finfish pens rather than the lease boundary in the new Environmental Standards.



In Tasmania, there are also broader environment and ecosystem performance requirements where the lease/s are situated. Exceedance of these requirements may result in management actions being taken by the EPA at the regional level (Ross et al. 2022).

Environmental requirements for the broader ecosystem are mostly established for water quality, while assessment of soft-sediment condition at the broader (region) scale generally relies on expert opinion toward temporal changes and 'unusual' observations of macrofauna community structure and sediment chemistry[5] (Thompson et al. 2008, Aquenal 2021, Ross et al. 2022, Strain et al. 2023).

Performance against these standards are assessed via the Broadscale Environmental Monitoring Programs (BEMPs) in place across the most intensive growing areas, including the D'Entrecasteaux Channel and the Huon, Macquarie Harbour and Storm Bay.

The ANZECC 2000 sediment quality guidelines include threshold levels for copper and zinc in marine sediments, and these are used within some environmental licences to manage potential contamination by metals within the lease area (EPA 2022a). There are no broad environmental requirements established for residues of therapeutants in marine sediments as there are for metals, although acceptable levels may be specified to the lease holder (EPA 2022a). Residue thresholds for some therapeutants and metals do exist for some biota used for human consumption, such as residue levels in fish flesh, as part of food safety regulation guidelines (FSANZ 1998), and these are often used to provide context to monitoring results (Aquenal 2023).

5 As documented in DNRET (2023a), the new Environmental Standards and associated Technical Standards will provide guidance on the limits of acceptable changes for all key receiving habitats in marine farming regions.

Example of the existing environmental conditions for leases in Tasmania

The licence holder must comply with a range of environmental conditions in carrying out operations on a lease area which include:

Outside the Lease Area

There must be no significant visual, physicochemical or biological impacts at or extending 35 m or more beyond the boundary of the Lease Area. The following impacts may be regarded as significant:

Visual impacts:

- Presence of fish feed pellets
- Presence of bacterial mats (e.g. *Beggiatoa* spp.)
- Presence of gas bubbling arising from the sediment, either with or without disturbance of the sediment
- Presence of numerous opportunistic polychaetes (e.g. *Capitella* spp., *Dorvilleid* spp.) on the sediment surface
- In the event that a significant visual impact is detected at any point 35 m or more from the lease boundary, the licence holder may be required to undertake a triggered environmental survey or other remedial activity determined by the Director.

Physico-chemical

- Redox – a corrected redox value which differs significantly from the reference site(s) or is < 0 mV at a depth of 3 cm within a core sample.
- Sulphide – a corrected sulphide level which differs significantly from the reference site(s) or is > 250 µM at a depth of 3 cm within a core sample.

Biological:

- A 20-times increase in the total abundance of any individual taxonomic family relative to reference sites
- An increase at any compliance site of greater than 50 times the total Annelid abundance at reference sites
- A reduction in the number of families by 50 per cent or more relative to reference sites
- Complete absence of fauna
- As natural environmental variation renders some locations more susceptible to significant changes in parameter values, the above thresholds will be considered in addition to baseline environmental information for determining the presence/absence of a significant impact.

Within the Lease Area

There must be no significant visual impacts within the Lease Area. The following impacts may be regarded as significant:

Visual impacts:

- Excessive feed dumping
- Extensive bacterial mats (e.g. *Beggiatoa* spp.) on the sediment surface prior to restocking
- Spontaneous gas bubbling from the sediment.

Where areas are fallowed due to visual impacts, the Lease Area must not be restocked until the sediments have recovered to the satisfaction of the Director.

3.2.2 Operational practices

Operational practices commonly used to manage soft sediment effects from marine finfish farming include fallowing, biomass and dissolved nitrogen output limits, therapeutant prescriptions, feed management systems, and biofouling management.

Fallowing

Short-term, regular fallowing (where the site is vacant) of farm sites is a widely used practice to sustainably manage seabed organic enrichment in many countries (Hansen et al. 2001, Chang et al. 2014). Regular fallowing of a farm lease between production cycles can allow seabed recovery in the absence of deposition. In many countries, there is mandatory regular fallowing between grow-out cycles (Macleod et al. 2006a, Sim-Smith & Forsythe 2013, Broch et al. 2017). Fallowing is also a strategy used to manage fish health issues, such as parasites and disease (Chang et al. 2014).

In Tasmania, fallowing is typically performed if environmental conditions after harvest are not immediately acceptable for restocking a site, but it can also occur through the rotation of stock between cages, or rotation of cages around the lease area. A fallowing directive can also be given in Tasmania if environmental conditions are not met (e.g. Crawford 2003, DPIPWE 2017), and is commonly used if the conditions at or beyond 35 m from the lease do not meet the environmental requirements (EPA 2022a). Fallowing directives typically occur for individual cages (EPA 2022a), although Macquarie Harbour is an example of where extensive non-compliance led to a destocking order for an entire lease for a year.

Biomass and dissolved nitrogen output limits (feed-N input limit)

Farmed biomass and feed caps via the total permissible dissolved nitrogen output (TPDNO) are the common management measures used to limit the scale of marine farming activities across a region. They are also an important means to control the staged development of farming in new areas, while information on environmental impacts (and their management) is still being developed (Giles et al. 2021). In Norway, biomass limits continue to be used (Oppedal et al. 2011), but in New Zealand, total feed input limits (usually annual) are the primary means of regulating inputs, sometimes in conjunction with nitrogen-feed limits (CBOI 2013, MDC 2016).

Maximum permissible biomass is a fixed limit on farmed biomass in the system at any one time, set for a Marine Farming Development Plan (MFDP) area and/or a specific area within it. This has been a common way to regulate the scale of finfish farming activity in Tasmania in the past, although they are now superseded by TPDNO across the state. Reduced biomass in a MFDP area may be ordered in response to the exceedance of environmental trigger levels.

For example, the maximum allowable biomass in Macquarie Harbour has been progressively reduced since 2017, from a peak of over 20,000 tonnes in 2014/15 to a cap of 9,500 tonnes in recent years. This is to allow for environmental recovery in response to low levels of dissolved oxygen and a deterioration in sediment health and non-compliance (Ross et al. 2022). However, while the biomass limit was complied with, annual feed inputs were still able to increase, so the input control was changed from biomass to TPDNO (EPA 2022b).

TPDNO limits the amount of nitrogen that may be released via feed inputs (primarily for management of water column eutrophication) into a defined area over a specified time (i.e. a 12-month rolling period; EPA 2022b). The TPDNO is set for specific periods for a MFDP area (and/or specific area within). At present, TPDNOs apply to finfish farms in the D'Entrecasteaux Channel and Huon River regional area, Macquarie Harbour, the Tamar Estuary, Port Arthur, and most recently Storm Bay.

One limitation of using TPDNO as an input control is that it does not directly regulate the inputs on other feed components, which can vary independent of the protein and nitrogen content of feed, such as carbon, potentially an important determinant of the seabed response. As such, it is important this is considered when managing the scale of finish farming in an area using TPDNO.

Therapeutant prescriptions

Therapeutant inputs in Tasmanian aquaculture are controlled through veterinary prescription, as they are in other food production sectors (AVM 2020, DNRET 2023a). License holders must also notify the EPA of medication events (DNRET 2023a).

Feed management systems

The most effective feed delivery systems minimise overfeeding and maximise uptake by the caged fish, thereby reducing benthic organic enrichment (Ackefors & Enell 1994, Backman et al. 2009, Bureau & Hua 2010, White 2013). In recent years there have been significant technical advancements in feed delivery systems.

Biofouling management

Since the phase out of antifoulant treatment of nets, biofouling growth on nets and other farm infrastructure has been managed with regular cleaning using an in-situ pressure-wash or suction system.

Best practice guidelines for net cleaning in Tasmania were developed in 2013, outlining general procedures and considerations for net cleaning regimes throughout the year (TSGA 2013). These guidelines encourage cleaning of nets when they are only lightly fouled or even when not visibly fouled, and discourages pressure washing on nets with anti-fouling to prevent dislodging the anti-foul coating. Depending on the season and cleaning regime at the farm, net cleaning may be continuously rotated among cages or may only be undertaken as elicited by visual inspection, and may change seasonally (TSGA 2013).

4 Methods for monitoring

Soft sediments are consistently used as a reliable indicator of ecosystem health, with numerous methods developed to better understand the relationship between finfish aquaculture and these habitats.

4.1 Overview of indicators

Methods to monitor the ecological interactions between finfish aquaculture and soft sediments are well established in Tasmania and across the globe (e.g. Macleod & Forbes 2004, Holmer et al. 2008, Volkman et al. 2009, Rector et al. 2022). As sediment biogeochemistry and macrofaunal communities broadly reflect the prevailing environmental conditions, they provide a basis for monitoring ecological effects (Ross & Macleod 2013). Biodiversity surveys of the macrofauna not only provide a direct measurement of benthic community state, macrofaunal-based indicators are also the most reliable indicators of organic enrichment levels (Strain et al. 2023).

However, macrofauna can be laborious to collect and analyse, so substantial research effort has been applied to establish reliable relationships with more accessible indicators to measure overall benthic health.

These include sediment chemistry indicators (e.g. sulphide and redox) and visual indicators from seabed imagery, which can now provide a means for reliably and rapidly assessing benthic health (Crawford et al. 2001, Wildish et al. 2001, Macleod et al. 2002, Macleod & Forbes 2003, Cranford et al. 2017, Sutherland et al. 2019).

Species-specific indicators and the application of the global benthic health index AMBI, described on pages 29-31, have also been developed (Macleod & Forbes 2004, Ross et al. 2016, Ross et al. 2025). Comprehensive monitoring surveys will typically utilise a combination of visual, physicochemical and biological indicators of seabed health, and may also include other parameters such as sediment grain size that help characterise the background environment to assist with interpreting the results (Table 1; Strain et al. 2023). Seabed health indicators need to be selected and validated at the local level to ensure monitoring methods are targeted to the environmental responses across a particular site or region (Ross et al. 2021).

Application of the AZTI Marine Biotic Index (AMBI) in Tasmania

Many biological indicators and indices have been developed to characterise anthropogenic effects on soft sediment habitats, most of which are based on the foundational work of Pearson and Rosenberg (1978) who described the community response to gradients of organic pollution or disturbance (Keeley et al. 2012a). The AZTI Marine Biotic Index (AMBI) is one such benthic quality index that has been widely adopted. Originally developed by Borja et al. (2000) for European estuaries and coastal environments, it has been adapted for local conditions around the world (Muniz et al. 2005, Callier et al. 2008, Teixeira et al. 2012).

AMBI is often used to characterise the impacts of organic enrichment from aquaculture. In one such case, Keeley et al. (2012a) compared a suite of benthic indices for assessing impacts from finfish aquaculture in New Zealand. They found that AMBI and the closely related multivariate version M-AMBI were among the most versatile indices of the 15 investigated.

AMBI indicates the level of impact at a site based on the weighted proportion of species (or families) that are known to be tolerant of or sensitive to disturbed conditions, with the most tolerant given an ecological grouping (EG) of V through to I for the most sensitive (see below).

Table 1: AMBI groupings adapted from Grall & Glémarec (1997) with local context based on the descriptions of Macleod & Forbes (2004).

Group	Description
Group I	Species very sensitive to organic enrichment and present in normal conditions. They include the specialist carnivores, and some deposit feeding polychaetes. *Ampelicsa* sp., *Apseudes* sp., higher numbers of crustaceans.
Group II	Species indifferent to enrichment, always present in low densities with non-significant variations in time. These include suspension feeders, less selective carnivores and scavengers.
Group III	Species tolerant of excess organic matter enrichment. These species may occur in normal conditions but their populations are stimulated by organic enrichment. These are only some of the surface-deposit-feeding species, for example spionids, which ingest the superficial film of organic matter deposited at the surface.
Group IV	Second-order opportunistic species. These are the small species with a short life cycle, adapted to a life in reduced sediment where they can proliferate. They are the subsurface deposit feeders essentially related to the cirratulids.
Group V	First-order opportunistic species. These are the deposit feeders that proliferate in sediments reduced up to the surface. *Capitella* sp., *Malacoceros tripartitus*, *Nebalia* sp. Some nematodes and oligochaetes may also be present.

Application of the AZTI Marine Biotic Index (AMBI) in Tasmania (continued)

Since it was first established, the AMBI species database has increased substantially to include a wider range of species from around the globe. However, its application in regions outside of Europe has required the development of locally applicable species databases. Ross et al. (2016) successfully applied AMBI in the unique Macquarie Harbour on the west coast of Tasmania. However, until now, it has not been developed and tested in other regions of Tasmania where benthic macrofaunal communities are far more diverse.

Ecological groupings (EGs) can be developed for local AMBI analyses in the absence of sufficient species coverage in the original database (Muniz et al. 2005, Keeley et al. 2012a, Keeley et al. 2012b). Typically, this includes a combination of quantitative analysis and best professional judgement (BPJ) based on a prior knowledge of impact levels. Strain et al. (2023) further developed and tested AMBI based on data collected at farms in Storm Bay where they allocated species to different species groupings based on prior knowledge of impact stages in Tasmania (Macleod & Forbes 2004, Edgar et al. 2010) and a calibration exercise using a range of response parameters (physicochemical, visual and biological) measured at varying distances from the source of enrichment at the Trumpeter Bay salmon lease off north Bruny Island in Storm Bay, over successive surveys.

This approach resulted in a simple classification of EGs based on family abundances at sites along an enrichment (impact) gradient.

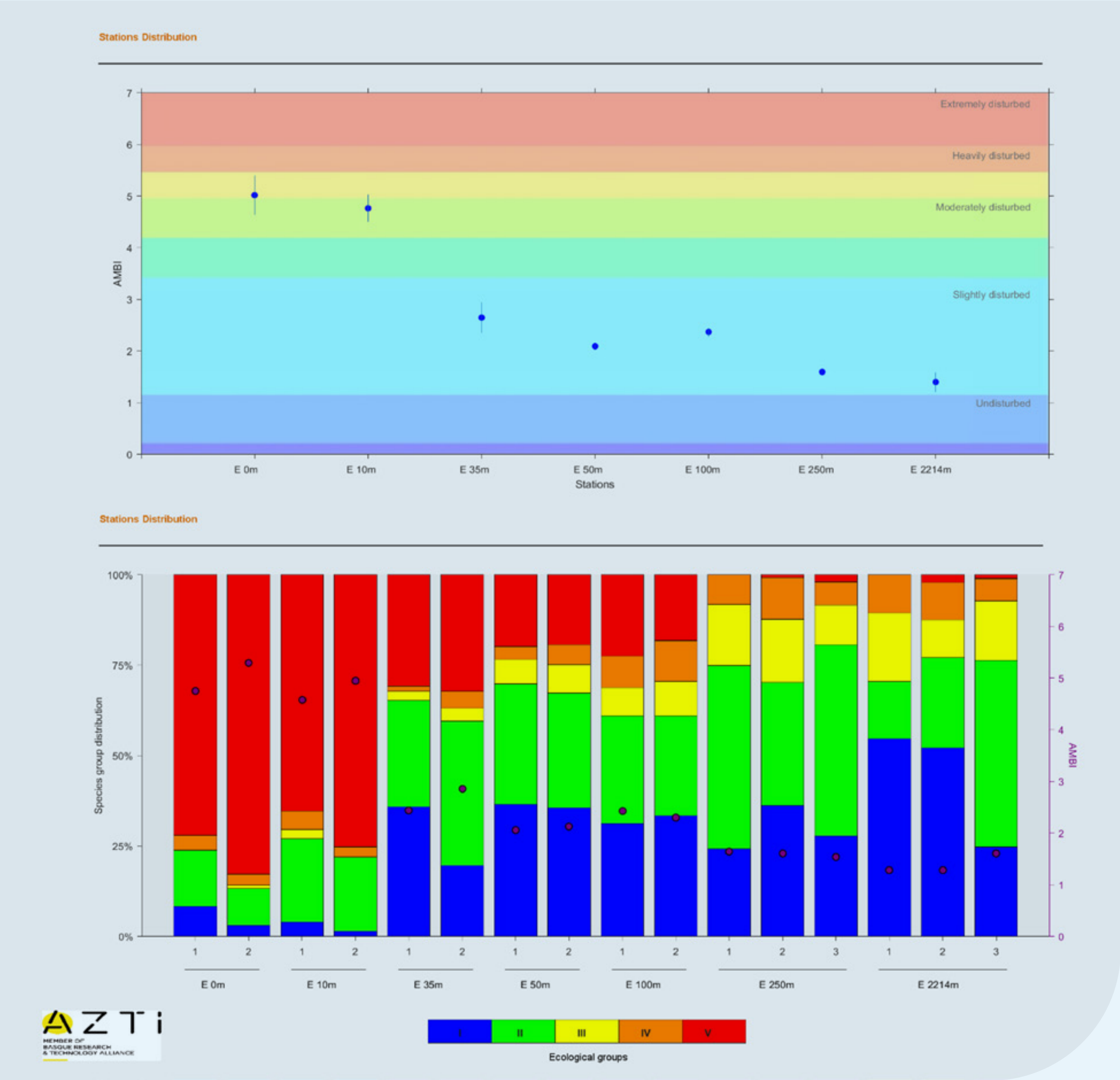
The efficacy of the new index (AMBI-TAS) was then tested using the dataset from Yellow Bluff. The AMBI-TAS classification of sites correlated well with other response parameters. In the example below, macrofaunal communities were classified as heavily disturbed at 0 m, moderately disturbed at 10 m and generally improved with increasing distance from the cage (see below). Further refinement including additional Group I families will refine the index and reduce the tendency to score control sites as slightly disturbed.

M-AMBI

Following from AMBI, a multivariate version (M-AMBI) was developed that incorporates measures of richness and diversity (Muxika et al. 2007). M-AMBI has proven useful in distinguishing and characterising extremely impacted sites where conditions are azoic. For the conditions encountered in this study we found very good agreement between both versions and for simplicity we only present AMBI results.

microgAMBI

The application of the AMBI index was also extended to environmental DNA (eDNA) metabarcoding at the Yellow Bluff lease and the results suggested the eDNA analogue of AMBI, microgAMBI, based on the bacterial community (16S rRNA) provides a sensitive measure of enrichment (Strain et al. 2023).



AMBI output for a transect from a lease in Storm Bay. The top panel shows the AMBI score at increasing distances from the cage and the bottom panel shows the relative proportion of each ecological group that contributed to the score.

A recent advancement in soft sediment monitoring is the development and application of environmental DNA (eDNA) metabarcoding to identify biological communities based on genetic material extracted from environmental samples such as sediment or water. It has also proven highly effective in measuring the response to organic enrichment from fish farming, with the potential to overcome some of the limitations of more traditional techniques (see page 33).

In New Zealand, bacterial eDNA was shown to accurately reflect organic enrichment levels and overall benthic health and is now being utilised within the regulatory framework (Fletcher et al. 2022). Recent work in Storm Bay provides early promise for its potential application to monitoring finfish sediment interactions in Tasmania (Strain et al. 2023, Coutts et al. 2024).

eDNA for examining fish farm – soft sediment interactions

Metabarcoding of DNA extracted from soft sediments has been tested globally as a powerful and cost-effective addition to the monitoring toolkit for salmon aquaculture environmental interactions (Rector et al. 2022). Several recent eDNA metabarcoding projects in the finfish farming context have developed biotic indices for environmental impact assessment (Aylagas et al. 2014, Aylagas et al. 2017, Keeley et al. 2018), cross-validating results with more traditional macrofauna and physicochemical metrics.

Measured using eDNA metabarcoding, bacterial assemblages around fish farms in New Zealand's Marlborough Sounds were shown to significantly differentiate organic enrichment gradients between reference sites, with repeatable and regionally transferable patterns (Keeley et al. 2018). Used in conjunction with the established eDNA-based bacterial index, which weights taxonomic units based on their tolerance to enrichment (Aylagas et al. 2021), results from this technique were able to assign organic enrichment impact levels equally as well as traditional sample indicators of organic enrichment (macrofauna and physicochemical indicators; Pochon et al. 2021). The eDNA-based bacterial index is now accepted and utilised as an alternative best practice to macrofauna sampling within soft sediment regulatory monitoring of the salmon farming industry in New Zealand (Fletcher et al. 2022).

eDNA metabarcoding has been tested in Tasmania in Storm Bay to measure both local scale farm-derived enrichment and broader scale environmental change in soft sediment habitats, with results indicating bacterial communities also provide a sensitive measure of enrichment (Coutts et al. 2024). The approach has been recommended for testing more broadly in Tasmania.

Finally, biochemical indicators can be useful for examining organic enrichment of sediments by finfish aquaculture. Where the biochemical composition (e.g. fatty acid profile, stable isotope composition) of fish waste is distinct from other inputs, it can be tracked in the environment and into connected food webs. This can allow the footprint of organic waste deposition to be mapped (Elvines et al. 2024), biological assimilation pathways to be identified (White et al. 2017, McMullin et al. 2021), and species-specific interactions to be examined (White et al. 2018, Woodcock et al. 2018).

Biochemical tracers may also be useful for determining contributions of finfish aquaculture to observed environmental changes more broadly within an ecosystem (Rector et al. 2022, Elvines et al. 2023).

While biochemical tracers have been applied for several decades, there have been technological advancements which have increased their accessibility and flexibility in environmental applications (White et al. 2019). Moreover, the evolution of finfish aquaculture feeds to include terrestrial fats and proteins in place of fish-meals has increased the efficacy of fatty acid and stable isotope tracers as these indicators more readily distinguish between marine and terrestrial carbon sources (White et al. 2019, Elvines et al. 2023). While biochemical tracers are not routinely used in monitoring, their application in targeted studies can facilitate our understanding of sediment interactions to better inform planning and management.

Table 2: Overview of indicators for assessing finfish aquaculture interactions with soft sediments

Indicators	Description
Macrofauna	**Generalised patterns of change in soft sediment communities are predictable and respond to specific stages of organic enrichment in soft sediments. The use of macrobenthic faunal community data is a sensitive and reliable measure of sediment condition in Tasmania and overseas. To collect sediment for macrofaunal community analysis, a Van Veen grab sampler is typically used (Figure 10d). A standardised volume of sediment is sieved to retain macrofaunal species (>0.5mm or >1.0mm), and animals are identified and counted. Important indicator species relevant to a given farming site are identified to the lowest taxonomic level possible.**
Community structure	The most comprehensive descriptor of macrofauna community involves analysis of all taxa and their abundances, and a multivariate approach which is most sensitive to detecting spatial and temporal community changes.
Species-specific indicators	The presence/absence and abundances of certain species can be used to indicate organic enrichment. However, effective enrichment indicators can vary across sites.
Community measures, Calculated/biological indices	Community metrics and calculated indices including total abundance (the number of individuals), species richness (number of different taxa) and species diversity. Shannon-index and AZTI Marine Biotic Index (AMBI) are also informative measures to indicate community health in response to disturbance including from organic enrichment (Strain et al. 2023).
Sediment physico-chemical properties	**Prevailing environmental and sediment condition can be captured using a range of indicators. A subset of these are also useful indicators to measure organic enrichment in sediments. These indicators are typically measured from surficial sediment samples (typically the top 1–3cm), usually collected via a sediment corer (Figure 10a-c).**
Sulphides, Reduction-oxidation potential (redox)	Two key methods exist for sampling sediment sulphides – the ISE method (Wildish et al. 1999) and the UV method (Cranford et al. 2017). The latter has proven to be a truer measure of free-sulphide concentrations in the sediments (Cranford et al. 2017, Cranford et al. 2020), and the results from both techniques are not directly comparable. Both are considered acceptable methods for monitoring organic enrichment. Measuring the redox potential of the sediments provides a broader measure of the oxidative state of the sediments, providing insight into the oxygen availability and microbial activity that change in response to organic enrichment.
C & N content, C & N stable isotope values, sediment texture	These parameters add to the interpretation of macrofaunal analysis (Ross et al. 2021) and help distinguish ecological changes due to natural vs anthropogenic pressures. In Tasmania, δ15N isotope values in sediments can be a useful indicator of enrichment levels (Ross et al. 2021). C and N content are historically used as indicators of organic enrichment (sediment accumulations of C and N), but are not the best indicators for this purpose in Tasmania.

Indicators	Description
Metals	Metals are part of natural sediment geochemistry and can be useful indicators adding to interpretation of temporal and spatial ecological changes. Copper and zinc are directly measured from sediments to investigate total sediment burdens and potential contamination from aquaculture. More in-depth analysis may be triggered depending on the test outcome. Other metals may also be included in monitoring, such as arsenic, cadmium, cobalt, chromium, manganese, nickel and lead (Aquenal 2019).
Therapeutants (e.g. antibiotics)	Measured as applicable in response to on-farm use. Directly measured from surficial sediments, collected from core samples up to 75mm deep (Macleod & Eriksen 2009, Aquenal 2022).
Visual indicators	**Seabed images collected via still or video capture are a means of rapidly assessing seabed condition, providing supplementary information for more comprehensive surveys.**
Level of *Beggiatoa* cover, sediment colour, presence of gas bubbles/ outgassing, feed pellets & faeces, farm debris, presence/density of indicator species, bioturbation, animal tracks/tubes/casts	Specific and predictable patterns of change in visual appearance of the sediment have been identified for Macquarie Harbour and the southeast of Tasmania in response to increasing stages of organic enrichment and recovery. Similar indicators have been developed and used globally (Hamoutene et al. 2014, Sutherland et al. 2019, Simone & Grant 2020).

A detailed overview of indicators is provided in the field guide by Ross et al. (2025), which provides an update to Macleod & Forbes (2004) based on contemporary research. Readers are referred to these documents for more detailed information.

Figure 10: Sampling equipment commonly used to collect sediment samples

a) Craib corer

b) Multicorer

c) HAPs corer

d) Van Veen grab

Adapted from Ross et al. (2025)

4.2 Monitoring design

Monitoring is typically used to capture finfish aquaculture interactions at two spatial scales – the lease (local) scale (Section 4.2.1) to monitor for the direct effects from particulate enrichment and the broadscale (Section 4.2.2) to monitor for environmental change and the potential for indirect effects at the broad (region) scale.

A sampling program that can account for background spatial and temporal variability and establish reference conditions is imperative when trying to monitor and measure the potential effects of finfish aquaculture on soft sediment habitats. Reference conditions for assessing the effects of individual leases (i.e. local scale) are captured during pre-farming baseline surveys and at reference or control site monitoring during active farming. Importantly, this will not only capture pre-farming conditions but also capture changes over time in response to external (non-aquaculture related) drivers, both natural and anthropogenic. Similarly, broadscale surveys conducted prior to active farming across the region are critical to assess the potential for indirect effects of aquaculture in the context of natural variability.

In Tasmania, baseline surveys are typically a requirement before marine farming activities commence and/or when there is a variation to a lease area (EPA 2022a). Once farming commences, the environmental licences require lease scale monitoring at compliance and control sites and broadscale monitoring[6] of sites encompassing the region.

Targeted monitoring/research may also be required in response to specific environmental concerns or information gaps. Technical Standards[7] for soft sediment monitoring are presently in development and these standards, which sit under the Environmental Standards (DNRET 2023a), will build on the existing regulatory requirements described in the following sections (DNRET 2023b).

4.2.1 Local scale

Organic enrichment

Local lease-specific monitoring usually focuses on quantifying the maximum impact at or under the pen, and the spatial extent of organic enrichment. In Tasmania, Macleod and Forbes (2004) characterised the sediment response to organic enrichment from farming into nine different stages of effect and/or recovery (Figure 11). Expected values/characteristics for each stage within farming areas in Tasmania are presently being updated (Ross et al. 2025).

While monitoring results are not formally assessed against these stages within the existing management framework, the nine-stage characterisation of impacts is an informative and structured context against which impacts can be assessed using a large range of monitoring indicators.

6 Typically a joint requirement of all marine farms operating in the region.

7 Technical Standards are being developed by the Environment Protection Authority to support the new Environmental Standard. The first Technical Standard, which is currently at consultation stage, pertains to soft sediment video survey methods (EPA 2024). Other Technical Standards are being developed for quantitative sediment surveys, other seabed habitats, baseline surveys and broadscale surveys: https://epa.tas.gov.au/business-industry/regulation/salmon-aquaculture/technical-standards-for-marine-finfish-farms.

Lease-scale monitoring is undertaken routinely in Tasmania as part of the conditions of an environmental licence. Overseas, monitoring is designed to target the time when feed input/deposition, and therefore impact, is the highest (EPA 2022a, Fletcher et al. 2022). In Tasmania, the timing of compliance monitoring in relation to intensity of impact has historically not been specified in the regulatory framework, but the environmental licences now typically require the surveys to be undertaken within 30 days of peak production.

Licence holders are required to assess for visual indicators (see page 25) using video survey methods which provide an accessible and rapid means for seabed assessment (see pages 19-20). Monitoring of compliance sites situated 35 m from the boundary of the lease area and at control site(s) against a more protective set of visual indicators is also required (Woods et al. 2004, DPIPWE 2017, Strain et al. 2020).

The default monitoring in Tasmania is a visual survey, which is conducted using ROVs or a frame-mounted downward-facing camera. A minimum of three minutes of footage is collected at each site and assessed against the environmental standards in the licence (see page 25). More detailed sediment sampling using quantitative indicators is also required (infauna, sulphides, redox, particle grain size, C & N stable isotopes) in baseline surveys of the lease area (Aquenal 2019, EPA 2022a) and for new leases it may be required during the first few years of production (see Strain et al. 2023).

More detailed sediment surveys may also be triggered if environmental conditions are not met. With the exception of some new leases, detailed sediment surveys are infrequent given other management responses are typically employed. These include fallowing the cage until sediment conditions have improved (assessed through visual surveys) so it can be restocked, or more regular or more detailed (e.g. increased spatial resolution) visual monitoring as required (EPA 2022a).

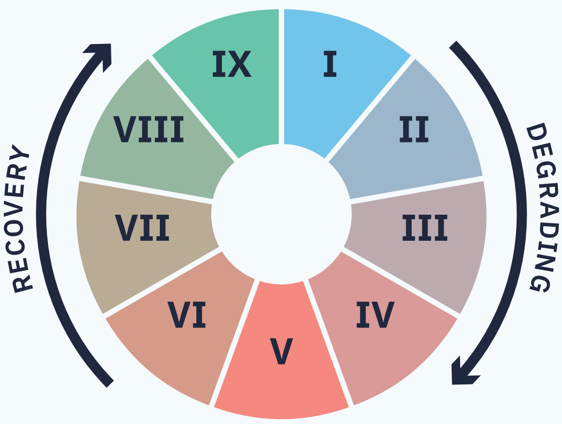
More targeted lease-scale monitoring may also be performed under collaborative research funding agreements. For example, the Storm Bay Observing System FRDC project (Strain et al. 2023) was implemented to assess the environmental performance of the industry at the lease and broad (region) scale and to make recommendations on the further refinement of the regulatory monitoring requirements at both lease and broadscales, based on the findings of the detailed sampling program enabled through the research project.

Similarly, in response to changes in farming practices, innovations in technology, and expansion of the industry into new areas, FRDC project 2015/024 – *Managing ecosystem interactions across differing environments: building flexibility and risk assurance into environmental management strategies* (Ross et al. 2021) included extensive sampling of leases in Storm Bay and the southern D'Entrecasteaux Channel in the southeast and Macquarie Harbour on the west coast of Tasmania. This was conducted to ensure that current approaches to monitoring and assessment of soft sediment habitats remain fit for purpose in different growing regions.

Metals

Sediment contamination is usually measured in the first instance by directly testing the total level of the metal within the sediment. If levels indicate the potential for ecological effects, then additional action is taken, which could entail more in-depth sample analysis and further monitoring or targeted investigations, such as to infer bioavailability in sediments (Macleod & Eriksen 2009, Fletcher et al. 2022). For example, specific investigations into environmental impacts from copper-based antifouling has included mass-balance calculations to inform the likely fate of zinc and copper from anti-foul coatings, bioavailability assessments of copper in sediments, and uptake of copper by oysters (Macleod & Eriksen 2009).

Figure 11: Stages of degradation and recovery of seafloor communities in response to organic enrichment associated with finfish aquaculture



Stage	Category	Description
I	Unimpacted	No evidence of farm impact
II	Minor effects	Slight infaunal and community change observed
III	Moderate effects	Clear change in infauna and chemistry
IV	Major effects (1)	Major change in infauna and chemistry
V	Major effects (2)	Bacterial mats evident, outgassing on disturbance
VI	Severe effects	Anoxic/abiotic, spontaneous outgassing
VII	Major effects	Monospecific fauna, major chemistry still clearly effected
VIII	Moderate effects	Fauna recovering, chemistry still clearly effected
IX	Minor effects	Largely recovered, although slight faunal/ chemical effects still apparent

Adapted from Macleod and Forbes (2004)

Therapeutants

If antibiotics or therapeutants are prescribed and administered on-site then the sediments below treatment cages, both inside and outside the lease area, are tested for residues and persistence. Residue in biota may also be tested. Sampling is typically done before, during and after treatment, or as prescribed by the EPA Director under the relevant environmental licence (Macleod & Eriksen 2009, Aquenal 2022, Aquenal 2023).

4.2.2 Broadscale

Broadscale Environmental Monitoring Programs (BEMPs) in Tasmania were developed based on research examining the potential for cumulative effects from multiple farms in a region (see Thompson et al. 2008, Ross & MacLeod 2013, Aquenal 2021, IMAS 2022, Strain et al. 2023). BEMPs have been implemented since 2009 for several growing regions and are a condition of the environmental licences for farms in the respective regions (EPA 2022a).

The BEMPs were originally centred on monitoring water column and soft sediment conditions, but in the past 5–10 years have been extended to include other receiving habitats, notably inshore and deep reefs and seagrass. The soft sediment component of the BEMPs typically require annual surveys at sampling sites spread across the region, with detailed sediment sampling to include benthic infauna communities and the full suite of physicochemical parameters – redox, sulphide, organic content, C and N content, stable isotopes, and particle size analysis (IMAS 2022).

Biogeochemical model predictions and alignment with sites and sediment information from previous studies are used to help inform site locations. The importance of good baseline information cannot be overstated, particularly in more exposed farming regions where there is greater potential for more widespread dispersion and deposition of farm derived particulate matter, either directly via feed, faeces and biofouling during cleaning or indirectly via organic matter produced from the assimilation of dissolved wastes. The absence of baseline information has the capacity to confound the interpretation of change in reference conditions (Keeley et al. 2019).

5 Summary

Deposition in the form of faecal material and uneaten feed is the major interaction pathway between finfish aquaculture and soft sediment habitats.

In summary:

- Deposition from finfish aquaculture through sedimentation of uneaten fed and faecal material leads to localised organic enrichment in soft sediment habitats, as well as potential contamination by metals and therapeutants.
- The benthic response to organic enrichment from fish farms follows a predictable succession of changes which has been well established in the literature – and typically there are highly modified benthic communities beneath the farm comprising high total abundances dominated by opportunistic deposit-feeding worms.
- Targeted research has revealed that within the generalised benthic response model, specific indicators of organic enrichment can vary according to local environmental differences including the amount of water movement and mixing in sheltered vs exposed locations, and functional adaptations of the biological communities present. These factors also influence the assimilative capacity of the soft sediment habitat, and how it recovers from organic enrichment after farming.
- More dispersive environments generally have a higher overall assimilative capacity for deposition due to stronger physical dispersal and reoxygenation, but in some areas the associated biological communities can be poorly adapted to deal with high organic loads.
- Community effects generally diminish within 150–500 metres from farms in Tasmania, depending on how dispersive the site is. Overseas studies have reported deposition into soft sediment habitats at greater distances than these in more dispersive locations.
- Careful consideration of farm siting and control of farm outputs that are appropriate to environmental carrying capacity are crucial initial steps to manage the severity of local-scale enrichment and reduce the risk of broader-scale enrichment. Tasmania's performance-based management framework that employs environmental monitoring and intervention responses based on these data is key to ongoing management.
- Targeted research into site-specific benthic response patterns has been important for developing and refining performance-based management in Tasmania.
- The implementation of broadscale monitoring is crucial for detecting potential system-wide cumulative effects, including on soft sediment habitats, and this regional-scale context has become increasingly important as finfish farming has continued to develop and expand in existing farming areas, and as farming expands into new locations.

6 Glossary and acronyms

AMBI

AZTI Marine Biotic Index. A biotic index of soft sediment health used to characterise the impacts of organic enrichment from aquaculture.

Benthic

The sediment surface and sub-surface layers at the bottom of the water body.

Biofouling

Unwanted organisms living on structures (usually man-made) in the water, e.g. algae and animals that settle on and colonise fish farm structures.

Biofouling debris

Debris from biofouling encompassing all types of organic/inorganic material originating from biofouling associated with finfish farm aquaculture.

BEMP

Broadscale Environmental Monitoring Program. A monitoring program within Tasmania to understand potential aquaculture effects within a larger spatial context, administered by the EPA.

Deposition

In the context of this review, organic material which settles through the water column. From fish farms, this includes uneaten feed pellets, faeces, and material generated from biological communities attached to the farm structures (e.g. from net-wash).

EG

Ecological groupings.

Environmental Licence

These licences to operate salmon farms in Tasmania are issued by the Environment Protection Authority under the *Environmental Management and Pollution Control Act 1994*. Environmental licences require compliance with the conditions of the licence.

EPA

Environment Protection Authority. Tasmania's principal environmental regulatory body.

Epifauna

Animals living on the surface of seabed sediments.

Hydrodynamics

Movement of fluids (in this context, seawater) within a system as affected by natural forces.

Infauna

Animals living within the sediment matrix.

Lease area

An area of water contractually allocated to a marine farming operator by the Tasmanian government.

Macrofauna

Animals above a certain size, usually those greater than 0.5–1mm.

MFDP

Marine Farming Development Plan. This plan is designed to integrate aquaculture activities with other marine and land uses while minimising impacts. It formally sets out areas of state waters the plan encompasses, marine farming zones within the plan, maximum leasable area within each zone, the types of species, and how farming is managed within each zone.

Net-wash

Material dislodged from the farm nets during cleaning.

Therapeutant

A chemical, antibiotic, hormone, pigment, antifoulant, disinfectant, cleanser or other substance used to treat or prevent conditions related to animal health.

TPDNO

Total Permissible Dissolved Nitrogen Output. This is a regulatory metric which limits the amount of dissolved inorganic nitrogen that can be released via feed inputs into a defined area over a specified time. It is based on estimated dissolved nitrogen released through waste outputs from farmed fish (EPA 2022b).

7 References

Ackefors H, Enell M (1994) The release of nutrients and organic matter from aquaculture systems in Nordic countries. J Appl Ichthyol 10:225-241

Adams TP, Black K, Black K, Carpenter T, Hughes A, Reinardy HC, Weeks RJ (2020) Parameterising resuspension in aquaculture waste deposition modelling. Aquacult Environ Interact 12:401-415

Alday V, Guichard B, Smith P, Uhland C (2006) Animal health issues: towards a risk analysis of antimicrobial use in aquaculture. In: WHO (ed) Antimicrobial Use in Aquaculture and Antimicrobial Resistance, Report of a Joint FAO/OIE/WHO Expert Consultation on Antimicrobial Use in Aquaculture and Antimicrobial Resistance. World Health Organization, Geneva, p 35-39

Anderson MJ (2008) Animal-sediment relationships re-visited: Characterising species' distributions along an environmental gradient using canonical analysis and quantile regression splines. J Exp Mar Biol Ecol 366:16-27

Aquenal (2019) Trumpeter Bay MF281 - East of Yellow Bluff: Baseline environmental assessment. Aquenal Pty Ltd, Tasmania

Aquenal (2021) Broadscale Environmental Monitoring Program - D'Entrecasteaux Channel and Huon Marine Farming Development Plan Sites, Annual Report 2020/2021. Aquenal Pty Ltd, Tasmania

Aquenal (2022) MF281 East of Yellow Bluff: Antibiotic Residue Monitoring Final Report (Surveys Round 1-3). Aquenal Pty Ltd, Tasmania

Aquenal (2023) MF109 Butlers: OTC Residue Monitoring (February 2023 medication) Final Report (Survey Rounds 1-5). Aquenal Pty Ltd, Tasmania

Aslam SN, Venzi MS, Venkatraman V, Mikkelsen Ø (2020) Chemical assessment of marine sediments in vicinity of Norwegian fish farms – A pilot study. Sci Total Environ 732:139130

AVM (2020) Veterinary chemical product use for aquaculture. Sub-Committee on Aquatic Animal Health's Aquatic Veterinary Medicines (AVM) Working Group, Canberra

Aylagas E, Atalah J, Sánchez-Jerez P, Pearman JK, Casado N, Asensi J, Toledo-Guedes K, Carvalho S (2021) A step towards the validation of bacteria biotic indices using DNA metabarcoding for benthic monitoring. Mol Ecol Resour 21:1889-1903

Aylagas E, Borja Á, Rodríguez-Ezpeleta N (2014) Environmental status assessment using DNA metabarcoding: towards a genetics based Marine Biotic Index (gAMBI). PLOS ONE 9:e90529

Aylagas E, Borja Á, Tangherlini M, Dell'Anno A, Corinaldesi C, Michell CT, Irigoien X, Danovaro R, Rodríguez-Ezpeleta N (2017) A bacterial community-based index to assess the ecological status of estuarine and coastal environments. Mar Pollut Bull 114:679-688

Backman DC, DeDominicis SL, Johnstone R (2009) Operational decisions in response to a performance-based regulation to reduce organic waste impacts near Atlantic salmon farms in British Columbia, Canada. J Clean Prod 17:374-379

Bannister J, Sievers M, Bush F, Bloecher N (2019) Biofouling in marine aquaculture: a review of recent research and developments. Biofouling 35:631-648

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 J Mar Sci 73:2408-2419

Bannister RJ, Valdemarsen T, Hansen PK, Holmer M, Ervik A (2014) Changes in benthic sediment conditions under an Atlantic salmon farm at a deep, well-flushed coastal site. Aquacult Environ Interact 5:29-47

Barros F (2016) Soft Sediment Communities. In: Kennish MJ (ed) Encyclopedia of Estuaries, Encyclopedia of Earth Sciences Series. Springer, Dordrecht, p 606

Bennett H, Elvines D (2018) 2017-2018 Annual environmental monitoring summary for the Clay Point salmon farm (Cawthron Report No. 3147). Cawthron Institute, Nelson

Bolam SG, Fernandes TF, Huxham M (2002) Diversity, biomass, and ecosystem processes in the marine benthos. Ecol Monogr 72:599-615

Borja A, Franco J, Pérez V (2000) A Marine Biotic Index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Mar Pollut Bull 40:1100-1114

Bravo F, Grant J (2018) Modelling sediment assimilative capacity and organic carbon degradation efficiency at marine fish farms. Aquacult Environ Interact 10:309-328

Broch OJ, Daae RL, Ellingsen IH, Nepstad R, Bendiksen EÅ, Reed JL, Senneset G (2017) Spatiotemporal Dispersal and Deposition of Fish Farm Wastes: A Model Study from Central Norway. Front Mar Sci 4:199

Brooks KM, Mahnken CVW (2003) Interactions of Atlantic salmon in the Pacific Northwest environment: III. Accumulation of zinc and copper. Fish Res 62:295-305

Brooks KM, Stierns AR, Backman C (2004) Seven year remediation study at the Carrie Bay Atlantic salmon (*Salmo salar*) farm in the Broughton Archipelago, British Columbia, Canada. Aquaculture 239:81-123

Brooks KM, Stierns AR, Mahnken CVW, Blackburn DB (2003) Chemical and biological remediation of the benthos near Atlantic salmon farms. Aquaculture 219:355-377

Burdige DJ (2006) Geochemistry of Marine Sediments. Princeton University Press, Woodstock

Bureau DP, Hua K (2010) Towards effective nutritional management of waste outputs in aquaculture, with particular reference to salmonid aquaculture operations. Aquaculture Research 41:777-792

Burridge L, Weis JS, Cabello F, Pizarro J, Bostick K (2010) Chemical use in salmon aquaculture: A review of current practices and possible environmental effects. Aquaculture 306:7-23

Byers JE, Grabowski JH (2014) Soft-sediment communities. In: Bertness MD, Bruno JF, Silliman BR, Stachowicz JJ (eds) Marine Community Ecology and Conservation. Sinauer Associates, Massachusetts, p 227-249

Callier MD, McKindsey CW, Desrosiers G (2008) Evaluation of indicators used to detect mussel farm influence on the benthos: Two case studies in the Magdalen Islands, Eastern Canada. Aquaculture 278:77-88

Capone DG, Weston DP, Miller V, Shoemaker C (1996) Antibacterial residues in marine sediments and invertebrates following chemotherapy in aquaculture. Aquaculture 145:55-75

Carvajalino-Fernández MA, Keeley NB, Fer I, Law BA, Bannister RJ (2020a) Effect of substrate type and pellet age on the resuspension of Atlantic salmon faecal material. Aquacult Environ Interact 12:117-129

Carvajalino-Fernández MA, Sævik PN, Johnsen IA, Albretsen J, Keeley NB (2020b) Simulating particle organic matter dispersal beneath Atlantic salmon fish farms using different resuspension approaches. Mar Pollut Bull 161:111685

CBOI (2013) Final Conditions as amended by Addendum dated 13 March 2013; Resource Consent for Richmond [Kopaua] issued to New Zealand King Salmon, Coastal Permit under the Resource Management Act 1991. Notice by the Chairman of The Board of Inquiry (CBOI), Auckland, NZ

Chamberlain J, Stucchi D (2007) Simulating the effects of parameter uncertainty on waste model predictions of marine finfish aquaculture. Aquaculture 272:296-311

Champeau O (2013) Chapter 10: Effects from Additives. In: Boustead N (ed) Literature review of ecological effects of aquaculture. A collaboration between Ministry for Primary Industries, Cawthron Institute and National Institute for Water and Atmospheric Research Ltd. Ministry for Primary Industries, Wellington

Chang BD, Coombs KA, Page FH (2014) The development of the salmon aquaculture industry in southwestern New Brunswick, Bay of Fundy, including steps toward integrated coastal zone management. Aquacult Econ Manage 18:1-27

Chang BD, Page FH (2011) Analysis of results from the Environmental Management Program Tier 1 monitoring of salmon farms in southwestern New Brunswick, Bay of Fundy: Relationships between sediment sulfide concentration and selected parameters, 2002-2008. Fisheries and Oceans Canada, New Brunswick

Chary K, Callier MD, Covès D, Aubin J, Simon J, Fiandrino A (2021) Scenarios of fish waste deposition at the sub-lagoon scale: a modelling approach for aquaculture zoning and site selection. ICES J Mar Sci 78:922-939

Clement D, Keeley N, Sneddon R (2010) Ecological Relevance of Copper (Cu) and Zinc (Zn) in Sediments Beneath Fish Farms in New Zealand (Report No. 1805). Cawthron Institute, Nelson

Condie SA, Gorton R, Hadley S, Little R, Macleod C, Ogier E, Proctor W, Ross DJ, Sporcic M, Wild-Allen K (2017) INFORMD2: Risk-based tools supporting consultation, planning and adaptive management for aquaculture and other multiple-uses of the coastal waters of southern Tasmania (FRDC Project No 2012/024). CSIRO Oceans and Atmosphere, Hobart

Coutts A, Zimmermann D, Davey A, Bowman JP, Ross DJ, Strain EMA (2024) A comparison of visual and molecular methods for inferring biological communities in aquaculture enriched sediments - Impact assessment and cost-benefit analysis. Mar Pollut Bull 209:117172

Cranford P, Brager L, Elvines D, Wong D, Law B (2020) A revised classification system describing the ecological quality status of organically enriched marine sediments based on total dissolved sulfides. Mar Pollut Bull 154:111088

Cranford PJ, Brager L, Wong D (2017) A dual indicator approach for monitoring benthic impacts from organic enrichment with test application near Atlantic salmon farms. Mar Pollut Bull 124:258-265

Crawford C (2003) Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture 226:129-138

Crawford C, Macleod C, Moltschaniwskyj NA, Forbes E (2008) Recovery of sediments associated with fin fish cage aquaculture - are some systems better adapted? In: World Aquaculture Society (ed) Book of Abstracts. World Aquaculture Society Conference, Busan, Korea

Crawford C, Thompson P, Jordan A, Foster S, Mitchell I, Bonham P, Willcox S (2006) Development of broad scale environmental monitoring and baseline surveys in relation to sustainable salmon aquaculture in the D'Entrecasteaux Channel region (Aquafin CRC Project 4.41). Tasmanian Aquaculture and Fisheries Institute, Hobart

Crawford CM, Mitchell IM, Macleod CKA (2001) Video assessment of environmental impacts of salmon farms. ICES J Mar Sci 58:445-452

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

Da Silva RRP, White CA, Bowman JP, Ross DJ (2022) Composition and functionality of bacterioplankton communities in marine coastal zones adjacent to finfish aquaculture. Mar Pollut Bull 182:113957

Dean RJ, Shimmield TM, Black KD (2007) Copper, zinc and cadmium in marine cage fish farm sediments: An extensive survey. Environ Pollut 145:84-95

Deng Y, Tan A, Zhao F, Wang F, Gong H, Lai Y, Huang Z (2025) Global distribution of antimicrobial resistance genes in aquaculture. One Health Advances 3:6

DNRET (2023a) Environmental Standards for Tasmanian Marine Finfish Farming 2023. Environment Strategic Business Unit, Department of Natural Resources and Environment Tasmania, Hobart

DNRET (2023b) Companion Paper – Environmental Standards for Tasmanian Marine Finfish Farming 2023. Environment Strategic Business Unit, Department of Natural Resources and Environment Tasmania, Hobart

DPIPWE (2016) Marine Farming Licence No 261 – Storm Bay off Trumpeter Bay – 1 December 2016. Water and Marine Resources Division, Department of Primary Industries, Parks, Water and Environment, Tasmania

DPIPWE (2017) Storm Bay North Marine Farming Development Plan (MFDP) November 2017. Water and Marine Resources Division, Department of Primary Industries, Parks, Water and Environment, Tasmania

DPIPWE (2018) Storm Bay off Trumpeter Bay North Bruny Island Marine Farming Development Plan August 2018. Water and Marine Resources Division, Department of Primary Industries, Parks, Water and Environment, Tasmania

DPIWE (2002) Huon River & Port Esperance Marine Farming Development Plan (MFDP) February 2002. Food, Agriculture and Fisheries Division, Department of Primary Industries, Water and Environment, Tasmania

Edgar GJ, Barrett NS, Graddon DJ (1999) A classification of Tasmanian estuaries and assessment of their conservation significance using ecological and physical attributes, population and land use. Tasmanian Aquaculture and Fisheries Institute, Hobart

Edgar GJ, Davey A, Shepherd C (2009) Broadscale effects of marine salmonid aquaculture and introduced pests on macrobenthos and the sediment environment in Tasmania between 1998 and 2003. Tasmanian Aquaculture and Fisheries Institute, Hobart

Edgar GJ, Davey A, Shepherd C (2010) Application of biotic and abiotic indicators for detecting benthic impacts of marine salmonid farming among coastal regions of Tasmania. Aquaculture 307:212-218

Edgar G, Macleod C, Mawbey RB, Shields D (2005) Broad-scale effects of marine salmonid aquaculture on macrobenthos and the sediment environment in southeastern Tasmania. J Exp Mar Biol Ecol 327:70-90

Ehrnsten E, Sun X, Humborg C, Norkko A, Savchuk OP, Slomp CP, Timmermann K, Gustafsson BG (2020) Understanding environmental changes in temperate coastal seas: linking models of benthic fauna to carbon and nutrient fluxes. Front Mar Sci 7:450

Elvines DM, MacLeod CK, Ross DJ, Hopkins GA, White CA (2023) Fate and effects of fish farm organic waste in marine systems: Advances in understanding using biochemical approaches with implications for environmental management. Rev Aquacult 16:66-85

Elvines D, Morrisey D, Smeaton M, Floerl L, Keeley N (2021a) Assessment of seabed effects for proposed ‘Blue Endeavour’ (revised) open ocean salmon farm (Cawthron Report No. 3489). Cawthron Institute, Nelson

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 (Cawthron Report No. 3521). Cawthron Institute, Nelson

Elvines DM, Smeaton M, Ross DJ, White CA, MacLeod CK, Bury SJ, Brown JCS, Hopkins GA (2024) Performance of biochemical tools as fish waste particle tracers in a dispersive marine system. Ecol Indic 158:111474

Elvines D, Taylor D, Newcombe E, Berthelsen A (2016) Environmental impacts of the Forsyth Bay salmon farm: annual monitoring 2015 (Cawthron Report No. 2783). Cawthron Institute, Nelson

EPA (2022a) Review of Tasmanian and International Regulatory Requirements for Salmonid Aquaculture. Environment Protection Authority, Hobart

EPA (2022b) TPDNO explanatory paper: compliance assessment methodology. Environment Protection Authority, Hobart

EPA (2024) Draft Technical Standard – Soft Sediment Video Surveys for Marine Finfish Farms. Environment Protection Authority, Hobart

Fenchel T, Bernard C (1995) Mats of colourless sulphur bacteria. I. Major microbial processes. Mar Ecol Prog Ser 128:161-170

Fernandez-Jover D, Martinez-Rubio L, Sanchez-Jerez P, Bayle-Sempere JT, Jimenez JAL, Lopez FJM, Bjørn P-A, Uglem I, Dempster T (2011) Waste feed from coastal fish farms: A trophic subsidy with compositional side-effects for wild gadoids. Estuar Coast Shelf Sci 91:559-568

Fisheries and Oceans Canada (2017) Regulating and monitoring British Columbia's marine finfish aquaculture facilities 2015-2016. Fisheries and Oceans Canada, Ottawa

Fletcher L, Atalah J, Elvines D (2018) Annual environmental monitoring at the Forsyth Bay, Waihinau Bay, Otanerau Bay and Ruakaka Bay salmon farms 2017 (Cawthron Report No. 3124). Cawthron Institute, Nelson

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

Fletcher L, Mitterwallner P, McGrath E, Laroche O, McMullin R, Falk H, Pochon X (2024) 2023–2024 Annual environmental monitoring summary for the Te Pangu salmon farm (Cawthron Report No. 4035). Cawthron Institute, Nelson

Fox C, Hicks N, Webb C, Grant J, Brain S, Fraser S, Abell R (2022) INCREASE and NAMAQI project report: Improving understanding of fish farm organic waste dispersal in higher energy environments (SAMS Report No. 313). Scottish Association for Marine Science, Oban

FSANZ (1998) Review of the Food Standards Code. Development of joint Australia New Zealand Food Standards. The establishment of guideline levels for contaminants (metals) in food: Generally Expected Levels (GELs). Food Standards Australia New Zealand

Giles H, Baxter A, Taylor D, Elvines D, Neale D, Jorgensen E, James M, Bunce M, Broekhuizen N, Wade O, Ford R, Cumming S, O'Connell-Milne S, Hart S-R, Hampton B (2021) Best practice guidelines for benthic and water quality monitoring of open ocean finfish culture in New Zealand (New Zealand Aquatic Environment and Biodiversity Report No. 278). Fisheries New Zealand, Ministry for Primary Industries, Wellington

Grall J, Glemaréc M (1997) Using Biotic Indices to Estimate Macrobenthic Community Perturbations in the Bay of Brest. Estuar Coast Shelf S 44:43-53

Gray JP, Herwig RP (1996) Phylogenetic analysis of the bacterial communities in marine sediments. Appl Environ Microbiol 62:4049-4059

Hadley S, Macleod C, Ross DJ (2017) Nutrient Dispersion Modelling for Proposed Marine Finfish Farming Zones in Storm Bay. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Hadley S, Reimer T, Ross J (2025) Storm Bay Observing System: Assessing the Performance of Aquaculture Development. Objective 3: Development and evaluation of a lease scale model for assessing the environmental footprint of dissolved and particulate farm inputs (FRDC Project No. 2018/131). Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Hale R, Depree C, Broekhuizen N (2023) Simulating fish farm enrichment and fallowing impacts reveals unequal biogeochemical recovery of benthic variables. Aquacult Environ Interact 15:115-131

Hamoutene D, Salvo F, Egli SN, Modir-Rousta A, Knight R, Perry G, Bottaro CS, Dufour SC (2018) Measurement of aquaculture chemotherapeutants in flocculent matter collected at a hard-bottom dominated finfish site on the South Coast of Newfoundland (Canada) after 2 years of fallow. Front Mar Sci 5(228):1-11

Hamoutene D, Sheppard L, Mersereau J, Oldford V, Bungay T, Salvo F, Dufour S, Mabrouk G (2014) Applicability of the use of visual indicators [presence of Beggiatoa and/or Opportunistic Polychaete Complexes (OPC)] to identify benthic changes due to aquaculture on various substrates (Research Document 2014/063). Fisheries and Oceans Canada, Ottawa

Han QF, Zhang XR, Xu XY, Wang XL, Yuan XZ, Ding ZJ, Zhao S, Wang SG (2021) Antibiotics in marine aquaculture farms surrounding Laizhou Bay, Bohai Sea: Distribution characteristics considering various culture modes and organism species. Sci Total Environ 760:143863

Hansen PK, Ervik A, Schaanning M, Johannessen P, Aure J, Jahnsen T, Stigebrand A (2001) Regulating the local environmental impact of intensive, marine fish farming II. The monitoring programme of the MOM system (Modelling-Ongrowing fish farms-Monitoring). Aquaculture 194:75-92

Hargrave BT, Holmer M, Newcombe CP (2008) Towards a classification of organic enrichment in marine sediments based on biogeochemical indicators. Mar Pollut Bull 56: 810-824

Holmer M, Hansen PK, Karakassis I, Borg JA, Schembri PJ (2008) Chapter 2: Monitoring of Environmental Impacts of Marine Aquaculture. In: Holmer M, Black K, Duarte CM, Marbà N, Karakassis I (eds) Aquaculture in the Ecosystem. Springer, Dordrecht, p 47-85

Holmer M, Kristensen E (1992) Impact of marine fish cage farming on metabolism and sulfate reduction of underlying sediments. Mar Ecol Prog Ser 80:191-201

Huon Aquaculture Company (2018) Environmental Impact Statement to accompany the Draft Amendment No. 3 to the Storm Bay off Trumpeter Bay North Bruny Island, Marine Farming Development Plan, July 1998. Huon Aquaculture Company Pty Ltd, Hobart

Husa V, Kutti T, Ervik A, Sjøtun K, Hansen PK, Aure J (2014) Regional impact from fin-fish farming in an intensive production area (Hardangerfjord, Norway). Mar Biol Res 10:241-252

IMAS (2022) Annual Broadscal Environmental Monitoring Program 2021/2022 for Environmental Licence 10180/1 Marine Farming Lease No. 281 at East of Yellow Bluff & Environmental Licence 10211/1 Marine Farming Lease No. 279 at West of Wedge Island. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Jordan A, Samson C, Lawler MM, Lucieer V (2003) Assessment of seabed habitats and heavy metals in Cornelian Bay. Tasmanian Aquaculture and Fisheries Institute, Hobart

Kalantzi I, Karakassis I (2006) Benthic impacts of fish farming: Meta-analysis of community and geochemical data. Mar Pollut Bull 52:484-493

Kalantzi I, Rico A, Mylona K, Pergantis SA, Tsapakis M (2021) Fish farming, metals and antibiotics in the eastern Mediterranean Sea: Is there a threat to sediment wildlife? Sci Total Environ 764:142843

Karakassis I (2001) Ecological effects of fish farming in the Mediterranean. Cah Options Mediterr 55:15-22

Keeley N (2013) Quantifying and predicting benthic enrichment: Lessons learnt from southern temperate aquaculture systems. PhD dissertation, Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Keeley NB, Cromey CJ, Goodwin EO, Gibbs MT, Macleod CM (2013b) Predictive depositional modelling (DEPOMOD) of the interactive effect of current flow and resuspension on ecological impacts beneath salmon farms. Aquacult Environ Interact 3:275-291

Keeley NB, Forrest BM, Crawford C, Macleod CK (2012a) Exploiting salmon farm benthic enrichment gradients to evaluate the regional performance of biotic indices and environmental indicators. Ecol Indic 23:453-466

Keeley NB, Forrest BM, Macleod CK (2013a) Novel observations of benthic enrichment in contrasting flow regimes with implications for marine farm monitoring and management. Mar Pollut Bull 66:105-116

Keeley NB, Forrest BM, MacLeod CK (2015) Benthic recovery and re-impact responses from salmon farm enrichment: Implications for farm management. Aquaculture 435:412-423

Keeley NB, Macleod CK, Forrest BM (2012b) Combining best professional judgement and quantile regression splines to improve characterisation of macrofaunal responses to enrichment. Ecol Indic 12:154-166

Keeley NB, Macleod CK, Hopkins GA, Forrest BM (2014) Spatial and temporal dynamics in macrobenthos during recovery from salmon farm induced organic enrichment: When is recovery complete? Mar Pollut Bull 80:250-262

Keeley N, Valdemarsen T, Strohmeier T, Pochon X, Dahlgren T, Bannister R (2020) Mixed-habitat assimilation of organic waste in coastal environments - It's all about synergy! Sci Total Environ 699:134281

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. Aquacult Environ Interact 11:161-179

Keeley N, Wood SA, Pochon X (2018) Development and preliminary validation of a multi-trophic metabarcoding biotic index for monitoring benthic organic enrichment. Ecol Indic 85:1044-1057

Kutti T, Ervik A, Hansen PK (2007a) Effects of organic effluents from a salmon farm on a fjord system. I. Vertical export and dispersal processes. Aquaculture 262:367-381

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

Law BA, Hill PS, Maier I, Milligan TG, Page F (2014) Size, settling velocity and density of small suspended particles at an active salmon aquaculture site. Aquacult Environ Interact 6:29-42

Law BA, Hill PS, Milligan TG, Zions V (2016) Erodibility of aquaculture waste from different bottom substrates. Aquacult Environ Interact 8:575-584

Leal JF, Santos EBH, Esteves VI (2019) Oxytetracycline in intensive aquaculture: water quality during and after its administration, environmental fate, toxicity and bacterial resistance. Rev Aquacult 11:1176-1194

Loucks RH, Smith RE, Fisher CV, Fisher EB (2012) Copper in the sediment and sea surface microlayer near a fallowed, open-net fish farm. Mar Pollut Bull 64:1970-1973

Lucieer V, Walsh P, Monk J, Flukes E (2025) Seamap Australia National Benthic Habitat Layer (NBHL) 2025. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Macdonald TA, Burd BJ, van Roodselaar A (2012) Size structure of marine soft-bottom macrobenthic communities across natural habitat gradients: Implications for productivity and ecosystem function. PLOS ONE 7:e40071

Macleod C, Bissett A, Burke C, Forbes S, Holdsworth D, Nichols P, Revill A, Volkman J (2004a) Development of novel methods for the assessment of sediment condition and determination of management protocols for sustainable finfish cage aquaculture operations (Aquafin CRC Project 4.1). Tasmanian Aquaculture and Fisheries Institute, Hobart

Macleod CK, Crawford CM, Moltschaniwskyj NA (2004b) Assessment of long term change in sediment condition after organic enrichment: defining recovery. Mar Pollut Bull 49:79-88

Macleod C, Eriksen R (2009) A review of the ecological impacts of selected antibiotics and antifoulants currently used in the Tasmanian salmonid farming industry (marine farming phase), FRDC Project 2007/246 – Final Report. Tasmanian Aquaculture and Fisheries Institute, Hobart

Macleod CK, Eriksen RS, Davey A, Kelly B, Ross DJ (2014) Long-term Recovery - Review of sediment condition at Marine Farm lease No. 76 (Gunpowder Jetty), North West Bay. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Macleod C, Forbes SE (2003) Development of quantitative visual assessment techniques for the assessment of sediment condition at marine finfish farms. Biodiversity and Conservation Conference, Hobart

Macleod C, Forbes S (2004) Guide to the assessment of sediment condition at marine finfish farms in Tasmania. Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Hobart

Macleod C, Forbes SE, Volkman J, Revill A, Holdsworth D, Burke CM (2002) Development of novel methods for the assessment of sediment condition and determination of management protocols for sustainable finfish cage aquaculture operations. In: Battaglene SC, Cobcroft JM (eds) Second Scientific Conference of the Atlantic Salmon Aquaculture Subprogram – Handbook. Atlantic Salmon Aquaculture Subprogram, Hobart

Macleod C, Moltschaniwskyj NA, Crawford C, Forbes SE (2006b) Site Selection: Further Considerations for Environmental Sustainability of Cage Aquaculture. Australasian Aquaculture Conference, Adelaide

Macleod CK, Moltschaniwskyj NA, Crawford CM (2006a) Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages. Mar Pollut Bull 52:1458-1466

Macleod CK, Moltschaniwskyj NA, Crawford CM, Forbes SE (2007) Biological recovery from organic enrichment: some systems cope better than others. Mar Ecol Prog Ser 342:41-53

Magna EK, Tornyie F, Ofosu-Koranteng F, Mensah ET-D, Larbi L, Mbage B (2024) Occurrence and environmental risk assessment of antibiotics in water and sediment from fish farms in the lower Volta Lake of Ghana. Chemistry Africa 7:409-419

Maroni K (2000) Monitoring and regulation of marine aquaculture in Norway. J Appl Ichthyol 16:192-195

McGrath E, Bennett H, Campos C (2019) 2018-2019 Annual environmental monitoring summary for the Te Pangu Bay salmon farm (Cawthron Report No. 3324). Cawthron Institute, Nelson

McGrath E, Berthelsen A, Campos C, Scheel M, Major R (2020) 2019-2020 Annual environmental monitoring summary for the Te Pangu Bay salmon farm (Cawthron Report No. 3509). Cawthron Institute, Nelson

McMullin RM, Sabadel AJM, Hageman KJ, Wing SR (2021) A quantitative analysis of organic matter inputs to soft sediment communities surrounding salmon farms in the Marlborough Sounds region of New Zealand. Sci Total Environ 773:145146

MDC (2016) Certificate of Resource Consent; Number U150081; Te Pangu Bay, Tory Channel. Decision of Marlborough District Council (MDC) under the Resource Management Act 1991 for The New Zealand King Salmon Company Limited. Marlborough District Council, Blenheim

Miranda CD, Godoy FA, Lee MR (2018) Current status of the use of antibiotics and the antimicrobial resistance in the Chilean salmon farms. Front Microbiol 9:1284

Morrisey D, Turner S, MacDiarmid AB (1998) Subtidal assemblages of soft substrata. In: Kingsford M, Battershill C (eds) Studying temperate marine environments: A handbook for ecologists. Canterbury University Press, Christchurch, p 194-226

Morro B, Davidson K, Adams TP, Falconer L, Holloway M, Dale A, Aleynik D, Thies PR, Khalid F, Hardwick J, Smith H, Gillibrand PA, Rey-Planellas S (2022) Offshore aquaculture of finfish: Big expectations at sea. Rev Aquacult 14:791-815

MPI (2023) Siting and management guidelines for open ocean aquaculture: Ecological considerations for benthic environments and water quality. Ministry for Primary Industries, Wellington

Muniz P, Venturini N, Pires-Vanin AMS, Tommasi LR, Borja Á (2005) Testing the applicability of a Marine Biotic Index (AMBI) to assessing the ecological quality of soft-bottom benthic communities, in the South America Atlantic region. Mar Pollut Bull 50:624-637

Muxika I, Borja Á, 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. Mar Pollut Bull 55:16-29

Nowak BF, Hayward CJ, González L, Bott NJ, Lester RJG (2011) Sea lice infections of salmonids farmed in Australia. Aquaculture 320:171-177

NSDFA (2011) Environmental monitoring program framework for marine aquaculture in Nova Scotia. Nova Scotia Department of Fisheries and Aquaculture, Halifax

Oppedal F, Dempster T, Stien LH (2011) Environmental drivers of Atlantic salmon behaviour in sea-cages: A review. Aquaculture 311:1-18

Pearson TH, Rosenberg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr Mar Biol Annu Rev 16:229-311

Petuna Aquaculture (2022) Antibiotic residue survey report MF178 final report July 2022. Petuna Aquaculture Pty Ltd, Devonport

Pochon X, Atalah J, Fletcher L, Laroche O, Elvines D, Zaiko A, Wood S, Keeley N (2021) A molecular index refined for benthic monitoring of salmon farms in the Marlborough Sounds (Cawthron Report No. 3602). Cawthron Institute, Nelson

Price C, Black KD, Hargrave BT, Morris Jr JA (2015) Marine cage culture and the environment: effects on water quality and primary production. Aquacult Environ Interact 6:151-174

Read P, Fernandes T (2003) Management of environmental impacts of marine aquaculture in Europe. Aquaculture 226:139-163

Rector ME, Weitzman J, Filgueira R, Grant J (2022) Environmental indicators in salmon aquaculture research: A systematic review. Rev Aquacult 14:156-177

Robinson SMC, Auffrey LM, Barbeau MA (2005) Far-field impacts of eutrophication on the intertidal zone in the Bay of Fundy, Canada with emphasis on the soft-shell clam, *Mya arenaria*. In: Hargrave BT (ed) Environmental Effects of Marine Finfish Aquaculture. Springer Berlin, Heidelberg, p 253-274

Rosenberg R (2001) Marine benthic faunal successional stages and related sedimentary activity. Sci Mar 65:107-119

Ross J, Bush F, White C, Hortle J, Davey A, Beard J, Macleod C (2025) Guide to the environmental monitoring and assessment for finfish aquaculture. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Ross DJ, McCarthy A, Davey A, Pender A, Macleod CM (2016) Understanding the Ecology of Dorvilleid Polychaetes in Macquarie Harbour: Response of the benthos to organic enrichment from finfish aquaculture. FRDC Project No. 2014/038. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Ross DJ, Macleod CK (2013) Evaluation of Broadscale Environmental Monitoring Program (BEMP) data from 2009-2012. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Ross J, Macleod C, White C, Hadley S, Moreno D, Bush F, Barrett N (2021) Managing ecosystem interactions across differing environments: building flexibility and risk assurance into environmental management strategies. FRDC Project No. 2015/024. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Ross DJ, Moreno D, Bell JD, Mardones J, Beard J (2022) Assessment of the Macquarie Harbour Broadscale Environmental Monitoring Program (BEMP) data from 2011-2020. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Ross LG, Telfer TC, Falconer L, Soto D, Aguilar-Manjarrez J, Asmah R, Bermúdez J, Beveridge MCM, Byron CJ, Clément A, Corner R, Costa-Pierce BA, Cross S, De Wit M, Dong S, Ferreira JG, Kapetsky JM, Karakassis I, Leschen W, Little D, Lundebye A-K, Murray FJ, Phillips M, Ramos L, Sadek S, Scott PC, Valle-levinson A, Waley D, White PG, Zhu C (2013) Carrying capacities and site selection within the ecosystem approach to aquaculture. In: Ross LG, Telfer TC, Falconer L, Soto D, Aguilar-Manjarrez J (eds) Site selection and carrying capacities for inland and coastal aquaculture. FAO/Institute of Aquaculture, University of Stirling, Expert Workshop. FAO Fisheries and Aquaculture Proceedings, Rome, p 19-46

Rožič PŽ, Dolenec T, Baždarić B, Karamarko V, Kniewald G, Dolenec M (2012) Major, minor and trace element content derived from aquacultural activity of marine sediments (Central Adriatic, Croatia). Environ Sci Pollut Res 19:2708-2721

Sanz-Lazaro C, Casado-Coy N, Calderero EM, Villamar UA (2021) The environmental effect on the seabed of an offshore marine fish farm in the tropical Pacific. J Environ Manage 300:113712

Sanz-Lázaro C, Marín A (2011) Diversity patterns of benthic macrofauna caused by marine fish farming. Diversity 3:176-199

Sarà G, Lo Martire M, Sanfilippo M, Pulicanò G, Cortese G, Mazzola A, Manganaro A, Pusceddu A (2011) Impacts of marine aquaculture at large spatial scales: Evidences from N and P catchment loading and phytoplankton biomass. Mar Environ Res 71:317-324

Schumann M, Brinker A (2020) Understanding and managing suspended solids in intensive salmonid aquaculture: a review. Rev Aquacult 12:2109-2139

Sim B-R, Kim HC, Kang S, Yoon S-P, Lee DI, Lee W-C (2024) Chemical recovery of a fish farm in Jinju Bay of South Korea after cessation of long-term fish farming. Ocean Coast Manage 251:107083

Simone M, Grant J (2020) Visually-based alternatives to sediment environmental monitoring. Mar Pollut Bull 158:111367

Sim-Smith C, Forsythe A (2013) Comparison of the international regulations and best management practices for marine finfish farming (MPI Technical Paper No. 2013/47). Ministry for Primary Industries, Wellington

SLR (2023) 2022 annual monitoring of low-flow salmon farm sites in the Marlborough Sounds. SLR Consulting NZ Limited, Nelson

Snelgrove PVR (1999) Getting to the bottom of marine biodiversity: Sedimentary habitats: Ocean bottoms are the most widespread habitat on Earth and support high biodiversity and key ecosystem services. BioScience 49:129-138

Snelgrove PVR, Butman CA (1995) Animal-sediment relationships revisited: cause versus effect. Oceanogr Lit Rev 8:668

Stigebrandt A, Aure J, Ervik A, Hansen PK (2004) Regulating the local environmental impact of intensive marine fish farming III. A model for estimation of the holding capacity in the Modelling–Ongrowing fish farm–Monitoring system. Aquaculture 234:239-261

Strain E, White C, Hartog M, Hortle J, Coutts A, Brasier M, Hadley S, Ross J (2023) The Storm Bay Observing System: An evaluation of the sampling parameters and design for assessing the performance of salmon aquaculture. FRDC Project No. 2018/131. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Strain E, White C, Ross J (2020) The Storm Bay Observing System: Preliminary review of the sampling parameters and design for assessing the performance of salmon aquaculture. FRDC Project No. 2018/131. Institute for Marine and Antarctic Studies, University of Tasmania, Hobart

Sutherland TF, Sterling AM, Shaw KL, Blasco NNJ, Bradford MJ (2019) Detecting indicator taxa associated with benthic organic enrichment using different video camera orientations. J Coast Res 35:467-479

Sweetman AK, Norling K, Gunderstad C, Haugland BT, Dale T (2014) Benthic ecosystem functioning beneath fish farms in different hydrodynamic environments. Limnol Oceanogr 59:1139-1151

Talman S, O'Connor N, Zampatti B, Cannon F (1996) Monitoring of benthic invertebrates in Macquarie Harbour, western Tasmania. Supervising Scientist, Canberra

Thompson P, Bonham P, Willcox S, Crawford C (2005) Baseline monitoring in D'Entrecasteaux Channel (Aquafin CRC Project 4.2). CSIRO Marine and Atmospheric Research, Hobart

Thompson P, Wild-Allen K, Macleod C, Swadling K, Blackburn S, Skerratt J, Volkman J (2008) Monitoring the Huon Estuary and D'Entrecasteaux Channel for the Environmental Effects of Finfish Aquaculture (Aquafin CRC Project 4.2[2]). CSIRO Marine and Atmospheric Research, Hobart

Teixeira H, Weisberg SB, Borja A, Ranasinghe JA, Cadien DB, Velarde RG, Lovell LL, Pasko D, Phillips CA, Montagne DE, Ritter KJ, Salas F, Marques JC (2012) Calibration and validation of the AZTI's Marine Biotic Index (AMBI) for southern California marine bays. Ecol Indic 12:84-95

Tomassetti P, Porrello S (2005) Polychaetes as indicators of marine fish farm organic enrichment. Aquacult Int 13:109-128

Troell M, Pihl L, Rönnbäck P, Wennhage H, Söderqvist T, Kautsky N (2005) Regime shifts and ecosystem services in Swedish coastal soft bottom habitats: when resilience is undesirable. Ecol Soc 10:30

TSGA (2013) Environmental Best Management Practice Guideline for In situ Net Cleaning of Salmon Cages using Marine Inspector Cleaner (MIC). Tasmanian Salmonid Growers Association, Tasmania

Valdemarsen T, Bannister RJ, Hansen PK, Holmer M, Ervik A (2012) Biogeochemical malfunctioning in sediments beneath a deep-water fish farm. Environ Pollut 170:15-25

Volkman JK, Thompson P, Herzfeld M, Wild-Allen K, Blackburn S, Macleod C, Swadling K, Foster S, Bonham P, Holdsworth D, Clementson L, Skerratt J, Rosebrock U, Andrewartha J, Revill A (2009) A whole-of-ecosystem assessment of environmental issues for salmonid aquaculture (Aquafin CRC Project 4.2[2]). CSIRO Marine and Atmospheric Research, Hobart

Wang C-D, Olsen Y (2024) Monitoring regional benthic environment of Norwegian salmon cage farms. Aquacult Environ Interact 16:71-90

Wang X, Olsen LM, Reitan KI, Olsen Y (2012) Discharge of nutrient wastes from salmon farms: environmental effects, and potential for integrated multi-trophic aquaculture. Aquacult Environ Interact 2:267-283

Weitzman J, Steeves L, Bradford J, Filgueira R (2019) Chapter 11 - Far-Field and Near-Field Effects of Marine Aquaculture. In: Sheppard C (ed) World Seas: An Environmental Evaluation, Volume III: Ecological Issues and Environmental Impacts. Elsevier Academic Press, Oxford, p 197-220

White CA, Bannister RJ, Dworjanyn SA, Husa V, Nichols PD, Dempster T (2018) Aquaculture-derived trophic subsidy boosts populations of an ecosystem engineer. Aquacult Environ Interact 10:279-289

White CA, Nichols PD, Ross DJ, Dempster T (2017) Dispersal and assimilation of an aquaculture waste subsidy in a low productivity coastal environment. Mar Pollut Bull 120:309-321

White CA, Woodcock SH, Bannister RJ, Nichols PD (2019) Terrestrial fatty acids as tracers of finfish aquaculture waste in the marine environment. Rev Aquacult 11:133-148

White PG (2013) Environmental consequences of poor feed quality and feed management. In: Hasan MR, New MB (eds) On-farm feeding and feed management in aquaculture. FAO, Rome, p 553-564

Wild-Allen K (2008) Huon Estuary and D'Entrecasteaux Channel Biogeochemical Model Scenario Simulations for 2002 and 2009: Farm Impacts on Seasonal Pelagic Biogeochemistry (Aquafin CRC Project 4.2[2]). CSIRO Marine and Atmospheric Research, Hobart

Wild-Allen K, Parslow J, Herzfeld M, Sakov P, Andrewartha J, Rosebrock U (2005) Biogeochemical Modelling of the D'Entrecasteaux Channel and Huon Estuary (Aquafin CRC Project 4.2). CSIRO Marine and Atmospheric Research, Hobart

Wildish DJ, Akagi HM, Hamilton N, Hargrave BT (1999) A recommended method for monitoring sediments to detect organic enrichment from mariculture in the Bay of Fundy. Fisheries and Oceans Canada, New Brunswick

Wildish DJ, Hargrave BT, Pohle G (2001) Cost-effective monitoring of organic enrichment resulting from salmon mariculture. ICES J Mar Sci 58:469-476

Wollenberger L, Halling-Sørensen B, Kusk KO (2000) Acute and chronic toxicity of veterinary antibiotics to *Daphnia magna*. Chemosphere 40:723-730

Woodcock SH, Meier S, Keeley NB, Bannister RJ (2019) Fate and longevity of terrestrial fatty acids from caged fin-fish aquaculture in dynamic coastal marine systems. Ecol Indic 103:43-54

Woodcock SH, Strohmeier T, Strand Ø, Olsen SA, Bannister RJ (2018) Mobile epibenthic fauna consume organic waste from coastal fin-fish aquaculture. Mar Environ Res 137:16-23

Woods G, Brain E, Shepherd C, Paice T (2004) Tasmanian Marine Farming Environmental Monitoring Report: Benthic Monitoring (1997 - 2002). Department of Primary Industries, Water and Environment, Hobart

Wu RSS (1995) The environmental impact of marine fish culture: Towards a sustainable future. Mar Pollut Bull 31:159-166

Wynne JW, Stratford C, Slinger J, Samsing F, Rigby M, McCulloch R, Quezada-Rodriguez P, Taylor RS (2020) The interaction between temperature and dose on the efficacy and biochemical response of Atlantic salmon to hydrogen peroxide treatment for amoebic gill disease. J Fish Dis 43:39-48

Ye L-X, Ritz DA, Fenton GE, Lewis ME (1991) Tracing the influence on sediments of organic waste from a salmonid farm using stable isotope analysis. J Exp Mar Biol Ecol 145:161-174

Yokoyama H, Inoue M, Abo K (2007) Macrobenthos, current velocity and topographic factors as indicators to assess the assimilative capacity of fish farms: Proposal of two indices. Bull Fish Res Agency Jpn 19:89-96

UNIVERSITY of TASMANIA

IMAS

Institute for Marine and Antarctic Studies