Fisheries New Zealand
Tini a Tangaroa

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

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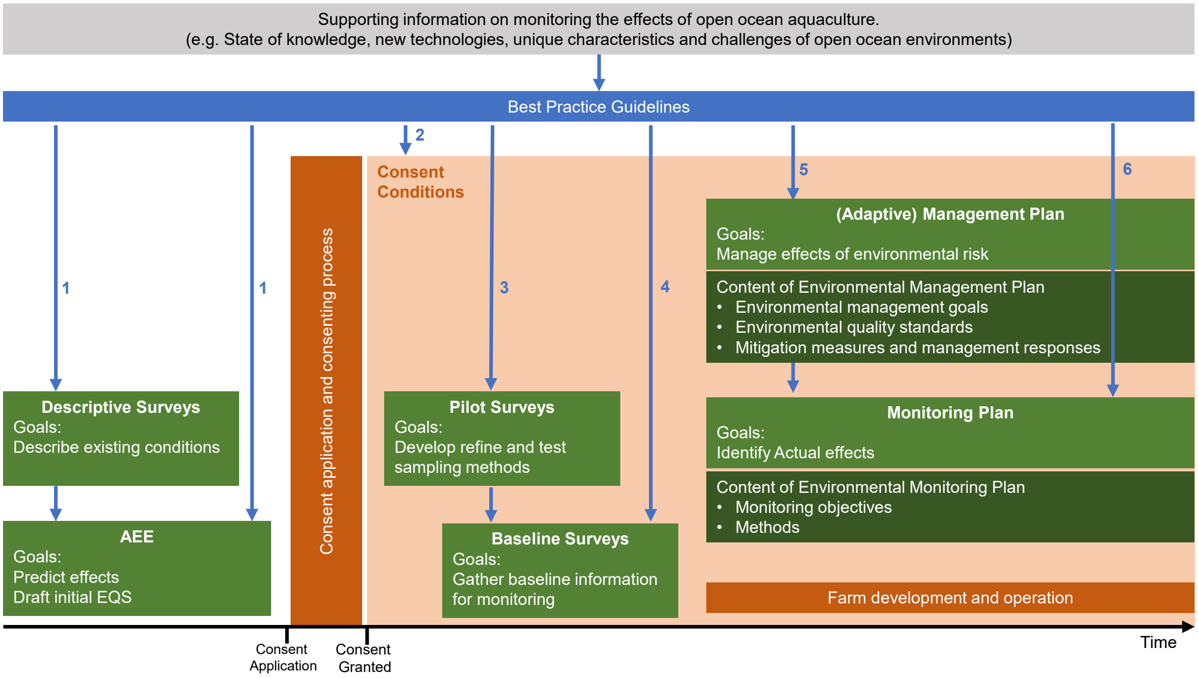
PREAMBLE

At Fisheries New Zealand we work to ensure New Zealand has world leading finfish farming practices which are environmentally, economically, socially, and culturally sustainable. The Government's Aquaculture Strategy has a goal of $3b in annual sales by 2035 and development of an open ocean finfish aquaculture industry is a key priority. Towards this goal of supporting sustainable open ocean finfish aquaculture, best practice guidelines for managing the effects of open ocean finfish farming on marine mammals, seabirds, water quality, and the benthic environment have been developed.

The guidelines for open ocean finfish farming ('the Guidelines') have been created to assist in the preparation of consent applications as well as inform consent conditions and decisions that will mitigate adverse environmental effects. The diagram below sets out the role of these guidelines in supporting OOA management under the Resource Management Act 1991.



1. The Guidelines inform the preparation of the assessment of environmental effects (AEEs) by: (a) describing potential effects and (b) outlining the specific information required for designing monitoring plans.
2. The Guidelines support the development of consent conditions by (a) recognising that monitoring and management aspects may need to be adapted over time and (b) promoting the development of Management Plans that enable the necessary flexibility in data collection and analysis.
3. The Guidelines provide assistance for planning pilot surveys to test sampling approaches.
4. The Guidelines inform the development of baseline surveys and monitoring programmes. Baseline information can be derived from descriptive surveys or existing reliable information sources.
5. The Guidelines support the development of management plans (EMAPs) by (a) Identifying the content required to link monitoring to effects management: and (b) Providing considerations for the development of environmental quality standards (EQSs).
6. The Guidelines support the development of monitoring plans (EMOPs) by providing technical guidance for monitoring effects and advice on how this can be implemented.

For seabirds and marine mammals, the Guidelines focus on mitigation of interactions through site selection, design, and operation of farm infrastructure. For benthic and water quality, mitigation of impacts is driven by the need for careful site selection and adaptive management practices that maintain acceptable environmental conditions within consented boundaries.

Because open ocean aquaculture is at an early development stage in New Zealand, carefully designed monitoring programmes will be required to assess the effectiveness of mitigation measures. The Guidelines therefore consider the range of monitoring options that may be appropriate in open ocean environments.

Effective monitoring will inform structural or operational improvements that deliver better environmental outcomes over time. The flexibility to evolve both monitoring and farming practices will be essential given the lack of New Zealand and international experience with open ocean aquaculture (OOA).

Guideline development process:

- Literature reviews were commissioned on the effects of open ocean aquaculture on **seabirds and marine mammals**. These provided a starting point for developing best practice guidelines. Following this, the Aquatic Environment Working Group and the Aquaculture Working Group (chaired by Fisheries New Zealand) provided peer review of the first and second drafts of this document.
- Fisheries New Zealand commissioned a report focused mainly on benthic effects of open ocean aquaculture overseas. This included several case studies from Norway regarding water quality effects. A technical working group was formed to discuss issues and develop guidelines for **benthic and water column effects** given the range of options available for monitoring and management of these effects.
- **Guidelines will be reviewed as needed** based on growing knowledge, so they can be improved. Considering this is the early days of open ocean aquaculture in New Zealand (and elsewhere), it is envisaged the initial review cycles will be relatively short with a backstop of 5 years from the development of the farms, or as required (e.g., if monitoring data and results are significantly different from anticipated).
- **Further research** is required into the effects of open ocean aquaculture upon the marine environment. In addition to known information gaps, research needs will be identified and documented as part of the guideline development and review process. These can then input to any ongoing or future Fisheries New Zealand research planning or prioritisation processes such as the proposed Aquaculture Innovation Plan and the proposed Aquaculture Strategy and Investment Roadmap, currently under development.

Factors to consider when reading and applying the Guidelines:

- Fisheries New Zealand is leading (with input from other government agencies) a process of considering options **for a future regulatory framework for open ocean aquaculture**. This future **open ocean aquaculture regulatory framework will take into consideration** concurrent regulatory change including the Biosecurity Act review, marine protected areas reform, and particularly the review of the Resource Management Act with a new Natural and Built Environment Act and Strategic Planning Act.
- While this regulatory framework may lead to recommended changes for any new farms proposed, **these guidelines are expected to remain relevant under any future regulatory framework.** The guideline review process can be used to manage required adaptations. Although we describe these guidelines within a Resource Management Act context, they are technical guidelines that will remain relevant under any regulatory framework.
- For the purpose of developing guidelines, **open ocean aquaculture is defined as “aquaculture outside of semi-enclosed bays and harbours or other sheltered locations around mainland New Zealand and larger offshore islands”.** This definition

includes the existing applications[1,2,3] at the time of publication of this document. Future applications are likely to be for areas that are at least as exposed as those in the existing applications. The process of considering different options for management frameworks for open ocean aquaculture may lead to a more precise definition of open ocean aquaculture. If so, that definition will be evaluated for adoption into these guidelines. Furthermore, although ‘open ocean’ is likely to contain areas considered non-dispersive (e.g., weak hydrodynamic regimes) (Bennett et al. 2020), we have assumed that only dispersive open ocean environments will be targeted for finfish farming and, thus, the guidance focuses on these environments.

Further guideline development will occur, for example:

- An approach to comprehensive biosecurity management for aquaculture is in the process of being developed. Officials advised Ministers in mid-2021 on recommendations to improve the management of aquaculture biosecurity, both marine (including open ocean) and land based.
- **Navigation issues** will continue to be considered under future regulatory frameworks. Fisheries New Zealand will continue to engage with Maritime New Zealand to ensure we meet our national and international obligations to provide for safe navigation.
- **Engineering guidelines** have been identified as a lower priority for government. This may be revised in the future.
- The development of guidelines for managing **shark or any other environmental interactions** with open ocean aquaculture has been identified as a lower priority for government. This may be revised in the future.

Mat Bartholomew

Director, Aquaculture, Fisheries New Zealand

[1] Blue Endeavour, New Zealand King Salmon.
[2] Hananui, Ngāi Tahu and Project South, Sanford.
[3] Project East, Sanford.

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List of acronyms

AEE	Assessment of environmental effects
CMA	Coastal marine area
DIN	Dissolved inorganic nitrogen
DOC	Department of Conservation
EEZ	Exclusive Economic Zone
EMAP	Environmental management plan
EMOP	Environmental monitoring plan
EMP	Collective term for EMAP/EMOP
EPA	Environmental Protection Agency
EQS	Environmental quality standard
MfE	Ministry for the Environment
NES-MA	National Environmental Standards for Marine Aquaculture
OOA	Open ocean aquaculture
RMA	Resource Management Act 1991
ROV	Remotely operated underwater vehicle
SOE	State of the environment
TN	Total nitrogen

EXECUTIVE SUMMARY

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Fisheries New Zealand works to ensure New Zealand has world-leading finfish farming practices which are environmentally, economically, socially, and culturally sustainable. The Government's Aquaculture Strategy has a goal of $3b in annual sales by 2035, and development of an open ocean finfish aquaculture industry is a key priority.

Within these guidelines, open ocean aquaculture (OOA) is defined as "aquaculture outside of semi-enclosed bays and harbours or other sheltered locations around mainland New Zealand and larger offshore islands". We have assumed that only dispersive open ocean environments will be targeted by industry, and the guidance is focused on these areas.

The purpose of these guidelines is to inform the sustainable development of open ocean finfish farming in New Zealand by providing robust and practical guidance for monitoring benthic and water quality effects. These guidelines are intended to inform the development of environmental monitoring plans (EMOPs) and related processes under the Resource Management Act 1991 (RMA) to ensure that monitoring is well integrated into the consent management framework. The EMOP defines the monitoring techniques, location, and frequency required to meet the environmental management objectives for the farm. These are generally defined within resource consent conditions and linked to environmental quality standards (EQSs) and farm management outcomes within an environmental management plan (EMAP). When establishing EQSs and the EMAP, care is required to enable monitoring plans to be adapted as our knowledge of monitoring OOA environments increases.

Associated RMA processes include the design of baseline surveys, assessments of environmental effects (AEE) preparation, writing or review of consent conditions, and design of adaptive management processes.

All activities within the marine environment produce some level of impact. The level of acceptable impact is defined based on a broad range of social, environmental, and economic value assessments made during the planning and consenting stages of farm development. Acceptable environmental effects are often defined as environmental quality standards, which the farming operation must not exceed. Monitoring is required to assess whether the predicted impact is above or below the permitted EQSs levels. The analysis of monitoring results allows assessment of compliance with agreed EQS limits and can inform the efficacy of mitigation measures to limit the effects of the farm on the environment.

Although there are few examples to draw on, the general perception is that open ocean aquaculture will have a reduced magnitude of environmental impacts compared with inshore farming. It is expected that offshore there will be increased dispersal and dilution, leading to reduced deposition and reduced accumulation of particulate waste adjacent to pens. The trade-off is a larger area of influence and potential effect.

Farming in offshore environments may therefore involve interactions with new habitats and species for which little is known about their tolerance to even low levels of organic deposition. This presents new challenges for developing appropriate EQSs and tools to monitor them.

These guidelines focus on the effects of open ocean finfish farming considered most likely to be ecologically significant and therefore require monitoring. The potential effects of aquaculture (especially finfish aquaculture) for the inshore marine environments can be broadly described as:

- The ecological effects of organic matter (faecal matter and uneaten food) from fish farms settling on the seabed.
- The effects of nutrients released from fish farms (primarily nitrogen and phosphorus) on phytoplankton production in the water column.
- The physical effects of shading and habitat change caused by the presence of farm structures. These are generally considered to be minor impacts in open ocean environments and are not discussed in detail in this guidance.

Uncertainty exists as to how these effects will manifest themselves in open ocean environments, and careful planning, monitoring, and, potentially, adaptive management will be required to avoid significant adverse effects.

Settlement of organic matter on the seabed can cause smothering of benthic fauna and alter the seabed biochemistry leading to changes in species abundance and diversity. At dispersive sites with complex bathymetry, hydrodynamics, and substrate types, there can be substantial patchiness in organic deposition. This may lead to highly localised effects associated with the accumulation of organic matter. The use of depositional models will help to predict the likely footprint of the farm and areas of potential accumulation and identify key sampling locations. Monitoring regimes must be designed to encompass the range of habitat types identified within that footprint. Although most farms will be sited to avoid impacts on important biogenic habitats, where there is uncertainty, it may be necessary to monitor biogenic habitats in the vicinity of the farm. Therefore, monitoring programmes will need to employ a range of sampling techniques that may include video surveys, grab samples, and acoustic surveys. Sampling programmes must also account for the logistical challenges of sampling in open ocean environments exposed to large waves and strong currents. Sampling frequency should align with peak discharge events and seasonal peaks in productivity.

The ecological effects of aquaculture on the water column are related to changes in nutrient levels resulting from the release of waste products. Increases in nutrient concentrations can lead to increases in marine phytoplankton growth rates and biomass and changes in species composition. Water sampling for chlorophyll *a* and nutrients are likely to be the key monitoring tools, although remote monitoring of plankton via satellite imagery remains a possibility for offshore farms with large dispersive footprints. Hydrodynamic modelling can describe the likely dispersion of nutrients from fish farms and inform appropriate monitoring locations. However, it is generally considered that fish farm nutrient inputs are not large enough to significantly affect phytoplankton populations in open ocean environments.

A glossary of terms used in the Guidelines is provided before the Appendices.

PART A: Introduction and consent-related resource management processes

1. INTRODUCTION

1.1 Background

Open ocean aquaculture[4] is an emerging industry in New Zealand. Fisheries New Zealand are committed to ensuring New Zealand has world leading finfish farming practices, which are environmentally, economically, socially, and culturally sustainable. The development of best practice guidelines for monitoring effects on the benthic environment and water quality ('the Guidelines') is a key contribution towards this goal[5].

It is important to note that there is no single definition of the term 'open ocean' (or 'offshore' or 'exposed' environments). Instead, there are different definitions of open ocean aquaculture, which can lead to apparent inconsistencies in expectations and published statements on the likely nature and magnitude of effects from open ocean aquaculture. For example, the National Environmental Standards for Marine Aquaculture define offshore marine farms largely in terms of distance from land.[6] Definitions of open ocean environments also differ in the scientific literature, including in the literature used to inform the Guidelines. Bennett et al. (2020) define open ocean aquaculture as "that which occurs in areas requiring an upgrade in structural technology or change in farming methods (compared to current inshore farming practices) to cope with open-ocean (i.e., exposed) conditions". Keeley (2020) highlights that the term 'open ocean aquaculture' is somewhat ambiguous and recommends that sites are viewed in terms of their physical dispersive properties, as determined primarily by currents, wave exposure, and site depth (Figure 1).

For the purpose of the Guidelines, open ocean environments are defined as areas outside of semi-enclosed bays and harbours or other sheltered or semi-sheltered locations around mainland New Zealand and larger offshore islands. We assume that open ocean environments targeted by industry for open ocean aquaculture will be dispersive in nature and the Guidelines are designed for those environments. Open ocean aquaculture is therefore defined by the dispersive nature of the environment, not by distance from land. This is appropriate for the development of monitoring guidance, as the suitability of monitoring approaches is to a large extent related to the dispersive characteristics of the environment. Depending on hydrodynamic conditions, open ocean aquaculture as defined in the Guidelines may be relatively close to shore or have impacts on nearshore environments.

Because industry is likely to target dispersive open ocean areas, the general perception is that finfish farming in open ocean environments may have reduced magnitude of environmental impacts compared to inshore farming through increased dispersal and dilution, reduced deposition, and reduced accumulation of particulate waste adjacent to pens (e.g., Environmental Protection Agency Tasmania 2019). The trade-off is a larger farm footprint.

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[4] Alternative terms used to refer to open ocean aquaculture include offshore aquaculture or exposed aquaculture.
[5] Other guidelines are being developed to inform the monitoring of other effect types (see preamble). We note that the focus of the guidelines differs. While the benthic and water quality guidelines focus on best practice for monitoring, those developed for marine mammals and seabirds providing operational best practice for minimising effects.
[6] Resource Management (National Environmental Standards for Marine Aquaculture) Regulations 2020, section 4.

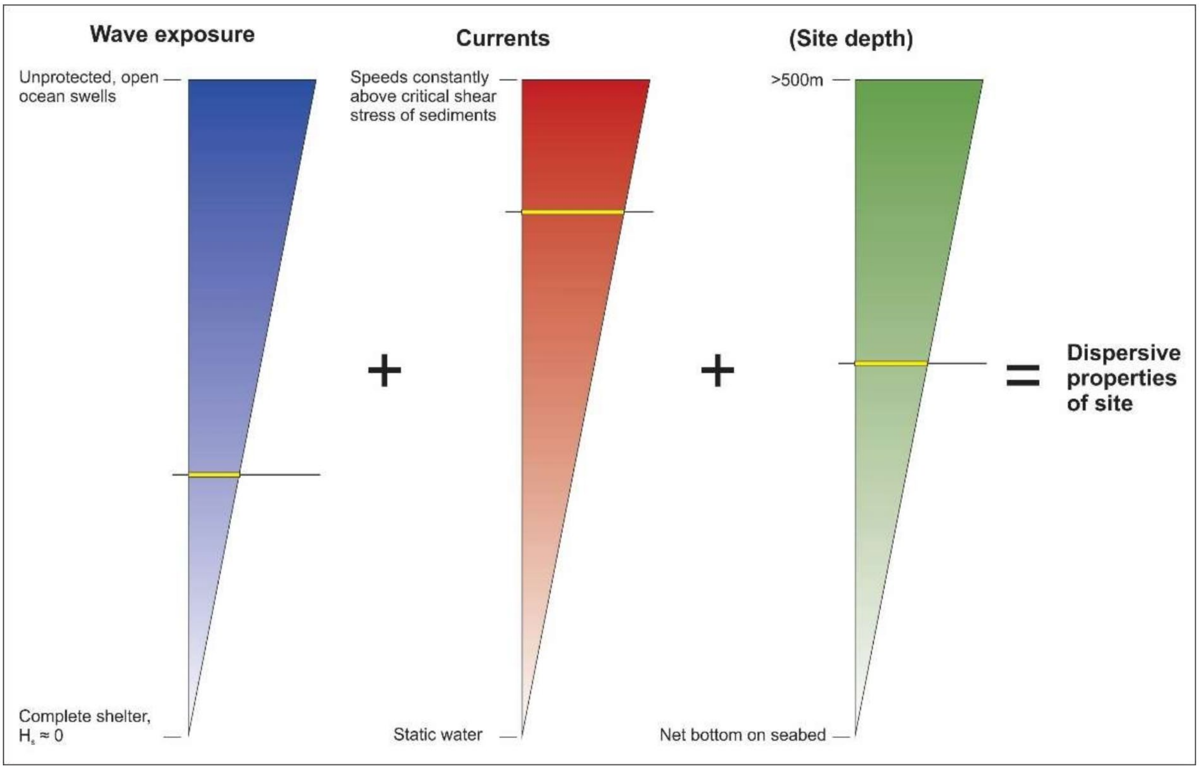

Figure 1: Wave exposure, currents, and water depth at the site, representing the main physical and hydrodynamic properties contributing to the waste dispersive properties of a theoretical marine aquaculture site. H_s = significant wave height. Source: Keeley (2020).

The effects of finfish farming on soft sediment benthic environments and water quality in New Zealand's inshore environment are relatively well understood in the Marlborough and Southland regions (e.g., Ministry for Primary Industries 2013). In New Zealand we do not yet have any experience with monitoring benthic and water quality effects of finfish farming in open ocean settings, and there are relatively few examples to draw on from overseas. The characteristics of open ocean environments may also create new logistical challenges for monitoring. For example, farming in these new open ocean environments will likely encounter new habitats and species for which little is known about their tolerance for organic waste (Keeley 2020). This uncertainty has been considered in the preparation of the Guidelines and recommended monitoring approaches.

The guidance provided focuses on potential effects of open ocean finfish farming, based on the state of knowledge at the time of writing the Guidelines; specifically, those considered most likely to require monitoring because of the potential for ecologically significant consequences. A relatively short initial (compared to the length of the consent) review period of the Guidelines will provide for the timely incorporation of new technical and operational knowledge derived from New Zealand experience with open ocean finfish farms.

1.2 Purpose, use, and scope of the Guidelines

1.2.1 Purpose

The purpose of the Guidelines is to inform the sustainable development of open ocean finfish farming in New Zealand by providing robust and practical guidance for monitoring benthic and water quality effects.

1.2.2 Use of the Guidelines

The primary use of the Guidelines is to inform the development of environmental monitoring plans (EMOPs). However, it is envisaged that the Guidelines are also considered in related processes under the Resource Management Act 1991 (RMA). These processes include design of baseline surveys, assessments of environmental effects (AEE), preparation, writing, or review of consent conditions, and design of adaptive management processes to ensure that monitoring is well integrated in the consent management framework. This is described in more detail in section 3.

In general, the Guidelines are aimed at a broad audience by providing information of relevance to everybody involved in resource management related to open ocean aquaculture. However, the technical guidance provided in Part B of the Guidelines is primarily targeted at the development and implementation of site-specific finfish farm EMOPs, particularly for councils and applicants.

1.2.3 Scope

The Guidelines cover general aspects for monitoring the benthic environment and water quality with respect to effects of all species of finfish that might be farmed in open ocean environments of New Zealand. Although the Guidelines have been tailored to the RMA framework, which applies within the 12 nautical mile limit where the council is the regulator, the guidance is generally applicable for sites beyond that limit if the need arises (noting that the Environmental Protection Authority (EPA) has decision making responsibilities in the exclusive economic zone, EEZ, i.e., beyond the 12 nautical mile limit).

Out of scope:

- The Guidelines do not provide guidance for monitoring needs that might arise from social or cultural concerns about benthic or water quality effects.
- The Guidelines do not apply to mobile aquaculture structures, i.e., finfish farm structures that are not fixed in position during a fish growth cycle (however, they do apply to farming approaches in which pens are moved within a consented area).
- The Guidelines do not specifically consider multi-trophic aquaculture (MTA) but apply to the finfish component of an MTA system.
- The Guidelines do not provide principles or recommendations for the locations of farms.
- The Guidelines do not determine whether monitoring is required because this is an inherently application-specific decision. Instead, the Guidelines provide technical guidance on how monitoring can be done, should it be deemed necessary.
- The Guidelines do not cover options for management/mitigation of effects (though monitoring results will inform decisions on these matters and the alignment of monitoring and management of effects is discussed).

1.3 Guideline development process

The Guidelines were developed by a working group involving representatives of Fisheries New Zealand, the Department of Conservation, the Coastal Special Interest Group (represented by coastal scientists from Marlborough District Council and Environment Southland), Te Ohu Kaimoana, Sanford, New Zealand King Salmon, Aquaculture New Zealand, and the Environmental Protection Agency as well as scientific experts and a community representative.

The Guidelines build on previous work in New Zealand that characterised the effects of aquaculture (Ministry for Primary Industries 2013) and developed best management practices for salmon farming in the Marlborough Sounds in terms of benthic and water quality impacts (Elvines et al. 2019, Keeley et al. 2019a) and more recent reviews of international lessons and assessments (Bennett et al. 2020 and

Keeley 2020; summarised in Appendix 1). The review of Loh et al. (2019) was also influential in terms of monitoring design.[7] The Guidelines represent the best information available at the time. The process followed by the working group to identify potential effects of open ocean finfish farming for the purpose of structuring the technical guidance is described in Appendix 2.

This first version of the Guidelines aimed to develop technical monitoring guidance for the effects that were considered most likely to require monitoring (Appendix 2). This narrowed approach was necessary to develop the Guidelines within the available timeframe. The timeline of the guideline development was such that the Guidelines can inform the resource consenting process for the current (as of the date of publication of this document) open ocean finfish farm applications in Marlborough, Southland, and Otago.

1.4 Review of the Guidelines

The Guidelines will be reviewed periodically to allow adaptation to new scientific, technological, or operational information and to make improvements in response to experience with their implementation. In addition to a periodic review cycle, there will be opportunities to review the Guidelines outside the review cycle, if necessary.

This document will be reviewed either:

a. approximately three years after the first open ocean aquaculture farm is operational;

b. if significant new scientific information is available that may require changes to the current guidelines (as judged by Fisheries New Zealand); or

c. at another time but no later than five years following publication of these guidelines.

Reviews will be at the discretion of and led by Fisheries New Zealand. They will include reviews of available information on monitoring and environmental effects from operational open ocean finfish farms in New Zealand and also consider wider relevant literature from New Zealand and overseas.

For any questions or comments on these Guidelines, including the review process, please contact the Aquaculture Strategy and Development team in Fisheries New Zealand (aquaculture@mpi.govt.nz).

2. PRINCIPLES

The Guidelines are not intended to be prescriptive, but rather to support processes related to benthic and water quality monitoring of open ocean finfish farms, such as designing EMOPs, conducting descriptive and baseline surveys, embedding monitoring requirements in consent conditions, or designing (adaptive)[8] management frameworks. To support this intent, the following principles have been developed for the Guidelines and the monitoring approaches they aim to support:

- The Guidelines promote the maintenance of healthy benthic and water column environments without imposing unwarranted financial or practical burden on marine farmers.

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[7] Although a systematic review of international literature was not performed as part of the guideline development process, the knowledge of literature and the experience held by the working group was considered sufficient scientific basis for the guidance provided. The extensive peer review process of the Guidelines provided scientific and industry feedback that augmented the robustness of the guidance.

[8] The Guidelines recognise that, although it is likely that many open ocean finfish farms will be managed under an adaptive management approach, this is not an inherent requirement and may not be the case for all open ocean finfish farms. The Guidelines apply to any management framework, but we acknowledge that some aspects were influenced by our thinking about the role of monitoring within an adaptive management framework.

- The Guidelines encourage national consistency while acknowledging that consent-specific circumstances may require deviations from recommended or consistent methods.
- The Guidelines promote monitoring that is fit for purpose to provide appropriate information on actual effects of consented open ocean finfish farms. By doing so, monitoring can inform operational and resource management decision-making aimed at managing effects and encouraging desired environmental outcomes to be achieved.
- The Guidelines recognise the characteristics and challenges for monitoring impacts of fish farms, both operationally and scientifically, in open ocean environments.
- The Guidelines support the collection of more data on natural environmental variability and long-term trends to develop or refine EQSs in the future.

3. RELEVANT CONSENT-RELATED RESOURCE MANAGEMENT PROCESSES

3.1 Alignment among key components and documents of the consent process

Under the RMA, managing a consented activity that has potentially significant and/or uncertain effects frequently requires monitoring. It is therefore important to consider where, within the relevant resource management processes, the Guidelines will be used, and how the monitoring and related science information is linked with these processes. An illustration of these linkages is given in Figure 2. The relevant consent-related resource management processes here include the preparation of consent applications, consideration of applications by decision makers, development of consent conditions, and ongoing demonstration of compliance with these conditions.

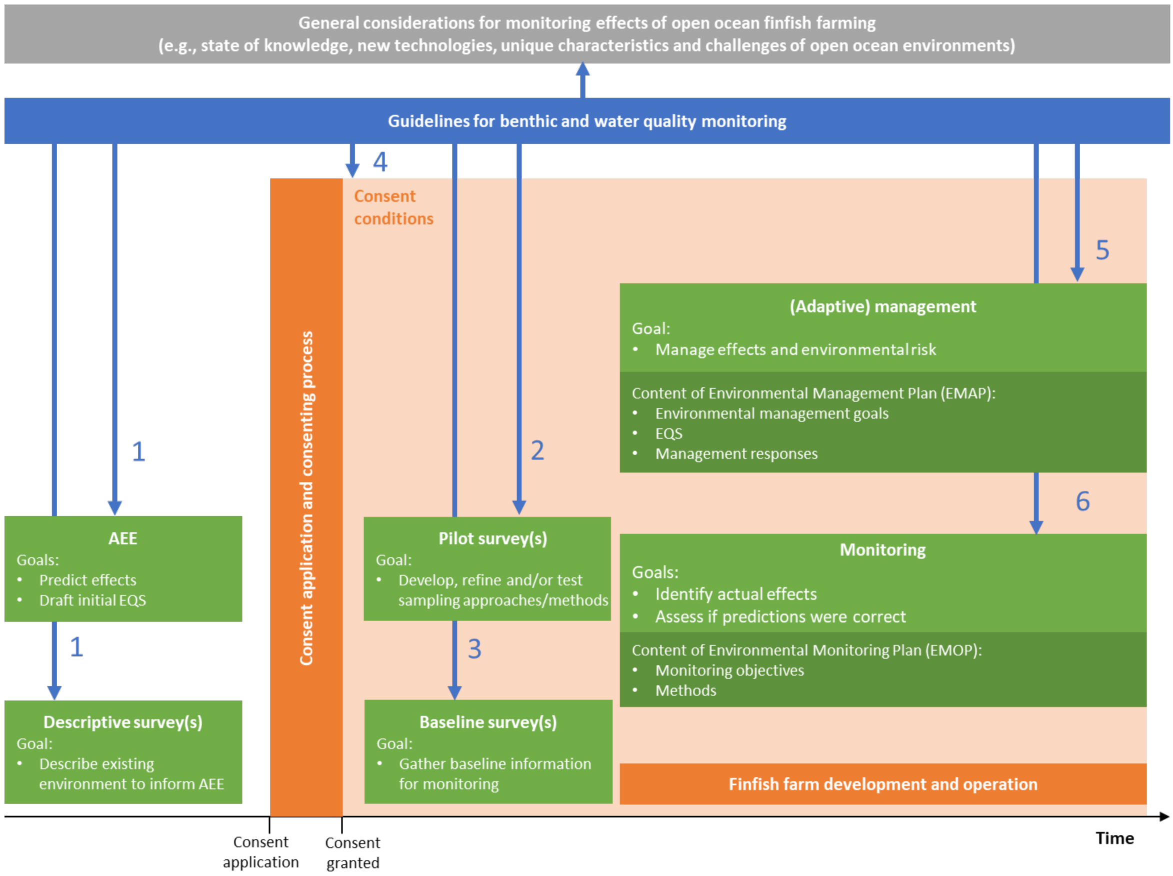
In addition, adaptive management is frequently used to manage the development of finfish aquaculture and it is anticipated that this will continue to be the case for open ocean finfish aquaculture. Accordingly, these open ocean aquaculture Guidelines allow for adaptive management frameworks to be implemented.

This section provides an overview of key components and documents of the consenting process (AEEs, consent conditions, EMAPs, and EMOPs) and emphasises alignment among these components is required to ensure that monitoring is well-integrated and relevant for application-specific circumstances.

Assessment of environmental effects

The preparation of an AEE is an obligatory resource management process that takes place before a consent application is submitted. AEEs assess the potential effects of the proposed activity on the environment, and identify any risks and uncertainties associated with those effects. If the scale and significance of the effects is such that monitoring will likely be required, an AEE may describe a basic strategy for how the effects are to be monitored.

AEEs are usually informed or supported by field observations to determine the characteristics of the existing natural environment. In the Guidelines, surveys conducted to inform AEEs are referred to as descriptive surveys. The purpose of a descriptive survey is to describe the existing natural environment to the extent required for the preparation of the AEE if this information does not already exist.



1 The Guidelines inform the preparation of AEEs and descriptive surveys by: (a) describing potential effects with special consideration to open ocean aquaculture (sections 7.2 and 8.2); and (b) outlining the application-specific information required for monitoring design, which may be gathered as part of the AEE and descriptive surveys (sections 7.1 and 8.1).

2 The Guidelines provide considerations for planning pilot surveys for developing, refining and/or testing sampling approaches, if deemed necessary, to ensure they are fit for purpose for baseline surveying and ongoing monitoring (section 6.3). Pilot surveys may be carried out before farm development or once development has commenced.

3 The Guidelines provide considerations for planning baseline surveys to inform monitoring programmes (section 6.4). In some instances, baseline data may be derived from descriptive surveys or other existing information if sufficiently robust and detailed.

4 The Guidelines support the development of consent conditions by: (a) recognising that aspects of monitoring and management of benthic and water column effects will likely require adaptation over time; (b) promoting two key documents that assist in achieving this adaptation (environmental management plans [EMAPs] and environmental monitoring plans [EMOPs]); and (c) providing guidance on how to align consent conditions, EMAPs and EMOPs (section 3.1).

5 The Guidelines support the development of EMAPs by: (a) the fundamental content to include in an EMAP to link monitoring to effects management; and (b) providing considerations for development of EQSs (section 3.1).

6 The Guidelines support the development of EMOPs by providing technical guidance for monitoring benthic and water quality effects (Part B), and advice on how this can be implemented in EMOPs (Part C).

Figure 2: Illustration of how the Guidelines (blue) can be used to inform consent-related resource management processes, including adaptive management. It is acknowledged that this description simplifies these processes, and that individual councils and operators may use different terminology.

Consent conditions

Consents are typically granted subject to conditions. If monitoring of potential adverse effects of the consented activity is required, conditions should describe the aspects of the environment (and the effects) that require monitoring and the environmental management goals to be assessed through monitoring. Consent conditions may also include EQSs, or provisional EQSs. The specificity of meaningful EQSs that can be developed at the time of consenting depends on the available level of knowledge and information (see discussion on EQSs in section 3.2). In some instances, detailed monitoring requirements are stipulated by conditions. However, this highly prescribed approach can constrain the ability for monitoring to adapt over time as information and technology evolve. Conditions may also include the requirement for further surveys to be conducted before a finfish farm is developed (e.g., baseline surveys).

Environmental management plans and environmental monitoring plans

Detailed management and monitoring requirements (e.g., how monitoring is to be undertaken, interpreted, and reported, and any adaptive management responses) are commonly captured in standalone documentation (EMAPs and EMOPs) that are tied to the consents by conditions requiring their preparation, content and objectives, certification by the consent authority, and regular reporting against their requirements.

Including the specific details of monitoring in EMAPs and EMOPs is an approach recommended for open ocean finfish aquaculture. For this approach to be effective, EMAPs and EMOPs need to be anchored in consent conditions to ensure the overall effects monitoring and management approach provides the level of certainty and enforceability required for granting consent.

If draft EMAPs and EMOPs are provided with the consent application, they may inform development of consent conditions, maximising alignment between these key documents. However, EMAPs and EMOPS may also be developed after the consent has been granted.

If potential effects of the consented activity necessitate certain environmental management practices to be stipulated, EMAPs should set out how these will be managed once the activity commences. EMAPs should include:

- environmental management goals (which may also be mirrored in the consent conditions) that are specific to the environmental risks that have been identified;
- the EQSs against which those goals will be assessed;
- a description of the management response framework to be followed in the event that EQSs are exceeded, and environmental management goals are not met. This does not necessarily need to include specific management actions but does need to include a process for working through these non-achievements and exceedances with the council. Specific management actions may be provided in separate documents and cross-referenced within the EMAP.
- If a consent is managed under an adaptive management approach, the EMAP should include all the necessary additional components required for effective adaptive management (as described in the next section).

EMOPs should describe the environmental monitoring required under the resource consent. As well as detailed technical information for carrying out environmental monitoring, EMOPs should include monitoring objectives aimed at measuring the relevant effects of the consented activity and assessing whether environmental management goals and EQS set in EMAPs are met (if relevant). More information on EMOPs is provided in Part C.

The EMAP is fundamental to inform the development of monitoring objectives that will be written into the EMOP and will also feed into other elements of the EMOP, such as reporting of monitoring results. EMAPs and EMOPs should thus be closely aligned so that monitoring has a clear purpose and is well connected to (potential) adverse effects of the consented activity or environmental management goals. This will ensure effective management responses if the EQSs or environmental management goals are not met. In some instances, EMAPs and EMOPs might be combined into one document.

EMOPs and EMAPs should be reviewed periodically to ensure they reflect current knowledge and advances in technology. This is deemed particularly important for open ocean aquaculture given the probability that technologies associated with, and knowledge concerning, open ocean finfish farming will evolve rapidly as the industry develops. Consent conditions should provide for the review of EMAPs and EMOPs with a review process built in and certification from the council.

3.2 Enabling effective adaptive management

3.2.1 General aspects of adaptive management

Adaptive management is commonly used to support the management of aquaculture activities where effects are uncertain. Adaptive management allows decisions to be made despite uncertainty. Adaptive management frameworks put emphasis on learning, largely through improved knowledge derived from monitoring (or research), and the subsequent adaptation of management based on lessons learnt (e.g., Giles 2019).

If an open ocean finfish farm is managed under an adaptive management approach, the EMAP needs to include pre-determined decision points, a process for assessing predicted against achieved outcomes (including a process for comparing monitoring results to EQSs and predicted effects), and management responses to be taken based on the results of monitoring and other outcomes. For example, the maximum allowable feed input to fish pens may initially be limited (as a practical proxy for initial limits on benthic effects) until a specific time at which benthic monitoring is conducted to assess actual effects against those predicted by the AEE (i.e., staged development, see section 3.2.2). Based on that example, there may be three pre-determined management responses based on monitoring results:

1. If actual effects are of the predicted nature and lower than, or similar to, those predicted at the initial feed input, feed discharge may be increased to a higher pre-determined level.
2. If actual effects are higher than those predicted at the initial feed input but are of the nature and within the pre-defined range of effects considered acceptable. If the initial effects predictions could be improved using the monitoring results, then this approach may help to reduce uncertainty for predicted effects at higher feed inputs. This could allow feed increases to proceed as planned. Alternatively, feed input may be increased by a lesser amount than the initially pre-determined level based on the improved prediction.
3. If actual effects are unexpected, potentially unacceptable, and do not reflect the nature of predicted effects (i.e., the environmental response differed substantively from that predicted), the level of understanding of the effects is insufficient. In this example, further scientific investigative work is required.

A core requirement of adaptive management is that, once a resource consent has been granted, it requires periodic adjustments to reflect new knowledge. Adjustments may include changes in management practices, modification of EQSs, or changes in monitoring plans. The process for making such adjustments differs depending on how the initial requirements were stipulated. For example, under the RMA it is typically easier to make changes to EMAPs or EMOPs than to consent conditions. However, consent conditions must provide certainty and assurance about the management of effects and there are limits on the extent to which the specifics of management or monitoring practices can be specified by EMAPs or EMOPs instead of being specified by consent conditions. Certainty is also expected by operators and other stakeholders.

The legal requirements for alignment of consent conditions and EMAPs and EMOPs as stated by the New Zealand courts have recently been examined by Giles & Barton (2020). A summary of these requirements is shown in Box 1 and a more comprehensive overview of the findings of Giles & Barton

(2020) is shown in Appendix 3. These findings provide helpful considerations for developing an effects monitoring and management approach for a specific application.

Initially, it is likely that considerable flexibility will be required for managing effects of open ocean finfish farming, and therefore it will be important to ensure that effective adaptive management is provided. It is expected that as our experience with open ocean aquaculture improves, benthic and water quality effects can be predicted, monitored, and managed with more certainty. As a result, the level of adaptability in the consenting framework for open ocean aquaculture may reduce over time. The systematic collection of robust monitoring data in these early stages, however, is paramount to ensuring that our experience with open ocean aquaculture over the coming years is accompanied by increased understanding and certainty of its effects.

3.2.2 Staged development approaches

Staged development with increasing development intensity in progressive stages is a specific application of adaptive management that is frequently used for aquaculture consents with potentially large but uncertain effects. It allows for pre-defined increments in the activity, such as addition of pens within the consented area or increased feed input to existing pens, if certain criteria have been met. The assessment of whether those criteria have been met is typically based on comparing monitoring results to performance measures, such as EQSs.

Planned staged development can take two general approaches (Figure 3), or a hybrid combination of both. The first is increasing development intensity in progressive stages; for example, initially a lower number of pens within blocks than what has been consented as the full intensity development[9].

The second staged development approach for open ocean finfish farming could involve full intensity development of an initial stage, but only in a limited number of blocks until a certain point. This approach may be more conducive to reducing uncertainties and information gaps associated with benthic effects, and to refining benthic monitoring programmes. This is because it allows the full spectrum of benthic effects to manifest within a part of the consented space, and thus be monitored.

Any refinements that need to be made to initial monitoring indicators and EQSs could then be implemented across additional blocks in the wider area with greater certainty. This approach may present a greater risk to the seabed affected by those blocks which are allowed to proceed rapidly to full intensity development and this risk needs to be managed. For example, blocks would need to be carefully chosen to manage risk of adverse effects on nearby significant ecological areas. It may also be less effective in increasing certainty around cumulative effects. Where the effects of open ocean finfish farming are expected to be of low to moderate environmental risk, this approach may be preferred because it can provide new knowledge on environmental effects of open ocean finfish farming in the shortest possible time, which may be of wider benefit for the industry, councils, and research.

Which approach is most appropriate for a specific application, if required at all, will depend on a range of factors, including the environmental concerns or other reasons requiring staged development.

[9] Full intensity development is typically conditional to criteria relating to environmental performance being met.

BOX 1. Anchoring EMAPs and EMOPs in consent conditions: legal requirements

The relationship between resource consent conditions and environmental management/monitoring plans (EMPs) has been well traversed by the New Zealand courts. See Appendix 3 for an overview of direction provided through court decisions and Giles & Barton (2020) for a detailed review.

In summary, based on the findings of the courts, the relationship between resource consent conditions and EMPs can be described as follows:

- Consent conditions must specify clear environmental outcomes and objectives for EMPs to apply.
- EQSs are to be specified in the conditions of consent, not in EMPs. Where it is difficult to formulate precise well-defined standards (e.g., numerical values), clear, practicable, effective, and enforceable standards should be specified in conditions of consent to protect the environment from adverse effects.
- EMPs should stipulate how objectives/EQSs specified in consent conditions will be met, including methods to be used for assessments.
- If the level of detail required in an EMP cannot be provided at the time of decision making on granting consent, as a minimum the applicant must provide evidence demonstrating in broad terms how the objectives of the EMP are to be achieved.
- Conditions of consents must specify implementation requirements for EMPs to ensure enforceability.
- It is not lawful to delegate substantive decisions (i.e., those which have a bearing on whether a consent should be granted) to EMPs or other processes conducted after the decision on granting consent has been made.
- Critical information gaps about the likely environmental effects at the time of the decision cannot be filled by post-decision information-gathering (e.g., through an EMP).
- In situations where knowledge is incomplete and expected to advance after granting consent, consent conditions may delegate certification to a later time but not decision-making that the consent authority should have exercised itself at the time of granting consent. For example, consent conditions may allow a delegated official to confirm that the EMP prepared after consent has been granted is the most appropriate means to achieve the objectives stated in the conditions but not allow a delegated official to make decisions on acceptable levels of EQSs.
- Care must be taken that consent conditions do not conceal the fact that what is being delegated is the power to certify a matter which is an essential element of the decision which should be made at the time of granting consent.

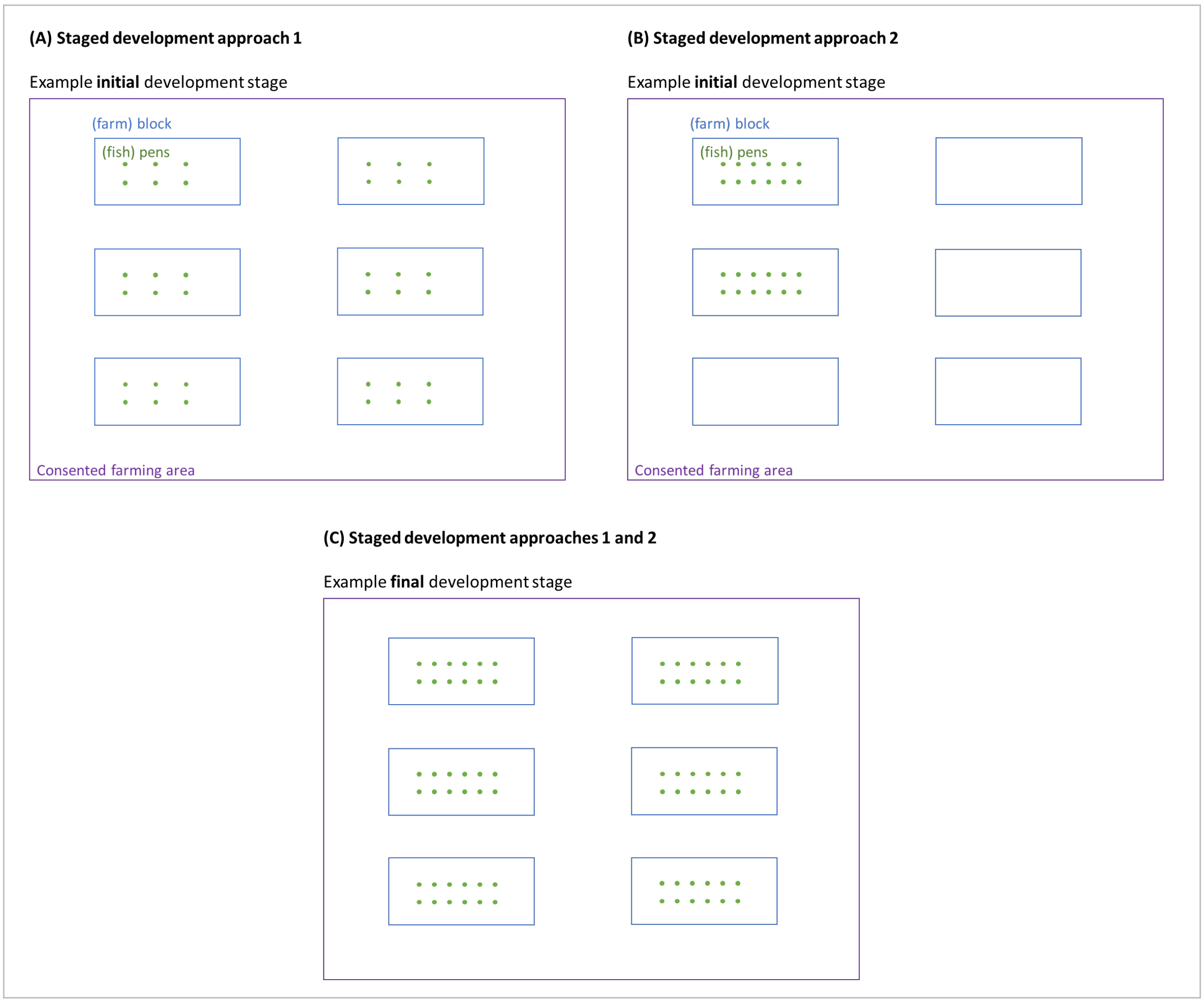


Figure 3: Illustration of the farm layout terminology used in the Guidelines and hypothetical examples of the two development approaches described in the Guidelines. A group of (fish) pens (including associated anchoring structures) is arranged in a (farm) block. The purple outline represents the area consented for finfish farming. These layouts are hypothetical examples only, used to clarify terminology used in the Guidelines. Finfish farming may take place in other configurations. The configuration in (A) represents the initial development stage under a staged approach of increasing development intensity in progressive stages. The configuration in (B) represents the initial development stage under a staged approach of initial full intensity development in a limited number of blocks. Both approaches have the same final development stage shown in (C).

4. POTENTIAL EFFECTS OF OPEN OCEAN FINFISH FARMING

The technical benthic and water quality guidance provided in sections 7 and 8 is structured around potential effects that may occur in open ocean finfish farming (Table 1). As noted earlier, although potential effects and associated monitoring are described in the Guidelines, this does not mean that monitoring for all potential effects will be necessary for all open ocean finfish farms.

The process followed by the working group to identify potential effects of open ocean finfish farming is described in Appendix 2. The potential effects listed in Table 1 are largely similar to the potential effects from inshore finfish farming.

Table 1: Potential effects of open ocean finfish farming. Column three indicates potential effects that would most likely require monitoring in an open ocean setting and to which the technical guidance is limited.

Environment	Potential effects	Monitoring guidance provided?
Benthic	Organic enrichment and associated changes in the benthic environment due to biodeposition or biofouling drop-off	Yes
	Smothering of benthic habitats and communities by biodeposits or biofouling drop-off	Yes
	Contamination of sediments due to the deposition of therapeutants, feed additives, antifoulants, or other potentially harmful substances used in finfish farm operations	Yes
	Changes in physical habitat due to biodeposition, biofouling drop-off, disturbance from mooring systems, or scouring due to the presence of the farm structure	No
	Shading of the seabed by farm structures	No
Water column	Increased concentrations of dissolved nutrients and subsequent enhanced phytoplankton biomass and growth	Yes
	Reduced concentrations of bottom water dissolved oxygen	Yes
	Reduced concentrations of surface water dissolved oxygen	No
	Increased concentrations of other contaminants such as therapeutants	Yes
	Effects from the presence of the farm upon waves, currents, water column stratification, and other hydrodynamic properties	No
	Effects from shading on water quality or water column biology	No
	Effects of artificial lighting on water quality or water column biology	No

Given the constraints for preparing the Guidelines, the working group developed guidance for the potential effects that would most likely require monitoring. No guidance is provided for:

- Changes in physical habitat due to biodeposition, biofouling drop-off, disturbance from mooring systems, or scouring due to the presence of the farm structure;
- Shading of the seabed by farm structures;
- Reduced concentrations of surface water dissolved oxygen;
- Effects from the presence of the farm upon waves, currents, water column stratification, and other hydrodynamic properties;
- Effects from shading on water quality or water column biology; and
- Effects of artificial lighting on water quality or water column biology.

Potential effects for which monitoring guidance is provided are described in more detail in the respective sections (section 7 for benthic effects and section 8 for water quality effects). Monitoring undertaken for benthic or water quality effects may also capture changes resulting from other effect types.

Potential effects may be revised, added, or removed in the future as our experience with open ocean finfish farming improves. As noted earlier, although potential effects and associated monitoring are described in the Guidelines, this does not mean monitoring will be necessary for all open ocean finfish farms.

PART B: Technical guidance

5. SCOPE AND STRUCTURE OF THE TECHNICAL GUIDANCE

The scope of the technical guidance is to provide design considerations for the collection of scientific information for monitoring effects of open ocean finfish farming. The guidance does not provide advice on what monitoring should be undertaken at a particular fish farm site.

Section 6 provides general guidance for monitoring, baseline surveys, use of models, and other related matters. This guidance is largely applicable to both benthic and water quality monitoring.

The technical guidance in section 7 (benthic monitoring) and section 8 (water quality monitoring) is structured as follows:

1. **Application-specific information requirements** for monitoring design.
2. **Guidance for design of monitoring** (structured by potential effects) relating to potential effects, covering:
    a. Sampling methods (the indicators to sample, and the sample collection techniques)
    b. Sampling effort (e.g., sampling locations, timing, and frequency; sample replication).
3. **Other considerations relevant to monitoring design.**

6. GENERAL GUIDANCE FOR MONITORING, BASELINE SURVEYS, USE OF MODELS, AND OTHER RELATED MATTERS

6.1 Addressing uncertainty and associated risk by varying monitoring effort

Open ocean aquaculture is only just beginning to emerge in New Zealand and monitoring will therefore be conducted in the context of uncertainties related to operational aspects, environmental response, and scientific methods. One way of dealing with the initial uncertainty and associated risk is to design monitoring so that monitoring effort relates to the level of certainty. Effort could be initially higher but decreases over time with increasing certainty.

As a general rule of thumb, greater uncertainty and associated risk requires greater monitoring effort. For example, a consent holder may be required to conduct monitoring to characterise the intensity and spatial extent of effects and confirm that effects are as predicted for the early stages of open ocean farming to address any uncertainty associated with the predictions. As monitoring results become available and are used to confirm or, if necessary, improve predictions, monitoring effort may be reduced (also see discussion in section 3.2 on adaptive management). Monitoring of effects against established EQSs may require fewer sampling stations, may only target specific spatial zones, or may require fewer biological indicators compared with monitoring designed to characterise effects.

It is important to recognise that information gaps and uncertainties can or may need to be answered by research projects (see section 12). Varying effort monitoring approaches should therefore allow for the incorporation of any new information, not just information derived from monitoring.
Scientific uncertainty and associated risk may arise in relation to the:

- Probability that the unwanted change can be corrected after it has been recognised.
- Timescale of reversibility.
- Effort required to achieve ecological recovery.

- Spatial scale over which the unwanted change might become evident.
- Far-field effects (noting that far-field effects may be of low/negligible ecological relevance).
- Ecological significance of the environmental resource that may be affected.
- Level of scientific understanding of environmental response to the stressor arising from the consented activity, e.g., response of key habitats or species to the discharge of finfish farm waste.
- The magnitude of the predicted environmental change resulting from finfish farming compared with the natural variability of relevant environmental characteristics.
- Cumulative effects on habitats and species including the level of stress experienced by habitats and species before finfish farm development. If the pre-development stress is high, there is a greater probability that even a small incremental stress will result in ecologically unacceptable adverse impacts. Equally, if a habitat or a species is naturally stressed, it may cope better with some additional stress.

New open ocean farm sites are expected to have limited robust site-specific environmental data and available data may be on a relatively coarse scale. Such data limitations are more likely in open ocean environments than in nearshore farming areas because of the general lack of SOE monitoring or pre-existing research in open ocean environments that are likely to be considered for aquaculture development.

Although effects are generally anticipated to be less intense in dispersive open ocean environments, there is some uncertainty regarding the response of some communities to organic enrichment. For example, communities in systems with lower natural rates of organic input may have a lower capacity to assimilate (or process) the inputs from finfish farms because these communities are not always pre-adapted to receiving large inputs of nutrient and organic matter in the way that more productive inshore areas can be (Holmer 2010, MacLeod et al. 2007) (see Appendix 1). However, such communities may adapt over time. In soft sediment habitats, macrofauna species and microbes adapted for more enriched conditions are expected to colonise enriched sediments, facilitating organic matter decomposition. It is possible that after this lag period the open ocean soft sediment environment may have an equal or possibly even greater assimilative capacity than inshore environments, but, in some cases, this could be at the loss of taxa performing other important functions, such as sediment stabilisation (MacLeod et al. 2007). The uncertainty inherent in not having existing data demonstrating the response of open ocean environments to organic enrichment can be readily addressed through monitoring. Once these processes are better understood, monitoring effort can likely be reduced. In general, we anticipate that the effects of open ocean finfish farming will generate less intense (but more extensive) environmental footprints, because we anticipate that industry will target dispersive areas for farming.

6.2 Tiered monitoring approaches

Management or monitoring frameworks with different levels of monitoring effort are appropriate when effects have been well characterised. Employing different monitoring levels can allow for monitoring resources to be used wisely and targeted to times or areas of most concern. For example, the effects of farming on soft sediments in the Marlborough Sounds are well characterised. Guidelines developed for managing salmon farming in the Marlborough Sounds (Keeley et al. 2019a, also referred to as the 'Marlborough Best Management Practices (Marlborough BMP)') has a monitoring approach comprising different monitoring levels, with increasing intensity of effort; Type 1, Type 2, and Type 3 (Box 2). Similar approaches are employed internationally.

For new farms in open ocean areas, it may be most appropriate to undertake relatively intensive monitoring in the first stages of development, and to revise this monitoring as field data become available.

Box 2. Examples of tiered monitoring approaches

Marlborough Best Management Practices (Marlborough BMP)

The Marlborough BMP has 3 monitoring levels, including a decision framework to determine which level is applicable. Less-intensive surveys can be undertaken to assess organic enrichment effects if a farm has been in stable operation for some time, and if certain criteria are met (Type 1 monitoring). If Type 1 environmental quality standards (EQS-defined limits for acceptable effects) are breached, more intensive (Type 2) monitoring is triggered. Type 3 monitoring is the most intensive tier of monitoring. It is undertaken, for example, at baseline and again at year 5, to confirm effects are as predicted; or employed as needed to address other specific issues. Type 3 monitoring is designed ad hoc. In the Marlborough Sounds, 'Type 2' is the default monitoring level.

Assessment of sediment contamination by copper and zinc also uses a tiered approach, with increased testing requirements or levels of investigation if initial results exceed trigger levels.

Norwegian context

Tiered monitoring also occurs in Norway and Canada, where the anticipated or measured degree of impact determines the level of monitoring required. For example, in the Norway MOM system, the degrees of impact are defined as:

- low risk of pollution with minimal impact (compared to the holding capacity) which requires simple monitoring;
- moderate impact, where more complex monitoring is required;
- high impact close to threshold values (EQSs) where comprehensive monitoring is required.

In **Canada**, exceedance of an EQS triggers more in-depth monitoring surveys.

6.3 Pilot surveys

Pilot surveys are a field component used not strictly for data collection, but to develop, refine, and/or test sampling approaches, if deemed necessary, to ensure they are fit for purpose for baseline surveying and ongoing monitoring. Pilot surveys may be carried out before farm development or once development has commenced, depending on specific objectives. There are several reasons a pilot study might be warranted. Two reasons discussed here are (1) establishing sampling techniques, and (2) planning the sampling design. A pilot survey would therefore serve to target potential effects monitoring sampling methods, sampling locations, sampling effort, and monitoring indicators. Pilot surveys may be required for water quality monitoring; however, it is envisaged that most applications will be in the context of benthic monitoring and this section therefore focuses mainly on benthic surveys. For example, pilot surveys are most likely required for establishing monitoring of complex habitats in environments where there are practical constraints on some seabed video methods.

1. Establishing sampling techniques

In recognition of the logistical challenges that might be presented in some open ocean farming environments (e.g., high currents, depth), it is relevant to consider the practical limitations when planning surveys. For example, there may be limitations to planning robust quantitative biodiversity surveys that rely on visual methods. At deep, high current sites, the best method for obtaining quantitative information with the required precision, is likely a Remotely Operated Vehicle (ROV)[10]. Sites with less challenging working conditions may be sampled with sufficient repeatability using a drop camera approach. Towed video methods may also provide quantitative (or at least semi-quantitative) information. The precision and repeatability will depend on the sampling method, and the habitats being surveyed (e.g., heterogeneous vs. homogenous) and as such a site-specific pilot study may be necessary prior to baseline surveying, to establish cost-efficient but appropriate sampling capability along with suitable sampling design for ongoing monitoring.

2. Planning the sampling design

Without accurate knowledge of the distribution and characteristics of habitats that might require monitoring in and around the farming area, baseline surveys cannot be planned in detail. Furthermore, sampling design may vary depending on the equipment used for sampling (less precise sampling methods are likely to require higher replication). Without prescriptive planning, there is a risk of baseline surveys not providing the necessary information needed to support a robust monitoring programme. Pilot studies may thus be necessary in circumstances where monitoring is required, but where habitat information is not of sufficient resolution, spatial coverage, or detail.

Information on habitat distribution and variation within habitat types (e.g., changes in cover type, patchiness) is important for deciding the sampling design approach, targeting specific locations for monitoring, including approximation of the appropriate replication needed at different sampling levels (particularly in a stratified and nested sampling design, or for habitats that are patchy in nature). Pilot studies may also be required for further characterisation of the habitats such that specific initial indicators could be determined if information from the site characterisation surveys are not of sufficient resolution for this purpose. We note there are limitations to selecting indicators when little is known about the functioning and species interactions within some communities (i.e., those that are not well studied); see note in section 7.2.1.3.1.1 in relation to biogenic habitats monitoring indicators).

6.4 Baseline data

The purpose of gathering baseline data for monitoring is to obtain baseline information from the area of interest for comparison and assessment of environmental changes through monitoring once production starts. Baseline data must therefore be sufficiently robust in time and space to enable effective monitoring of environmental change.

Baseline data can be collected before and/or after consent has been granted. Data can be derived from a variety of sources, including descriptive surveys and other information that could have been collected independently of the consent application. Often, these other information sources are not robust enough to comprise a full baseline dataset, and therefore dedicated baseline surveys will be required. All baseline data must be collected before farming operations commence.

Although no specific guidance is provided for baseline data collection within the Guidelines, a baseline dataset should encompass information requirements in sections 7.1 and 8.1 (likely provided as part of the descriptive surveys and AEE) and be developed in accordance with the site-specific monitoring objectives, and concepts described for monitoring in the technical guidance sections. Baseline datasets should sufficiently describe the environment and establish natural variability or fine-scale descriptions of the existing natural environment that are to be monitored. Baseline surveys may also be required to fill a specific information gap identified during the consenting process and are commonly required to refine monitoring details, such as exact sampling locations, replication level, and sampling methods, once the final details of the application are known. Designing surveys aimed at gathering baseline data

[10] However, we point out that some of the most capable ROVs are likely to be very expensive at the intensity recommended in the following sections. It is out of scope of the Guidelines to make determinations on what level of expense is warranted but we note that cost will be a factor to be taken into consideration.

therefore requires a good understanding of monitoring needs and survey reports should provide recommendations to inform the design of EMOPs.

Specific objectives for collecting baseline data to support ongoing monitoring include:

- select appropriate and representative sampling sites to detect effects of the farm within the area of interest;
- identify suitable reference sites; and
- characterise baseline conditions at those locations, with an emphasis on relevant indicators and natural variability in both space and time.

6.5 Using models to inform monitoring

Numerical models are often used to support the prediction of effects of aquaculture. Although they are most commonly used to inform AEEs, model results are often also used to identify the location of monitoring sites and, depending on their complexity, monitoring indicators. Models may also be used in an ongoing capacity, for example to refine predictions of effects for subsequent development stages.

It is outside the scope of the Guidelines to provide technical advice on modelling. However, we do provide below a short background of the use of models to inform monitoring, followed by: (1) a short caution on choosing appropriate boundary conditions; and (2) a reference note to promote adherence to modelling best practice. Additional brief notes on modelling requirements are provided in sections 7.1 and 8.1.

Numerical model simulations can provide an indication of the spatial extent of discharges from finfish farms. In a benthic sense, this is often called the 'depositional footprint'. For water quality effects, the term 'plume' is often used to represent the extent of discharge from the farm. An important consideration for interpreting model simulations is the ecological relevance of predictions. For example, a model may predict the spatial extent of a numerically detectable effect, but this may not necessarily equate to an ecologically meaningful, or even measurable, effect. Also, effects may manifest outside this modelled spatial extent. For example, phytoplankton effects may manifest outside the modelled nutrient plume. The spatial extent and magnitude of the depositional footprint or plume depend on what metrics are simulated and what ecological condition they represent. Depositional farm footprint and plume modelling is normally done in an AEE and is important for informing monitoring design.

Models used to simulate the fate of finfish farm derived discharge have boundaries (i.e., a fixed area that is covered by the model simulations) and careful thought must be given to where the model boundaries are placed, and the nature of the boundary conditions that are applied. If inappropriate choices are made, the simulation results can be very misleading. It is important that model boundaries are chosen in a manner that ensures that slowly decaying material is not falsely lost from the system over relevant timescales. Allowing excessive false loss overestimates the influence of dilution processes. Equally, showing extremely low concentrations for a plume may overestimate, at least visually, the extent of effects.

It is also important to make appropriate assumptions about the fate of discharged contaminants. For example, if a contaminant does not decay after it has been generated and continues forever (spatially and temporally), the relative shape of the farm footprint or plume will evolve towards a quasi-stable equilibrium but the absolute quantity of contaminant at any location will grow indefinitely.

Numeric models are complex, and many decisions must be made on parameter values and the representation of ecological processes in the models. However, the Guidelines strongly recommend adherence to best practice for modelling to ensure that model results are scientifically sound and provide a robust evidence base in the decision-making process.[11] Best practice for modelling processes is available (e.g., Jakeman et al. 2006, USEPA 2009) and a strategic framework for implementing best practice guidelines in environmental modelling has recently been suggested by Jones et al. (2020).

6.6 Limitations and opportunities of monitoring techniques for open ocean

As previously discussed, inshore finfish farming has a moderately long history and benthic and water quality monitoring methods are relatively well established. In contrast, little experience has yet been accrued with open ocean finfish farming.

Furthermore, there are limited data on the characteristics of many open ocean areas. Where there is a lack of experience or techniques as applied to open ocean, the guidance for monitoring provided in sections 7 and 8 is largely based on extrapolation (where deemed appropriate) from inshore experience, taking into account the unique characteristics of, and logistical challenges for, benthic and water quality monitoring in open ocean environments.

Open ocean environments are likely more exposed than existing finfish farming areas. Winds will likely be stronger offshore, and swell and waves are more likely to impact working conditions. Sites chosen for open ocean finfish farms are also likely to have higher peak current speeds. Vessels may have difficulty holding station under some wind/current conditions. Adverse weather is also likely to cause more frequent delays and interruptions to planned sampling campaigns. Accordingly, site conditions may dictate the nature and timing of sample collection, and/or the type of vessel used. Sample collection devices may also need to be more heavy duty to cope with working in more exposed sea conditions (swell and/or waves). In addition, some vessels (e.g., those with higher sides) can make instrument deployment and retrieval more difficult than in coastal waters.

Specific challenges for water quality monitoring are described in Appendix 5. At the time of preparing this version of the Guidelines, the pool of effective water quality monitoring approaches and technologies were limited (even for inshore); however, there are emerging opportunities for new approaches. For example, an opportunity discussed in the Guidelines is the potential use of satellite imagery for water quality monitoring (Appendix 6). Benefits and challenges of continuous monitoring platforms are also discussed in section 8. The anticipated challenges that might apply to open ocean seabed monitoring are discussed throughout section 7 where relevant.

6.7 Overarching recommendations

The following general recommendations are made for monitoring:

- Monitoring, descriptive, baseline and pilot surveys should be designed and conducted by independent, suitably qualified, and experienced persons.
- Laboratory analyses must be conducted at qualified facilities that exhibit good laboratory practice according to standard methods, including taxonomy, preferably by accredited laboratories.

[11] To illustrate the need for this we refer to Özkundakci et al. (2018) who reviewed 68 New Zealand legal decisions where modelling evidence was challenged in legal proceedings. All the legal challenges were substantive, relating to the scientific components of the model (e.g., assumptions, input data, and parameters), model evaluation, or application. None of the challenges were regulatory process challenges. The authors concluded that it appears that modelling best practice guidelines are not always being followed and that, if models are to be of substantial help in environmental decision-making, then modellers and decision-makers will need to ensure that there is a clear understanding of the purpose of a model, the modelling process is transparent, limitations are acknowledged and considered, and that best practice guidelines are followed.

- Information on the environmental effects of open ocean aquaculture, including research findings and monitoring results, should be made accessible and publicly available in a timely and meaningful manner to achieve transparency and support public awareness and engagement. Over time, consideration should be given to incorporating the data (where not commercially sensitive) in an existing, or new, public portal for this information (e.g., an 'open ocean dashboard').
- To facilitate effective reviews of the Guidelines and sharing of knowledge for science innovation, all parties, including open ocean finfish farming consent holders, science providers, and all branches of government are encouraged to make available the findings from monitoring, including monitoring reports, where these are not commercially sensitive. This includes access to descriptive, pilot, baseline, and monitoring survey reports and data.

7. BENTHIC MONITORING

7.1 Application-specific information requirements for monitoring design

Monitoring design requires a certain level of information on the environmental characteristics of the relevant area of interest and the predicted nature and spatial extent of effects of the finfish farm operation. The technical benthic monitoring guidance provided in this section works on the basis that the following supporting information will be available to inform monitoring design:

- Spatially explicit **depositional modelling**. Modelling should be specific to the farming operation. Depositional modelling should include farm deposition flux to the seabed[12] and, in dispersive sites, resuspension of farm deposition (i.e., secondary dispersal) and potential for accumulation in far-field accumulative areas. For sites where currents and resuspension are likely to vary across the farming area, the modelling should incorporate spatially varying current fields. Furthermore, in areas where wave-driven currents are prevalent (especially near the seabed), wave-driven currents should also be accounted for in the depositional modelling. Model runs should simulate sufficiently long periods to enable the incremental organic matter at the seabed to be close to equilibrium. Modelling methods, uncertainties, and limitations should be clearly listed.

  If the scope of the proposed farming operation as described and modelled in the AEE changes through the consenting process, depositional modelling may need to be updated prior to the development of the EMOP.
- A **habitat map** displaying substrate type (e.g., sediment texture, rugosity) and conspicuous infaunal and epi-benthic features. Ideally these data would be obtained through swath mapping of the potential impact area, which would provide a spatially robust coverage of the area (for example, side-scan sonar or multi-beam echo-sound surveys). Acoustic methods should be ground-truthed using sediment sampling and seabed imaging (e.g., still photos or video) and be carried out at a resolution that ensures sensitive habitats are not missed. The habitat map should cover the areas potentially affected by organic enrichment.
- Broad **characterisation of those habitats (and associated communities of organisms)**. This should include a general description of that habitat/community, including common and important[13] taxa that are present, as well as a general description of its rarity or representativeness when compared with a broader spatial extent. Benthic habitats and species of high ecological value or susceptibility to deposition from finfish farms should be identified.

[12] The amount of particles accumulating on the seabed.
[13] For example, species that have a fundamental function within the community, rare species, or protected species.

- **Assessment of how the habitats and communities are predicted to be affected** by the proposal. This should include a summary of uncertainties.

7.2 Guidance for monitoring

The potential benthic effects that may result from open ocean fish farming and for which technical guidance is provided, as identified in section 4, are:[14]

1. **Organic enrichment** and associated changes in the benthic environment due to biodeposition or biofouling drop-off.
2. **Smothering** of benthic habitats and communities by biodeposits or biofouling drop-off.
3. **Contamination** of sediments due to the deposition of therapeutants, feed additives, antifoulants, or other potentially harmful substances used in finfish farm operations.

Table 2 shows the structure of the technical benthic monitoring guidance provided. For organic enrichment and smothering, this is given according to habitat types which require different monitoring approaches: (a) soft sediment habitats, and (b) biogenic and other complex habitats.

With respect to monitoring contamination of sediments from potentially harmful substances, we provide specific guidance for potentially harmful substances that are commonly used in New Zealand (zinc, copper). Some considerations are given for monitoring other potentially harmful substances, but this is not intended to be exhaustive because monitoring will require contaminant-specific decision-making.

Table 2: Potential benthic effects for which monitoring guidance is provided with reference to habitat type and section.

Potential effect	Section reference and habitat type
Organic enrichment and associated changes in the benthic environment due to biodeposition or biofouling drop-off Smothering of benthic habitats and communities by biodeposits or biofouling drop-off	Section 7.2.1* Soft sediment (section 7.2.1.2), Biogenic and other complex habitats (section 7.2.1.3)
Contamination of sediments due to the deposition of therapeutants, feed additives, antifoulants, or other potentially harmful substances used in finfish farm operations	Section 7.2.2 Soft sediments only

* Combined guidance is provided for monitoring effects from both organic enrichment and smothering since the techniques for monitoring them are similar in principle.

7.2.1 Organic enrichment and smothering of benthic environments due to biodeposition or biofouling drop-off

7.2.1.1 Background

Similar strategies would be employed to monitor effects of both organic enrichment and smothering from biodeposition, and this section provides guidance for both.

Microbial decay of organic wastes from fish farms can substantially alter the sediment chemistry of the seafloor, e.g., deplete oxygen levels, elevate free sulphides, and reduce redox levels. Visible bacterial cover may occur even at moderate levels of deposition. Changes in sediment chemistry, increased

[14] See Appendix 2 for a description of the process followed by the working group to identify these potential effects.

organic flux (as a food source), and smothering can all contribute to modification or displacement of benthic infaunal and epifaunal species/communities, as well as algal communities.

The physical characteristics of the farm site and attributes of the farms themselves[15] influence the accumulation of organic material on the seabed. The flushing potential and environmental assimilation of farm wastes at a given site are largely dictated by water depth and water movement, including current speed, and, to a lesser extent, seasonal factors such as water temperature. Increased flushing not only reduces local biodeposition but generally also promotes oxygenation of sediments (Findlay & Watling 1997). Sites in deep water with strong water current velocities tend to feature a more dispersed depositional footprint with lesser maximal organic enrichment than shallower sites with lower flushing ability (Pearson & Black 2001, Aguado-Gimenez & Garcia-Garcia 2004, Keeley et al. 2013a). This is due to the greater range of dispersal of fine particles and flocculent material and increased levels of particle resuspension (Law 2019).

Resuspension/dispersion of deposited particles is likely to be a feature of sites targeted for open ocean fish farming[16]. Accordingly, there is likely to be less accumulation of organic material within the sediments in the immediate environs of the pens, and severe organic enrichment impacts are thus less likely to manifest in those regions. For example, monitoring of the seafloor at an open ocean finfish aquaculture facility in the western Gulf of Maine, USA, indicated minimal or no significant differences in benthic communities between impact and reference areas (Grizzle et al. 2014). Nonetheless, there can still be substantial changes to seabed habitats and fauna. New Zealand and overseas studies at sites considered to be dispersive have shown that benthic effects tend to be most evident directly beneath the pens and exhibit a strong gradient of decreasing impact with increasing distance (Keeley et al. 2013b, Keeley et al. 2019b). However, at some dispersive sites with complex bathymetry (and hydrodynamics) and substrate types, there can be substantial patchiness in deposition and associated effects (Broch et al. 2017). Accumulation of organic material is most likely to occur in seafloor depressions, and in areas with higher rugosity (surface roughness), such as rocky or cobbly substrates. In these areas, lower shear velocities encourage deposition of particles, and the variable seabed relief can protect them from resuspension.

Given the widespread nature of organic wastes and complexities associated with predicting its distribution through resuspension processes at dispersive sites, more emphasis may be required to understand the patchiness of benthic deposition at dispersive sites, in particular areas of potentially disproportionate accumulation (e.g., far-field accumulation zones). We do note that not all open ocean areas are dispersive, and that the monitoring design will clearly need to be tailored to the site conditions.

The technical guidance for monitoring effects from organic biodeposition is detailed in the following sections and is broken into two broad habitat types (representing the conceptual monitoring focus areas): soft sediments, and biogenic and other complex habitats (e.g., rocky reef communities).

7.2.1.2 Soft-sediments

7.2.1.2.1 Limitations of soft-sediment sampling methods

The soft-sediment sampling methods described here are based on those commonly used for monitoring organic enrichment in inshore finfish farm monitoring in New Zealand, as well as overseas, and include strategies employed for monitoring soft sediment communities for other purposes, including open ocean areas (e.g., benthic effects of oil and gas operations). The methods are inherently focused on sediments and infaunal communities and, therefore, are not an appropriate approach for monitoring epifauna that may inhabit soft sediment habitats (e.g., scallops, brittlestars). Soft sediment sampling methods are inherently limited in their ability to represent large, patchy, or rare infauna (e.g., *Nephrops*, burrowing

[15] Including fish stocking density, the settling velocity of fish faeces, the type of feed and feeding system, the type of pen structure utilised, and the amount of flow modification caused by the pen system.
[16] Though we do note that some open ocean sites may not be highly dispersive.

anenomes). If such organisms are required to be monitored, then special consideration should be given to these within the sampling design[17] on a species/site specific basis (a pilot study may assist with this, see section 7.3.1).

7.2.1.2.2 Sampling methods

7.2.1.2.2.1 Monitoring indicators

The default monitoring indicators selected for soft sediment monitoring of organic enrichment habitats are based on those commonly used in New Zealand and global finfish aquaculture (e.g., Keeley et al. 2019a) and are generally consistent with organic enrichment indicators used globally (Environment Protection Authority Tasmania 2019).

Specifically, default quantitative soft sediment indicators are given below.

- bulk measures of sediment organic matter — total organic matter (% ash-free dry weight),
- sediment chemistry indicators — redox potential and total free sulphides[18], and
- infaunal community health measures — total infaunal abundance, number of taxa, and calculated diversity/biotic indices[19,20].

Some of these organic enrichment indicators could later be replaced with new indicators (once methods have been assessed as robust for monitoring and relationships with infaunal communities have been established). One candidate for potential future use is eDNA, which could be used as an indicator of benthic functioning and enrichment state. Information on eDNA and possible applications for monitoring effects of open ocean aquaculture are provided in Appendix 4.

Quantitative indicators should always be used in preference to qualitative indicators, particularly for compliance purposes. However, qualitative indicators can be relatively easy and cheap to obtain and may supplement[21] the sampling programme. Common qualitative indicators are:

- visible bacterial coverage — visible bacteria can manifest in patches at dispersive sites at moderate to high levels of enrichment, with coverage increasing in higher enrichment conditions;
- sediment colour/texture/odour — black, sticky, sulfidic smelling sediments indicate low-oxygen sediments that can be associated with organic enrichment; and
- outgassing (e.g., normally on disturbance of the seabed with a camera frame or similar, but also by presence of mobile sub-surface bubbles in the cores) once the sample is at the surface—this is typically only seen at very high or excessive enrichment levels that are typically beyond peak-of-opportunist conditions.

We note the use of visual indicators for seabed enrichment (particularly video methods) are often limited in their capacity to detect effects at highly dispersive sites, particularly at moderate to intermediate impact levels (Hamoutene et al. 2018, Keeley et al. 2019b).

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[17] For example, increased replication, larger sample sizes, alternative collection methods.

[18] This includes analysis using either the traditional ion-specific electrode (ISE) method (e.g., Wildish et al. 1999), or the UV spectrophotometer method (which more accurately measures free sulphides in sediments; Cranford et al. 2017, Cranford et al. 2020). We note that the data acquired using these respective methods have differing relationships with infaunal health which would need to be accounted for in the data interpretation.

[19] For example, see table 9 of the Marlborough BMP (benthic) (Keeley et al. 2019a).

[20] In environments with mobile sediments and/or naturally depauperate infauna (e.g., high physical disturbance regime), infaunal communities may have high temporal variability. In addition, microbial assimilation pathways may be more important waste processing mechanisms in some of these areas. In such cases, eDNA may be considered as a better indicator to use for mapping the organic enrichment effect footprint (see section 'footprint mapping').

[21] For example, at later stages of development, tiered monitoring may use qualitative indicators.

7.2.1.2.2.2 Sample collection

A grab sampler (e.g., van Veen) is the most widely used method for collection of samples in soft sediments and is equally effective in open ocean environments. The larger samplers (e.g., van Veen) are likely to be the most versatile, but any type of grab could be used as long as it successfully collects sediments with an undisturbed surficial layer and penetrates the sediments such that the sample depth and volume allowed for adequate sub-sampling for the relevant indicators across the appropriate sediment profile. Sediments with components of large shell material (or other hard biogenic structure) and cobble/rock can be challenging to sample effectively using a van Veen grab. In these environments, collection techniques may need to be modified, or other equipment (e.g., ROV with core sampler) may be required to obtain good sediment samples.

Conditions in some open ocean environments may present practical challenges for grab sampling (and seabed imaging) or other sample collection methods. For example, drift of the grab during its descent due to high currents or misfiring of the grab before it reaches the seafloor, due to the ship listing in swell or waves. These challenges can typically be offset by equipment modifications such as using the grab in a frame or adding extra weight for consolidated sediments. However, at locations near to the farm structures (pens, mooring lines), there is increased risk of entanglement of sampling equipment and also a risk of interaction between the sampling vessel and the surface structures. In some instances (e.g., certain ocean conditions or tidal states), this risk may preclude the ability to sample close to the farm, so careful planning is required. We recommend a map of sub-surface structures is given to the science provider at the monitoring planning stage, to help minimise risk of interaction with sampling equipment.

Vessel and sampling equipment drift can be substantial (10s to 50s of metres or more) in deep and high current areas, or high wind conditions. Although vessels with dynamic positioning systems can reduce/prevent vessel drift from the target location, these will not counteract drift of the deployed equipment during descent through the water column. The use of underwater positioning systems (USBL) is recommended to determine sampling precision where necessary (relative to target, spatial repeatability, etc.). Alternatively, such sampling should be constrained to slack tide.

7.2.1.2.3 Sampling effort

7.2.1.2.3.1 Sampling locations

Initial sampling locations should be chosen based on information from the depositional modelling (patterns of organic deposition and accumulation), areas identified as priorities for monitoring in the AEE, the habitat map available for the site (e.g., substrate type, epibiota), and bathymetry. Sampling locations may be refined following the baseline survey, and certainly following any depositional footprint mapping exercise (if undertaken; see section 7.3.2 for more information on depositional footprint mapping).

Monitoring within the farming area could be approached with an emphasis on a single block or could target the broader farming area (see Figure 4). A broadscale approach is likely to be more effective if there is reasonable interaction of organic wastes between farm blocks within the wider area. The most appropriate approach to take will depend on site-specific environmental and operational characteristics, as well as the monitoring objectives and site specific EQSs (if any).

Although the station selection will be site-specific, the initial sampling concept for both broad-scale and farming block approaches would ideally encompass the two minimum sampling location types listed below:

1. **Areas of maximal effect (AME**). As informed by the AEE and, as the name suggests, the area where the highest effect is likely to occur. The AME may be directly beneath the pen, or some distance from the pen depending on the prevailing site conditions. In either case, it is recommended that the beneath-pen area is monitored at least in the initial monitoring period.

When the area to be monitored is beneath the pen, sampling locations should be located as close as practicable without risk of entanglement in farm structures.

AME stations may not necessarily need to be allocated at all farm blocks, as long as enrichment conditions at the block(s) selected for monitoring are likely to be representative of (or greater than) at the other block(s) within the farming area. The number of AME stations allocated to any given block must be sufficient to capture the magnitude of effects and should also reflect the level of certainty/uncertainty associated with the predicted effects.

2. **Reference areas (REF)**. Placement of initial reference stations should target areas of comparable flow, substrate, and depth to the predicted impact areas. Reference stations should be located outside the area of interest (likely kilometres at dispersive sites) with a buffer between them and the predicted impact area.

Additional sampling stations may also target:

3. **A boundary of specified effect level (BSEL)**. The necessity of these stations will be site-specific; for example, if effects of a certain magnitude are required to be spatially constrained[22]. In this case, EQSs may be developed to set the allowable degree of change from background organic enrichment conditions at these boundary locations.

   The placement of these stations along the boundary could be along the main current axis and/or oriented in the direction of depositional footprint distortion as indicated by modelling/monitoring, or systematic design (e.g., gradient, radial, or regular interval). Note that BSEL monitoring (Figure 4A) may not be as desirable for farm sites where there are irregularly shaped or patchy effects footprints.

4. **Areas of possible accumulation (APA)**. These stations should target areas outside the 'farm footprint' that have been identified as being susceptible to accumulations of organic deposits from multiple farms or groupings of farms, and/or areas where organic material at otherwise low-levels might naturally accumulate to levels that might induce an effect (e.g., depositional areas like eddies, large seafloor depressions, boundaries of more complex substrates[23], or areas where deposition footprints overlap). Placement of these stations should be informed by depositional modelling and may include an element of expert judgement (for example, if depositional modelling has not taken account of spatial variation in resuspension probabilities). Areas of possible accumulation outside the 'farm footprint' will typically be placed in the far-field as a means for longer-term surveillance of broader-scale enrichment effects. The need for these stations is also likely to be dependent on spatial constraints of effect and how dispersive the site is (e.g., less likely to be needed at less dispersive sites).

[22] The requirement of any such boundary is likely to be defined through the consenting process.

[23] It is noted that rugose substrates may require a different sampling technique (i.e., may not be effectively sampled using a grab sampler). Also, these areas may require different or supplementary monitoring approaches because habitats may not be uniform soft-sediments.

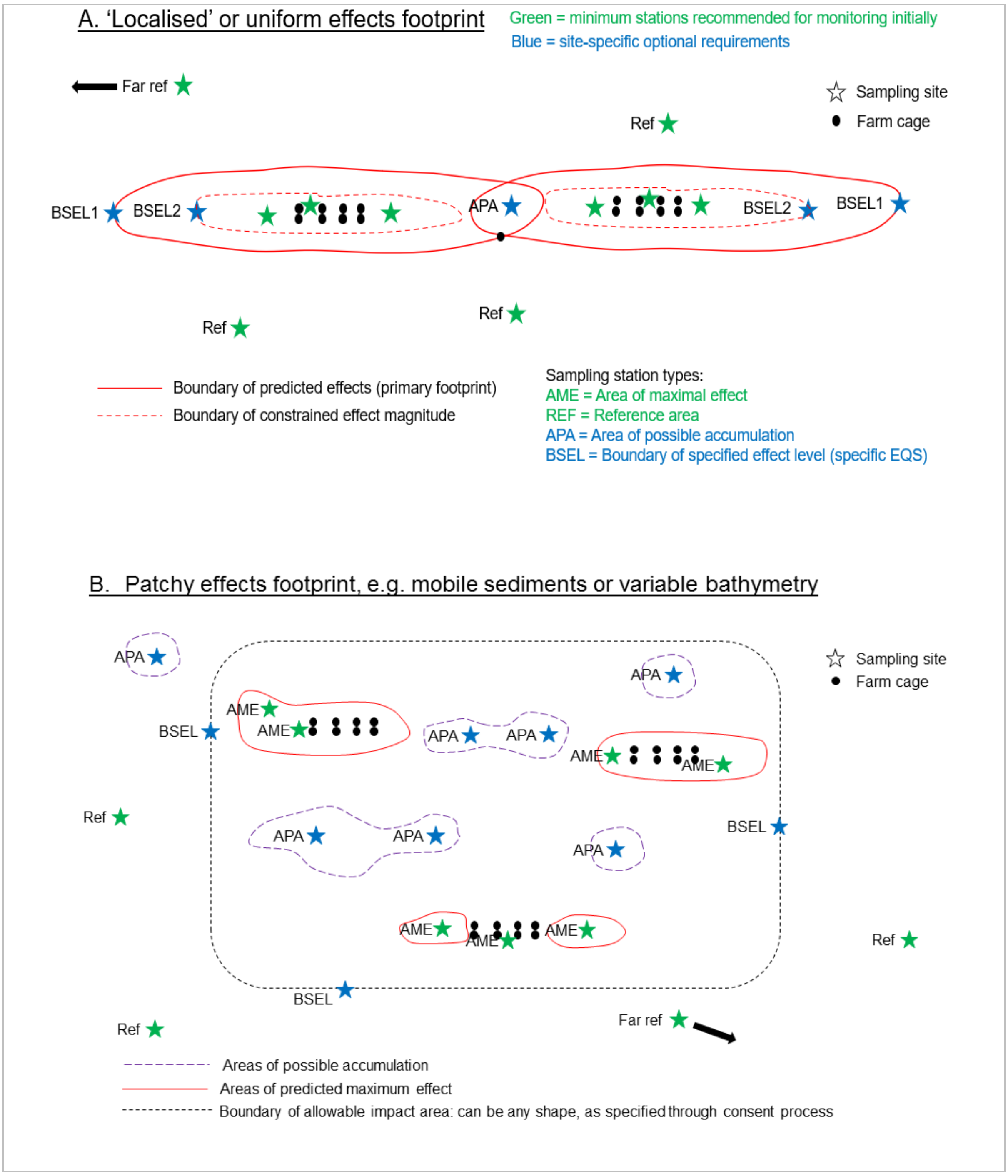

Figure 4: Stylised example of possible sampling approaches for two farming areas: (A) possible sampling approach for a scenario where the farm footprint is expected to be localised or uniform; and (B) possible sampling approach for a scenario where the farm footprint is patchy. Note that this stylised approach provides examples only for locations to be monitored, not the level of appropriate sampling effort. Sampling effort should be allocated on a site-specific basis.

7.2.1.2.3.2 Replication

For effects-based monitoring programmes, experimental design must be such that the agreed level of effect can be detected reliably, and appropriate replication is one element of design efficacy.

Hypothesis testing (including equivalence testing) is a robust approach to determine effects in effects-based monitoring programmes. To avoid inference mistakes (i.e., Type I and Type II errors) when using this approach, it is critical to perform a power analysis and use this to determine the level of replication. Key considerations of a power analysis include a) the statistical test to be used, b) the acceptable level of uncertainty, c) the effect size to be detected[24], and d) the existing variability in the data. The existing variability in the data is specific to the site (e.g., site-specific natural variability), and the indicator/s being used to measure the effect, thus would require data from a pilot or baseline survey[25]. Appropriate stratification can be used to reduce variability and, therefore, improve statistical power.

Power analysis may also be appropriate to evaluate later stages of the monitoring programme. For example, once variability within affected community types has been established, it can also be used to check that the level of replication is still appropriate or whether changing the indicators used for monitoring is required. Importantly, we note that the replication required to reliably detect effects using hypothesis testing can sometimes be beyond practicality (e.g., when there is high variability, or for detecting a small effect size). In this case, adjustments may be made like setting a new EQS, using a different indicator, or adjusting other aspects of sampling design (e.g., frequency and/or stratification to allow for more replication effort per strata). If the required level of replication to detect the effect is prohibitive, and there are no workable and acceptable trade-offs, hypothesis testing may not be an appropriate approach.

An alternative statistical approach is to use point estimates (averages) and the associated intervals, typically 95% confidence intervals, which consider the mean and the variation for compliance determination. This approach is used by Keeley et al. (2019a), for inshore salmon farm monitoring. It should be noted that when using this approach, wide confidence intervals indicate a lack of information and should be considered a warning that the sample sizes are too small.

Whichever approach is selected (e.g., hypothesis testing or otherwise) this should be captured in the EMOP, along with the process or justifications used for determining replication.

7.2.1.2.3.3 Sampling timing and frequency

Generally, it is best practice for monitoring of organic enrichment from fish farms to target the period of highest biological impact. However, timing (and frequency) of sampling is likely to be somewhat farm/site specific depending at least on the farming operations and environmental considerations, so this section discusses general considerations and concepts that might be relevant to inform the timing of monitoring under site-specific EMOPs.

The highest near-field enrichment effects to seabed communities during a single year class production cycle in inshore finfish farming are typically seen around peak biomass (when feed discharge is or has been at its highest) (SEPA 2019, Keeley et al. 2019a, Keeley et al. 2019b). Monitoring of effects at single year class open ocean finfish farms should target this period. By contrast, monitoring of farms that have a reasonably constant feed discharge throughout the course of a year (e.g., multiple year classes in a single block) may be best sampled during summer, when water temperatures are likely to be warmest[26], biological activity is highest, and thus when enrichment effects are likely to be more pronounced. Summer is also often when the period of highest feed discharge occurs in this type of

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[24] For example, x% change of one or more indicators, compared with reference conditions. The effect size should be relevant to the limits of acceptable change or EQS.

[25] It is recommended that replication for soft sediment infauna sampling in the baseline surveys defaults to a minimum of n=5 (or, collect 5, analyse 3, and archive the remaining 2 in case further replication is required). Higher or lower replication may be appropriate if the variability (in space and time) of communities is known. This level of replication is best determined on a site-specific basis.

[26] We note that near-seabed water temperatures may not fluctuate a large amount at some sites (e.g., deep sites, and those depending on site-specific currents).

farming operation. As mentioned in section 7.2.1.2.3.1, some stations (e.g., AME stations) may not necessarily need to be sampled at all farm blocks every time, as long as enrichment conditions at the block(s) selected for monitoring are likely to be representative of (or greater than) the other block(s) within the farming area. Sampling could also alternate between farm blocks when there is a good understanding of the environment/effects.

By contrast, within a single farming area, farm blocks may either be operated synchronously (stocked and harvested at the same time), with variable timing for stocking, as well as with variable stocking/harvesting strategies between blocks (single vs. multiple year classes). These operational details should be considered when determining timing (and frequency) of monitoring at near- and far-field stations within the farming area. For example,

- farming areas where individual farm blocks are operating synchronously could all be sampled at the same time at or just after peak biomass (see Figure 5A), and, in this case, near-field sample timing could be easily determined using the farming areas feed discharge forecasts[27];
- greater sampling frequency (i.e., within the production cycle) during the first operational period may also be appropriate if there is a high level of uncertainty/risk associated with near-field effects; and
- far-field monitoring (for accumulative effects) could take place at the same time as monitoring targeting the period of highest near-field effects (i.e., for a 20 months production cycle, this would be approximately every 2 years), or could simply be done annually, depending on site-specific considerations.

Monitoring strategies for both multiple year class and single year class farm operations should aim to capture progressive increases in effects, both in terms of intensity and spatial extent. Progression of effects may be less likely to occur if fallowing practices are used[28] (and are of sufficient duration), and/or if the effect intensity is low.

When farms are not operating synchronously (i.e., stocked at different times, whereby some farm blocks may be at peak discharge when others are fallowed; Figure 5B) timing that is fixed based on calendar years or seasons (e.g., annual) may be more appropriate. This approach may be particularly useful for farming areas with an emphasis on monitoring effects in the far-field, and at sites where there may be strong temporal variability (e.g., seasonal, shallower areas with a lot of storm driven flows at the seabed). Alternatively, a combined approach could be employed, for example:

- Where a broad-scale monitoring approach is used with fixed survey timing (e.g., annually, bi-annually), but a sub-set of farm blocks (i.e., those that are incidentally at peak discharge) are also sampled during the surveys.
- The timing for sampling for the AME and far-field monitoring stations could be decoupled from one-another. Therefore, the AME and other farm-scale stations are sampled during peak biomass, and the broader scale stations (e.g., APA, Reference and BSEL) are sampled at fixed intervals such as annually. A regime such as this should be carefully considered so that it is not too onerous or impractical to implement.

[27] Inshore farms growing King Salmon have a grow-out period of approximately 20 months. In a single year class operation, this could mean sampling is only undertaken approximately every 2 years (considering a fallow period between harvesting and re-stocking). Production cycle lengths for open ocean farms are unknown. These may vary considerably from the inshore grow-out duration, depending on how large the fish are when transferred to site, as well as other determining factors such as harvest size and how the fish cope in open ocean conditions.
[28] Though we note that fallowing may not be required if effect levels are low.

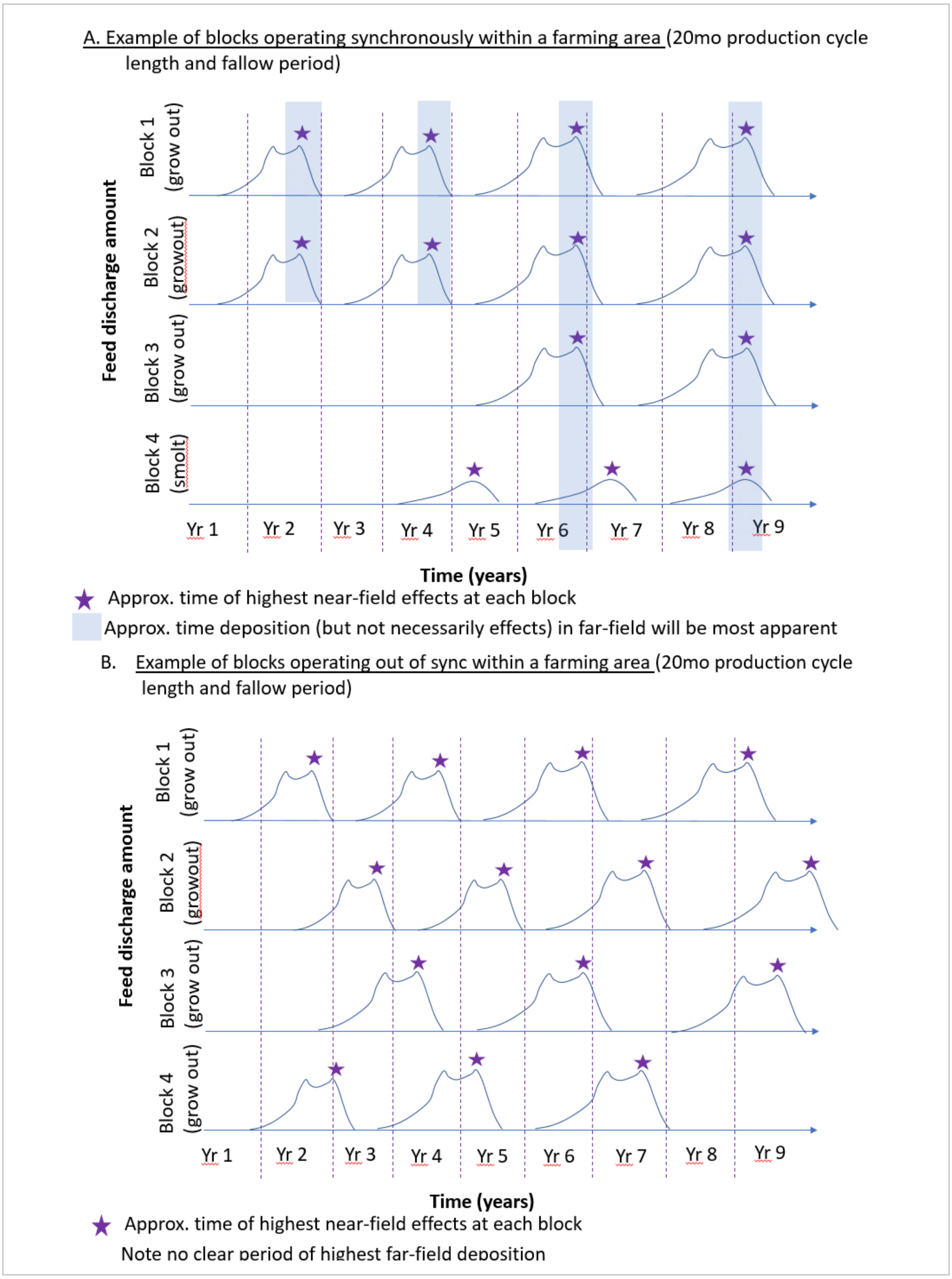

Figure 5: Example of different farming operational strategies for informing selection of sample timing in near- and far-field locations. (A) Example of farm blocks operating synchronously within a farming area; and (B) example of farm blocks operating out of sync within a farming area.

7.2.1.3 Biogenic and other complex habitats

Although we assume a preference for farm siting will aim to avoid biogenic or other complex seabed habitats (hereafter complex habitats) of ecological importance, they may be present in the area potentially affected by organic deposition (near- or far-field) and may need to be monitored. Not all biogenic and other complex habitats are considered to be 'ecologically important'[29], and the guidance in this section is generally applicable to monitoring of most epifaunal features.

It is noted that many complex habitats, particularly in deeper or offshore waters are data-poor, often with little known about underlying ecosystem processes (Loh et al. 2019) and their initial response or resilience to anthropogenic impacts such as organic enrichment. In such cases, it is expected that monitoring techniques for complex habitats (and indicators) will evolve through time. It is also acknowledged that dedicated research programmes may play a key role in refining monitoring programmes or addressing risk and uncertainty associated with farming in areas in or adjacent to complex habitats, and such research should feed into the EMOP as it becomes available. Where complex habitats are data-poor, logistically challenging to sample, or sampling designs require detailed site-specific information, we recommend that a pilot survey (see section 7.3.1) is undertaken prior to the baseline survey to ensure that proposed sampling techniques for baseline surveys and ongoing monitoring are appropriate.

We used a recent review paper (Loh et al. 2019) to inform monitoring design guidance provided in this section, and readers are referred to the original document for more detail.

7.2.1.3.1 Sampling methods

7.2.1.3.1.1 Monitoring indicators

Indicators selected for monitoring will clearly depend on habitat type and, importantly, the habitat values intended to be preserved and/or the level of protection to be achieved. We discuss potential monitoring indicators here as 'key taxa' and 'key parameters'. Key taxa are specific organisms or taxonomic groups of interest with regard to effects, and key parameters reflect what is measured about those organisms/groups.

Key taxa

Generally, the main habitat forming taxa within the complex habitats are likely to be most fundamental to indicating habitat health (Loh et al. 2019), so that is where most monitoring should focus. There may also be other taxa within the community that perform functions critical to the health of the habitat (e.g., keystone species). For example, horse mussels in sufficient densities and extent form beds which then provide hard surfaces for other encrusting animals to attach to, and, in combination, these create habitat for various mobile invertebrates and fish. In this example, the horse mussel density/abundance/extent may be the best indicators for the health of the wider benthic community/ecosystem.

There may also be taxa (or groups of taxa) that provide early indications of habitat change (i.e., those sensitive to increased organic loading); for example, within areas of accumulation there may be increased prevalence of scavengers and detritivores, or reduced condition of encrusting communities. Other taxa that might be important for monitoring are endangered or protected species within the community.

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[29] The term 'ecologically important' is used on purpose in a vague sense to allow for context-specific interpretations. Many councils have criteria to identify significant ecological areas, and these could be used to determine ecological importance. Because there is variability among regions as well as between regional criteria and national classifications (e.g., the Department of Conservation Key Ecological Area criteria), we did not attempt to define specific criteria for this guidance. We recommend that in instances where it is necessary to determine ecological importance existing scientific guidance is used, especially Anderson et al. (2019) and MacDiarmid et al. (2013).

Where community function and resilience/response to organic enrichment are not well-known, a broader analysis of indicators could be included initially, to be refined as understanding of the effect increases, either through collation of monitoring results over time or separate research programmes aimed at indicator development (see section 7.2.1.2.2.1).

Key parameters

Centred around key taxa[30] as defined above, monitoring will most likely include measuring parameters to indicate either habitat quality, ecological functioning, and/or spatial extent. With respect to habitat quality indicators, key organisms could be monitored by tracking taxa-specific density (e.g., number per square metre), biomass, percentage cover, and population size structure (of habitat forming organisms). Samples of individuals from key taxa for laboratory analysis may also provide early warning signs of stress and chronic effects in key organisms, for example, physical condition index or biochemical changes in tissues.

If used, indicators of ecological functioning should also be habitat specific, although successful larval recruitment into the benthic community (e.g., potentially affected by sedimentation) is likely to be a relevant ecological function indicator to most complex habitats.

Other general indications of organic enrichment may also be relevant to indicate potential deterioration of the habitat, for example, visible increases in sedimentation or bacterial growth.

7.2.1.3.1.2 Sample collection

Given that most open ocean farming sites are less likely than inshore sites to be suitable for survey by divers, monitoring is likely to use remote seabed imaging methods.

Video transects (ideally fixed start/end points) can provide a broad overview of habitat and may be used to help assess spatial extent of habitat (e.g., measure patch size/patch density) or counts of conspicuous larger or rarer organisms for density measures. By contrast, still images (e.g., quadrats) provide for higher resolution analyses that might be required for some indicators of habitat quality (e.g., percentage cover, density of inconspicuous or small organisms, size of individuals).

Seabed footage and images could be collected by divers (where appropriate), towed video, drop camera (with attached quadrat frame), or remotely operated vehicle with imaging attachment/s. The equipment used will depend on site conditions, and we note high energy sites can present challenges for many of these collection methods. For example, in strong tidal currents remotely operated underwater vehicle (ROV) tethers can create substantial drag and ROV navigation becomes difficult. Equally, video collection can be challenging when vessel positioning is not well controlled (i.e., influence of surface conditions on drift direction, vessel angle, and speed relative to water current direction).

Methods for physical sample collection will depend on the habitat and the sample being collected. Because many biogenic habitats contain hard structures, van Veen and similar grab sampling methods are unlikely to be appropriate; diver collection or ROV collection are likely to be the best collection methods (where they are practical).

Surveys related to measuring extent of biogenic habitat should include remote acoustic methods (side-scan, multibeam) coupled with ground-truthing through georeferenced video or still imagery (see above). However, if only a small area of biogenic habitat extent needs to be monitored, video transects may be a more appropriate method.

[30] Or their surrogates, if established.

7.2.1.3.2 Sampling effort

7.2.1.3.2.1 Sampling locations and replication

Sampling station placement should be determined based on habitat mapping, pilot surveys (where necessary), and depositional modelling. Sampling locations may be refined following the baseline survey, and if additional depositional footprint mapping is undertaken (refer section 7.3.2).

Potential effect stations

Sampling station placement will be highly dependent on the distribution of the biogenic habitat and the distribution of the predicted (or observed[31]) depositional footprint. Firstly, sampling stations may need to be stratified by different habitat characteristics to enable effects to be detected against background variations. Secondly, sampling stations may need to be stratified by distance to block/s or be allocated to zones based on relative accumulation of waste as shown in depositional modelling.

The number of sampling stations sufficient to capture potential farming effects will depend largely on the heterogeneity of the habitat (or the indicators being used) and the spatial scale and approach[32] of the sampling technique (Loh et al. 2019). For example, fixed sampling stations[33] may reduce the required sampling replication (as opposed to random sampling designs) without compromising statistical robustness. This is discussed further in section 7.2.1.3.2.2. We note that fixed sampling stations can limit the generality of conclusions, so would ideally be supplemented with sampling from non-fixed locations.

Reference stations

As with soft sediment monitoring, the placement of initial reference sites should target areas of comparable flow, substrate, habitat characteristics, slope, and depth to the potential impact areas being monitored. Placement of reference stations should be located such that there is a reasonable buffer from the predicted impact area (at least 2x the distance of the predicted effects footprint in the relevant direction). The number of reference sites, and placement of sites, should account for future possible development stages (i.e., enough that some can be sacrificed to future development impacts). In addition, the number of reference sites should also consider natural variability and statistical power for detecting farm effects.

Sentinel sites in soft sediment areas

Soft sediments are likely to be an easier habitat to monitor than biogenic habitats and soft sediment habitat is likely to surround areas of biogenic habitat. If the soft sediments are thought to be equally (or more) sensitive to organic enrichment than the biogenic habitat[34], then soft sediment sentinel monitoring sites could be placed on the boundaries of the biogenic habitat. If organic enrichment at these sentinel sites was detected, then this monitoring would serve for an early warning that the extent of organic enrichment is close to the biogenic habitat boundary, and, if necessary, invoke an early warning for a potential management response (including potentially monitoring the biogenic habitat).

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[31] i.e., if footprint mapping has taken place.

[32] Even/random vs. stratified.

[33] Relocation managed by using relocatable marker tags (e.g., pins drilled into rock or driven into sediment), or GPS fixes for quadrat or transects sample points.

[34] Therefore, show measurable changes at lower levels of organic enrichment than for biogenic habitats. This does not necessarily mean that the soft sediment communities are more sensitive; rather, that soft sediment monitoring is able to measure change at lower enrichment levels, as opposed to monitoring of the biogenic habitat, because biogenic habitats can be more variable (e.g., spatial variability).

7.2.1.3.2.2 Replication

The general guidance provided for soft sediments is applicable also to monitoring of biogenic and other complex habitats. For ease of reference, it is repeated below. Other considerations specific for biogenic and other complex habitats are also discussed.

For effects-based monitoring programmes, experimental design must be such that the agreed level of effect can be detected reliably, and appropriate replication is one element of design efficacy.

Hypothesis testing (including equivalence testing) is a robust approach to determine effects in effects-based monitoring programmes. To avoid inference mistakes (i.e., Type I and Type II errors) when using this approach, it is critical to perform a power analysis and use this to determine the level of replication. Key considerations of a power analysis include a) the statistical test to be used, b) the acceptable level of uncertainty, c) the effect size to be detected[35], and d) the existing variability in the data. The existing variability in the data is specific to the site (e.g., site-specific natural variability), and the indicator/s being used to measure the effect, thus would require data from a pilot or baseline survey[36]. Appropriate stratification can be used to reduce variability and, therefore, improve statistical power.

Power analysis may also be appropriate to evaluate later stages of the monitoring programme. For example, once variability within affected community types has been established, it can also be used to check that the level of replication is still appropriate or whether changing the indicators used for monitoring is required. Importantly, we note that the replication required to reliably detect effects using hypothesis testing can sometimes be beyond practicality (e.g., when there is high variability, or for detecting a small effect size). In this case, adjustments may be made like setting a new EQS, using a different indicator, or adjusting other aspects of sampling design (e.g., frequency and/or stratification to allow for more replication effort per strata). If the required level of replication to detect the effect is prohibitive, and there are no workable and acceptable trade-offs, hypothesis testing may not be an appropriate approach.

An alternative statistical approach is to use point estimates (averages) and the associated intervals, typically 95% confidence intervals, which consider the mean and the variation for compliance determination. This approach is used by Keeley et al. (2019a), for inshore salmon farm monitoring. It should be noted that when using this approach, wide confidence intervals indicate a lack of information and should be considered a warning that the sample sizes are too small.

Whichever approach is selected (e.g., hypothesis testing or otherwise) this should be captured in the EMOP, along with the process or justifications used for determining replication.

Biogenic and other complex habitats are often patchy and/or heterogenous in nature (e.g., broad changes in habitat cover types). Accordingly, high levels of replication at the ‘sampling site’ and ‘replication’ level is generally required, although this can sometimes be reduced by adopting stratified sampling designs. The level of replication at the ‘station type’ (potential impact vs. reference), ‘station’/’transect’, and ‘replicate’ level should be determined based on variability within that level and would most likely require a pilot study for the appropriate level to be determined (see section 7.3.1 pilot study and narrative around power analysis in previous paragraph). As a general rule, greater replication of shorter transects yield relatively more robust data than fewer, longer transects within an area. It is also important to consider that replication may be indicator dependent.

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[35] For example, x% change of one or more indicators, compared with reference conditions. The effect size should be relevant to the limits of acceptable change or EQS.

[36] It is recommended that replication for soft sediment infauna sampling in the baseline surveys defaults to a minimum of n=5 (or, collect 5, analyse 3, and archive the remaining 2 in case further replication is required). Higher or lower replication may be appropriate if the variability (in space and time) of communities is known. This level of replication is best determined on a site-specific basis.

As noted for soft sediment sampling, vessel and sampling equipment drift can be substantial in deep and high current areas, and actions should be taken to control for or mitigate this where appropriate.

7.2.1.3.2.3 Sampling timing, frequency, and duration

We note that little information exists on enrichment effects manifestation timeframes in complex habitats; effects may not necessarily be closely aligned with the period of highest deposition. For monitoring focused in the far-field, fixed sampling intervals (e.g., annual) may be most appropriate, especially if multiple farms are operating without synchronised stocking/harvesting (refer Figure 5B). For the near-field, monitoring of effects to complex habitats may initially target the period of highest expected deposition (at or just after peak feed discharge). However, the most appropriate timing is likely to depend on the habitat, predicted effects, and the indicators being used for monitoring, and type of operations, so timing should be justified on a site-specific basis in the EMOP.

In a more general sense, the interval between monitoring surveys in complex habitats needs to be large enough to allow for time for change to manifest, but small enough that effects are detected early such that effective management action can be taken. For example, in habitats that have strong temporal variation (e.g., seasonal) and the risk of adverse effects is potentially high, six-monthly or quarterly monitoring may be proposed initially. However, such intensive sampling frequency should be reassessed and refined at the earliest possible time (i.e., once temporal variability has been sufficiently captured[37]) so that monitoring does not become cost-prohibitive. Monitoring frequency during the initial monitoring period should be informed by recommendations made in the baseline report (i.e., may only be appropriately determined once the pilot study and baseline survey results are available).

It is worth noting that stress manifests in different timeframes for different species (months to years) and also depends on the effect, and indicator, being monitored. These are fundamental considerations in determining when monitoring should no longer be required. The minimum duration of the monitoring programme should include multiple years (at least) and, for longer lived species, is likely to be substantially longer. Dedicated research (e.g., experimental) may be able to be used as another line of evidence to provide more certainty around effects on a shorter timescale.

7.2.2 Contamination of sediments due to the deposition of therapeutants, feed additives, antifoulants, or other potentially harmful substances used in finfish farm operations

7.2.2.1 Background and considerations for monitoring

Substances used in normal farming operations (e.g., additives in feed, stock treatments, antifoulant) could result in (sometimes persistent) contamination of sediments. At sufficiently elevated levels, this can induce ecological effects through sediment toxicity and/or bioaccumulation within marine organisms.

Those contaminants most commonly used and monitored for inshore fish farms are copper and zinc. Copper derives from antifoulant coatings, when used, and zinc is added to feed and can accumulate in the sediments from deposition of uneaten feed and faecal material. Copper and zinc deposits are more likely to accumulate at less dispersive sites where organic matter more readily accumulates in the sediments. It is noted that most finfish farms in New Zealand do not use copper anti-foulants anymore.

Chemical therapeutants such as antibiotics, antibacterials, and parasiticides have not historically been used in the New Zealand salmon aquaculture industry (Tucker 2014, Environment Guide[38]) and are not

[37] It may be possible to capture this during baseline surveys or develop a nested design where only some years have intra-annual sampling variability (if required).

[38] http://www.environmentguide.org.nz/activities/aquaculture/environmental-effects/king-salmon-and-indigenous-fish/

presently used by industry (Seafood Watch[39]). However, their use in the future cannot be ruled out. Overseas, where such additives are used extensively, the main concern is the potential for these compounds to affect non-target organisms (phytoplankton and zooplankton, sediment bacteria) and the rise of resistant bacteria and/or parasites. The fate and environmental consequence of these potential contaminants from finfish farms is not well known. It is thought that most therapeutants are water soluble and disperse and degrade readily (Ministry for Primary Industries 2013), but some may be more persistent (Hamoutene et al. 2018).

Monitoring for potential sediment contamination is expected to be site-specific, tailored to the types and amounts of potentially harmful substances used on the farm, the nature of their discharge and dispersal (i.e., source specific), and their toxicity.

7.2.2.2 Sampling methods

7.2.2.2.1 Monitoring indicators

For monitoring of zinc (an essential element in feed) and copper (historically used in antifouling coatings) accumulations, the total recoverable concentrations should be measured as a 'screening' step. Experience at inshore finfish farm sites has shown that bulk sediment recoverable metal concentrations can be variable and assessing the likelihood of environmental effects arising from the contamination can require further testing to better characterise the nature of contamination. For example, for copper from antifoulant it may be necessary to measure concentrations present in fine sediment (< 250 µm) fractions and also to use a different sample preparation method prior to analysis. This situation can be most efficiently addressed using a tiered monitoring approach where effort is minimised when it can be demonstrated that sediments beneath farms are maintained below appropriate trigger levels for each metal. A decision tree for determining which analytical techniques apply for increasingly elevated metal concentrations (if found) is provided in the Marlborough BMP (benthic) (Keeley et al. 2019a). This approach is considered to be generally applicable for monitoring of copper and zinc in an open ocean context.

Sediment metal concentrations should be assessed against the widely accepted default guideline values (DGV) of ANZG (2018); previously ANZECC (2000).[40]

Inclusion of other indicators may be required, based on the types of operational discharges specific to the farm. Where DGVs are available, these should be utilised to inform analytical testing techniques.

7.2.2.2.2 Sample collection

Grab sampling is likely to be the easiest method of collection, using van Veen grab or similar that can collect an undisturbed surficial sediment sample with sufficient depth and volume to allow for adequate sub-sampling as necessary. Sample material should comprise only surficial sediments[41] (i.e., no deeper than 3 cm into a sediment profile or as specified for comparison to guideline values).

[39] https://www.seafoodwatch.org/-/m/sfw/pdf/reports/s/mba_seafoodwatch_chinook_salmon_newzealand_report.pdf

[40] https://www.waterquality.gov.au/anz-guidelines/resources/previous-guidelines/anzecc-armcanz-2000

[41] Consistency in sampling depth over the duration of a monitoring programme is critical to ensure that observed trends reflect actual changes in contaminant concentrations and not artefacts resulting from a change in sampling depth.

7.2.2.3 Sampling effort

7.2.2.3.1 Sampling locations

Monitoring for zinc and copper need only be in the area of highest net accumulation (possibly AME and/or APA stations). Targeting these stations is also likely to be the most appropriate for sampling of other contaminants that are sourced from waste feed or faecal material, particularly those bound to inorganic particulates.

We note that 'degradation' rates for copper and zinc are much lower than for organic matter. In dispersive environments where resuspension of deposited material occurs, copper and zinc particles may travel farther than the organic matter they were originally deposited with.

Reference stations need not be sampled at the same time if background concentrations were established in the baseline survey, and if the DGVs are well above the background concentrations.

For copper antifoulant, the seabed area beneath and around the pens is typically the target sampling area. However, as noted earlier, at dispersive sites, particulates from antifoulant coatings may disperse some distance from the farm and monitoring sites may need to be positioned accordingly to capture potential (far-field) accumulation areas.

For other potentially harmful substances, locations to target for sampling will depend on the nature of the discharge source (e.g., feed, external stock treatments, etc.), the predicted dispersal and depositional characteristics, and the decay/dissolution characteristics of the contaminant in question.

7.2.2.3.2 Replication

Composite samples from the surface of at least 3 soft sediment replicates (the remainder of the surficial sediments from each replicate should be archived) may be appropriate when farm-derived contaminant concentrations are expected to be lower, and as a first screen for contaminant levels.

If levels of potential contaminants are shown to be in exceedance of established limits of potential concern (i.e., DGV), further testing could be carried out on archived replicate samples to better characterise the nature of contamination. This approach is incorporated in the decision tree in the Marlborough BMP (benthic) (Keeley et al. 2019a).

7.2.2.3.3 Sampling timing and frequency

For zinc, initial monitoring should take place in the period of highest expected biodeposition (peak biomass) during successive production cycles. Frequency can be reassessed (for example, reduce frequency of monitoring if concentrations are well below that which would cause ecological concern). If the level of deposition and accumulation is likely to be low, it may be appropriate to wait until a number of production cycles have taken place before monitoring is undertaken because monitoring early on may not detect any change because contaminants have not had time to accumulate. Alternatively, monitoring could be conducted less frequently during the initial production cycles.

For antifoulant coatings, fixed interval sampling is likely to be appropriate. For other potentially harmful substances, the best sample timing will depend on the nature of the discharge.

If sufficient monitoring information exists to indicate that the relevant contaminants will not accumulate in the sediments (i.e., are not becoming increasingly elevated in sediments over time), monitoring could be discontinued.

7.3 Other considerations for benthic monitoring

7.3.1 Pilot surveys

Pilot surveys are a field component used not strictly for data collection, but to develop, refine, and/or test sampling approaches, if deemed necessary, to ensure they are fit for purpose for baseline surveying and ongoing monitoring. Pilot surveys may be carried out before farm development or once development has commenced, depending on specific objectives. There are several reasons a pilot study might be warranted. Two reasons discussed here are (1) establishing sampling techniques, and (2) planning the sampling design. A pilot study would therefore serve to target potential effects monitoring sampling methods, sampling locations, sampling effort, and monitoring indicators. Pilot studies are most likely to be required for establishing monitoring of complex habitats in environments where there are practical constraints on some seabed video methods. There may also be other situations where a pilot study is necessary or desirable.

1. Establishing sampling techniques

In recognition of the logistical challenges that might be presented in some open ocean farming environments (high currents, depth), it is relevant to consider the practical limitations when planning robust quantitative biodiversity surveys that rely on visual methods. At deep, high current sites, the best method for obtaining quantitative information with the required precision is likely to be a ROV[42]. Sites with less challenging working conditions may be sampled with sufficient repeatability using a drop cam approach. Towed video methods may also provide quantitative (or at least semi-quantitative) information. The precision and repeatability will depend on the sampling method, and the habitats being surveyed (e.g., heterogeneous vs. homogenous), and a site-specific pilot study may be necessary prior to baseline surveying, to establish cost-efficient but appropriate sampling capability along with suitable sampling design (see next section) for ongoing monitoring.

2. Planning the sampling design

Without accurate knowledge of the distribution and characteristics of habitats that might require monitoring in and around the farming area, baseline surveys cannot be planned in detail. Furthermore, sampling design may vary depending on the equipment used for sampling (less precise sampling methods are likely to require higher replication). Without prescriptive planning, there is a risk of baseline surveys not providing the necessary information needed to support a robust monitoring programme. Pilot studies may thus be necessary in circumstances where monitoring is required, but where habitat information is not of sufficient resolution, spatial coverage, or detail.

Information on habitat distribution and variation within habitat types (e.g., changes in cover type, patchiness) is important for deciding on the sampling design approach, targeting specific locations for monitoring, including approximation of the appropriate replication needed at different sampling levels (particularly in a stratified and nested sampling design, or for habitats that are patchy in nature). Pilot studies may also be required for further characterisation of the habitats such that specific initial indicators could be determined if information from the site characterisation surveys is not of sufficient resolution for this purpose. We note there are limitations to selecting indicators when little is known about the functioning and species interactions within some communities (i.e., those that are not well studied); see note in section 7.2.1.3.1.1 in relation to monitoring indicators for biogenic habitats).

7.3.2 Depositional footprint mapping

Mapping of the organic waste dispersal and effects area associated with a farm is an effective way to obtain more comprehensive information about the spatial extent and distribution of effects at new farming sites, particularly where critical uncertainties exist in the depositional modelling, or the

[42] However, we point out that the use of some of the most capable ROVs is likely to be very expensive at the intensity recommended in the following sections. It is out of scope of the Guidelines to make determinations on what level of expense is warranted, but we note that cost will be a factor to be taken into consideration.

relationships between the deposition level and its ecological effect. For example, deposition footprint mapping can be used to confirm if biodeposition effects and spatial extent are similar to those predicted for the site. Such an exercise may be particularly useful if effects are required to be spatially constrained (e.g., if adjacent habitats require protection from effects related to enrichment), or if there is critical uncertainty around the extent of dispersal or effects. The findings can be used to refine monitoring at the site after the initial operational period (station placement, sampling effort, replication), as well as for model validation to improve confidence around the modelling[43] or environmental risk associated with further developments.

Footprint mapping should be a spatially robust design, ideally according to the predicted distribution of effects. For example, a radial sampling design with 4 transects may be appropriate where the footprint is reasonably uniform, or a gridded sampling design where the footprint is likely to be irregularly shaped. The sampling design will depend on the purpose of the exercise, the site characteristics, and what is known about the effects. However, some general considerations for footprint mapping include:

- Footprint mapping for assessing waste dispersal (including that in the far-field) should include measurement of organic matter tracers. By contrast, footprint mapping of benthic effects should focus on ecological effects indicators discussed in the relevant sections of the benthic guidance.
- Footprint mapping surveys should always be performed during the period of highest expected bio-deposition (for dispersal mapping), or effect (for effects mapping).
- Depending on the purpose of footprint mapping, it may not need to be carried out at all blocks within a farming area (provided the response is expected to be the similar across blocks).
- Ideally, footprint mapping of effects would be performed after a block was operating at approximately full capacity for 3 repeated production cycles, or a period of sufficient duration after farming operations have commenced, if the farms are operated with multiple year classes of fish. This is because effects can take some time to fully manifest.
- After footprint mapping, the EMOP should always be reviewed in light of the results.
- Footprint maps for farm-derived organic matter may not serve as good proxies for other contaminants released alongside the farm-derived organic matter if the degradation/solubility or sinking/resuspension characteristics differ from those of the farm-derived organic matter.

Footprint mapping surveys are also an excellent opportunity to refine indicators for use in future monitoring (for example, establish potential proxy indicators that are more easily sampled or cost effective). Accordingly, archiving additional sediment samples[44] is encouraged so that there are opportunities for further testing later. Footprint mapping could also be combined with a model validation exercise, if further modelling is intended to be done at the site, to inform for further development stages.

8. WATER QUALITY MONITORING GUIDANCE

8.1 Application-specific information requirements for monitoring design

Design of a monitoring programme requires a certain level of information on the environmental characteristics of the relevant area of interest and the predicted nature and spatial extent of effects of

[43] For example, using the field measurements for model validation if further modelling is needed to assess effects of further staged developments of the area. The validation would ideally be done before those further developments are modelled.

[44] For example, small amounts of surficial sediments could be sampled into sterile containers and frozen for eDNA collection (see Appendix 4).

the finfish farm operation. The technical water quality monitoring guidance provided in this section works from the basis that the following supporting information will be available to inform monitoring design:

- A description of water quality in the existing natural environment, including **spatio-temporal patterns of water quality** (including concentrations of nitrogen, oxygen, and chlorophyll, as well as associated information on phytoplankton community structure in some cases) in near-field and in far-field regions that have been identified as relevant (for example, because preliminary modelling has shown that they are likely to be within a plausible footprint of the farm).
- The **probable nature, extent, and severity of effects,** obtained from spatial modelling of the dispersal of farm-derived nutrients and resulting chlorophyll and/or particulate organic matter (e.g., using nutrient-phytoplankton or simpler tracer simulation models with transport driven by spatially explicit hydrodynamic models).

The working group considered it unlikely that finfish farming will induce serious (widespread, persistent) water quality change in open water in highly dispersive environments because of the rapid dilution and mixing of discharges in the open ocean environments likely targeted by industry. Consequently, open ocean farm sites where water quality changes could be widespread or persistent (e.g., low hydrodynamic energy, or sites connected to systems susceptible to water quality effects) have not been considered within this guidance. Accordingly, in these situations, more intensive monitoring may be required than is provided for in these Guidelines.

8.2 Guidance for monitoring

The potential water quality effects that may result from open ocean fish farming and for which technical guidance is provided, as identified in section 4, are[45]:

1. **increased** concentrations of **dissolved nutrients and** subsequent enhanced **phytoplankton** biomass and growth (section 8.2.1);
2. **reduced** concentrations of **bottom water dissolved oxygen** (section 8.2.2); and
3. **increased** concentrations of other **contaminants** such as therapeutants (section 8.2.3).

For the third potential effect (increased concentrations of other contaminants such as therapeutants and disinfectants) only high-level guidance is provided because of the inherent need for case-by-case design of monitoring based on the contaminants identified as being of concern.

8.2.1 Increased concentrations of dissolved nutrients and subsequent enhanced phytoplankton biomass and growth

8.2.1.1 Background

Nutrient enrichment resulting from finfish farming can promote eutrophication. Finfish release ammoniacal nitrogen through their gills and urine, and nitrogen (N) is also released in faeces and waste feed. Faeces and waste feed also contain phosphorus (P) (mainly particulate), but little (if any) silicon (Si). Nutrient (notably, dissolved inorganic nitrogen) concentrations are one of the key determinants of marine phytoplankter growth rates. In turn, individual phytoplankter growth rates influence population growth rates and the potential for phytoplankton biomass to accrue. Nutrient introduced via finfish farms may therefore lead to greater phytoplankton growth and biomass. Furthermore, the nutrient release may stimulate changes in the taxonomic composition of the phytoplankton community because the molar N:Si and P:Si ratios within the nutrients that finfish farming generate are much lower than those naturally present in the sea and this difference favours different phytoplankton species to

[45] See Appendix 2 for a description of the process followed by the working group to identify these potential effects.

proliferate. However, it is considered unlikely that fish farm nutrient inputs are deemed large enough to materially change the N:Si or P:Si ratio of the water passing through. Some types of phytoplankton can be harmful at high concentrations.

When monitoring is carried out to assess the risk of eutrophication, it is important that the monitoring design considers the possibility of symptoms consistent with farm-induced eutrophication arising through entirely different causes (e.g., upwelling of nutrient-rich, oxygen-poor deep water across the continental shelf, unusually stable weather conditions that promote stagnation of the water body). It will usually be difficult to confidently determine how much influence farming activities have had upon any individual manifestation of eutrophication symptoms.

Regarding toxic effects from ammonium build-up, it is almost inconceivable that ammonium concentrations would reach toxic levels even within the pens (also, if they did, the farmed fish would suffer). Ammonium released by the fish will gradually (in some cases rapidly) be incorporated into living matter and then into dead organic matter before recycling. We therefore provide no guidance on monitoring potential ammonium toxicity effects.

8.2.1.2 Monitoring design

The key indicators of nutrient enrichment are total nitrogen (TN), particulate nitrogen (PN) and carbon (PC), phytoplankton biomass (measured as chlorophyll *a*), and, for context (because they are not indicators of farm effects), temperature and salinity. Algal composition (including harmful algae) and nutrient ratios of (P:Si and N:Si) may also be appropriate indicators to monitor if there is a risk of key nutrient ratio (and therefore phytoplankton composition) shifts. The application of these indicators in monitoring is described in the following paragraphs and summarised in Table 3, along with sampling methods.

Additional information on measuring TN, PN, and chlorophyll *a* is provided in Appendix 7. In addition to the indicators in Table 3, other indicators may also be relevant, depending on the environmental risks specific to the site and the objectives of the monitoring. Specifically, it may also be useful to monitor solute nutrients of nitrogen and phosphorus separately (nitrate-N, ammonium-N, dissolved phosphorus), as well as dissolved reactive silica because this can dictate phytoplankton community dynamics.

It is generally recommended that any monitoring should be targeted towards persistent (i.e., seasonal scale) change rather than shorter events. High-level guidance on other elements of monitoring design (e.g., locations, frequency), and baseline data collection, is described in Table 4.

Tracing nitrogen plumes

TN provides is a measure of the overall nitrogen abundance and includes almost all[46] the nitrogen species into which ammonium may cycle. We anticipate that the detectable ammonium and TN increments generated by a fish pen may extend further from an open ocean pen than from a typical inshore fish pen (because the open ocean is expected to be more dispersive, and pens are likely to be larger and carry more fish). The ammonium increment will likely become undetectable more rapidly than the TN increment and TN will therefore be a better tracer of the far-field fate of farm-derived nitrogen.

[46] Only the comparatively inert nitrogen gas (N_2) is excluded.

Table 3: Recommended indicators of nutrient enrichment, phytoplankton biomass, and associated sampling methods. Appendix 7 provides more detailed information on biogeochemical parameters and analytical procedures for water quality monitoring. Note: sampling methods and indicators do not necessarily match horizontally.

Indicators and broad recommendations	Sampling method
Total nitrogen (TN) is recommended as a tracer of the fate of farm-derived nitrogen. Measurement of TN will also capture urea and particulate N stemming from the fish farms. Ammoniacal should also be considered in the near-field to provide information on the possible source of TN. Particulate nitrogen (PN) and particulate carbon (PC) are recommended as indicators of particulate organic matter if monitoring is required because it is anticipated that phytoplankton proliferation might arise in the vicinity of sites sensitive to such environmental change. We recommend that chlorophyll (chlorophyll *a*) be adopted as the primary proxy of phytoplankton abundance. It is comparatively easy and cheap to measure. A case-by-case assessment will be required to decide whether monitoring of harmful algal species will be necessary. In the unlikely event that fish farm nutrient inputs are deemed large enough to materially change the N:Si or P:Si ratio of the water passing through, there may be justification for monitoring phytoplankton taxonomic structure (including harmful algal species) upstream of the pens and at some distance (one-two days of travel time) downstream of the farm. We anticipate that this would be a relatively short-term exercise (e.g., a few months (perhaps, repeated over a few years) as an extension to routine monitoring, or a few days in an intensive study). It is unlikely to be an indefinitely repeated exercise. Water temperature and salinity may provide helpful contextual information for interpreting monitoring results.	Water samples should be collected from surface water and at depth(s).[47] Water sampling using a surface grab or depth integrated sample for surface water (10–15 m) and sampling open-closing bottle for sampling at depth. Needs sufficient volume to allow for adequate sub-sampling, as necessary. Chlorophyll *a* data may be collected using in-situ fluorometry. In this case, appropriate calibration of instruments must be ensured. See section 8.3.4 for considerations on continuous *in situ* monitoring. Under some circumstances, it can be measured remotely using satellite imagery (see Appendix 6). Towed fluorimeter data could assist with assessing the patterns of spatial variability. Sampling may be challenging because of strong currents and large waves (see Appendix 5). Sampling from deep within the water column will be especially difficult close to the pens.

Monitoring phytoplankton structure and biomass

The phytoplankton community comprises many species. Although individual cells can be counted and identified and sized under the microscope (and weight calculated), this is a time-consuming, manual (expensive) process[48]. Thus, it is common practice to measure chlorophyll[49] concentration as a proxy of total phytoplankton biomass. The phytoplankton taxonomic structure (including harmful algal species) may need to be monitored at times; for example, if a risk of harmful algae blooms has been identified. It may be possible to interrogate other sources of information regarding harmful algae and

[47] The actual depth of bottom water sampling requires site-specific assessment due to the potentially very deep waters in open ocean environments.

[48] Automated flow-through systems with accompanying image analysis software have recently become available but they remain expensive to purchase and operate at present. No doubt these systems will become cheaper and better suited to routine monitoring in the future.

[49] Chlorophyll is a photosynthetic pigment that is present in all phytoplankton. It is comparatively easy to measure in the laboratory. Chlorophyll concentration provides only a crude indication of total phytoplankton biomass because the cell-specific chlorophyll:biomass ratio varies amongst taxa (and even within individual cells through time).

the symptoms they produce (e.g., on-farm monitoring done by the consent holder, shellfish safety monitoring data, public health records of shellfish poisoning events).

Table 4: Considerations for sampling locations, timing, and duration of monitoring for baseline data gathering and monitoring increased concentrations of nutrients and phytoplankton biomass after farm operations have commenced.

Phase of programme	Sampling locations and considerations for timing and duration of monitoring
Baseline data gathering	Locations need to be selected on a site-specific basis. They should include any areas that have been identified as potentially impacted areas. Ideally, they will also include reference areas that have been identified as 'usually upstream' and 'usually downstream'. Reference upstream and potentially far-field downstream stations should ideally be beyond the extent of the tidal excursion perimeter.
After farm operations have commenced	We consider that it is generally not necessary to undertake one-off or repeated fine-scale spatial mapping. Instead, we consider it likely that such monitoring may only be required at specific development stages and/or to assess effects on areas deemed sensitive to nutrient enrichment. If the decision to grant a consent was strongly influenced by nutrient-dispersal modelling, it might be appropriate to undertake a one-off detailed nutrient mapping exercise to verify the model predictions, including the predicted plume.[50] Such a study may be useful if it is undertaken after farming has reached moderate scale but before it reaches full scale. For example, it might be appropriate to verify model predictions before permitting expansion to full scale. If monitoring is required at areas sensitive to nutrient enrichment far from the farm, attributing cause-and-effect will be challenging.[51] If monitoring of nitrogen is required far from the farms, it will be appropriate to monitor TN instead of solute inorganic nitrogen. We note that until the farm has grown large enough to make it likely that the footprint will reach sensitive areas, there is little value in monitoring these areas (other than to improve the respective baseline datasets). While monitoring is required, water samples should be collected monthly by default, but collections may be dependent on the nature of the sensitive area (more or less frequent sampling, e.g., in selected periods of the year, may be appropriate).

If monitoring is required because it is anticipated that phytoplankton proliferation might arise in the vicinity of sites sensitive to such environmental change, we recommend that particulate nitrogen and particulate carbon are monitored (as measures of particulate organic matter that include both living and dead phytoplankton).

Any phytoplankton proliferation that is stimulated by farm-derived nutrients is likely to manifest itself downstream of the farm. Oceanic conditions are likely to fluctuate in space and time. Unless flow

[50] If it is deemed necessary to try to verify the performance of a simulation model of tracer dispersal, an intensive mapping exercise may be warranted but this should be delayed until the farm has grown large enough that it could conceivably generate a measurable far-field nutrient plume. We see no value in undertaking repeated intensive surveys to map the farm plume. If tracer dispersal modelling indicates that nutrients are likely to penetrate into areas particularly sensitive to nutrient enrichment (e.g., bays, reefs), these may require monitoring from the outset of baseline monitoring.

[51] To help determine whether any incremental TN stems from the farm or elsewhere, it would be beneficial to also monitor at a location (or locations) 'usually upstream' of the farm.

patterns and biological processes are very stable through time, it would be difficult to predict where farm-derived proliferation is most likely to arise on any specific occasion. Phytoplankton proliferation is a gradual process that is influenced by many factors, including processes unrelated to aquaculture (e.g., favourable weather conditions promoting phytoplankton growth).

If a phytoplankton bloom is ever detected, it will often prove difficult (even impossible) to unequivocally determine the cause of an individual proliferation event. However, changes in the frequency, duration, location, and spatial extent of proliferation events that are characteristic of those expected to arise as a result of fish farming might be evidence that finfish farming is having an effect. If the changes are larger than those anticipated for the scale of finfish farming, this might be evidence that the initial modelling underpredicted the nature of nutrient enrichment. Determining whether the frequency, duration, extent, etc. of proliferation events has changed demands robust and appropriate levels of baseline data and long-term monitoring data.

Satellite imagery is emerging as a tool for measuring chlorophyll in coastal and ocean waters. Satellite sensed imagery now stretches back 20 years or more. Near-surface water quality properties (chlorophyll concentration, turbidity, temperature) can be inferred from these images. Appendix 6 discusses potential applications of satellite data in monitoring effects of open ocean aquaculture. We believe that there is an opportunity to explore ways in which satellite data could be used to assess broad-scale changes in chlorophyll in the waters around open ocean finfish farms and contribute to the monitoring of effects of these farms on phytoplankton biomass. In addition to monitoring, of particular interest is that, once calibrated, historical satellite imagery could be interrogated to determine its use for establishing baseline information, including inter-annual and inter-seasonal variability as well as characterising naturally occurring events, such as phytoplankton blooms. To our knowledge, satellite imagery has not yet been used for consent-related monitoring of aquaculture, although it has been applied in a research context to determine the influences of large mussel farming areas on near-surface chlorophyll and water temperature (Pinkerton et al. 2018).

8.2.2 Reduced concentrations of bottom water dissolved oxygen

8.2.2.1 Background

Oxygen[52] is consumed by fish directly and through the decomposition of organic matter, including phytoplankton and finfish farm waste, in the water column and the sediment. High levels of oxygen depletion can be harmful to aquatic organisms. Oxygen consumption by the farmed fish themselves may reduce dissolved oxygen concentrations at the farm and farmed fish are likely more sensitive to reduced oxygen than their surrounding environment (Vaquer-Sunyer & Duarte 2008). This means that finfish farmers will have strong motivation to ensure that oxygen levels remain adequate inside the pens. We believe that there will usually be no need to monitor oxygen inside or close (horizontally) to the pens for the purpose of assessing effects on the environment (as opposed to the fish themselves) from oxygen reductions resulting from open ocean finfish farming.

Potentially more relevant for open ocean aquaculture may be bottom water reduction of oxygen due to high oxygen uptake from enriched sediments.[53] Replenishment of oxygen is more difficult at depth because there is less exchange with the atmosphere and because in low light intensities algae don't produce oxygen. The need for bottom water oxygen monitoring is likely determined as part of the benthic effects assessment. If a risk of excessive organic sediment enrichment is identified, a trigger point could be specified based on benthic effects that, if reached, would require bottom water oxygen monitoring.

[52] We use the term oxygen to refer to dissolved oxygen.

[53] Excessive organic enrichment of sediments and its subsequent metabolism creates significant oxygen demand in the sediments. Oxygen is replenished by uptake from the water overlying the sediment, which can reduce oxygen in overlying waters.

8.2.2.2 Monitoring design

Oxygen concentration and oxygen saturation state of the bottom water are recommended as indicators of reduced bottom water oxygen (Table 5). Here, the focus is less on low DO generated by direct fish respiration but more on bottom water oxygen reduction driven by decay of organic matter in the sediment. Recommended sampling methods are summarised in Table 5. Table 6 presents considerations for sampling protocols.

If it is decided that oxygen monitoring will be undertaken, it will be useful to align with benthic monitoring and possibly monitoring of phytoplankton to obtain integrated data relating to biological and chemical activity involving oxygen transformation processes. Sampling protocols for bottom water oxygen will depend on monitoring objectives. One of the key considerations with respect to adverse effects from oxygen depletion is the duration of depletion events. If monitoring aims to describe such events, high-frequency monitoring or spot checks during high-risk periods may be most appropriate. If monitoring aims to describe general patterns of oxygen concentration, monthly monitoring may be appropriate but near-bed sensors collecting high-frequency time-series data would provide added value and better describe patterns in oxygen concentration.

Table 5: Recommended indicators of reduced bottom water oxygen and associated sampling methods. Appendix 7 provides more detailed information on biogeochemical parameters and analytical procedures for water quality monitoring.

Indicators and broad recommendations	Sampling method
Oxygen concentration and oxygen saturation state of the bottom water	Oxygen profiles can be measured using conductivity, temperature, depth (CTD) casts interfaced with a dissolved oxygen sensor.
Water temperature and salinity may provide helpful contextual information for interpreting monitoring results.	Near-bed sensors could be moored to the seabed to provide high-frequency time-series data. These sensors would have to be serviced regularly (also see considerations described in section 8.3.4). Vertical profile data could be collected during the service trips.

Table 6: Considerations for sampling protocols for monitoring reduced concentrations of bottom dissolved oxygen.

Phase of programme	Sampling locations and considerations for timing and duration of monitoring
Baseline data gathering	Locations need to be selected on a site-specific basis. They should include any areas that have been identified as potentially sensitive to, or experiencing, reduced oxygen concentrations. Therefore, the selection of locations is most likely to be informed by benthic effects assessment. Ideally, they will also include areas that have been identified as outside the predicted farm footprint.
After farm operations have commenced	Oxygen may need to be monitored at locations identified as sensitive to reduced oxygen concentration, triggered by benthic effects monitoring, or the assessment of benthic effects from organic enrichment in the AEE, as well as at reference stations. Timing and duration of monitoring will depend on the monitoring objective. If monitoring aims to identify intensity and duration of depletion events, continuous monitoring, high-frequency monitoring during high-risk periods, or spot checks during high-risk periods may be required. If monitoring aims to describe general patterns of oxygen concentration, monthly monitoring may be appropriate.

To assist with the interpretation of monitoring results, there may be value in monitoring oxygen at a small number of stations outside potential impact areas, e.g., outside the farm footprint. This would provide information on natural variability. Monitoring water temperature and salinity also provides helpful contextual information.

Until farming has reached an intensity that makes it likely that bottom water oxygen could plausibly be notably reduced, the primary value in monitoring bottom water oxygen is in improving the baseline dataset (and, if a trend of falling bottom-water DO becomes apparent, gaining advance warning of a possible future issue).

8.2.3 Increased concentrations of other contaminants such as therapeutants and disinfectants

8.2.3.1 Background

A variety of therapeutants may be used on marine farms for general health of the fish, or as treatments for specific concerns, such as parasites or infections. At present no therapeutants are known to be used in the New Zealand finfish farming industry, though they may be used in future. Other chemicals used on the farm may include disinfectants, anaesthetics, and general farm cleansers. All these chemicals are designed to be biologically active. Some may be harmful to aquatic organisms above certain concentrations, although their ecological toxicity is not always known.

8.2.3.2 Monitoring design

Monitoring of contaminants (or their effects) identified as being of concern and requiring monitoring will need to be designed on a case-by-case basis (Table 7). In some circumstances, eco-toxicological studies may also be appropriate.

Detection limits will differ across contaminants. It is possible (even probable), that it will prove impossible to reliably detect some (or many) of the contaminants except very close to the pens in the immediate aftermath of treatments. Where these substances are detectable in the receiving waters, it is possible that the combination of difficult field sampling conditions and delayed laboratory analysis may make comparisons with toxicity levels very difficult. Table 8 presents high-level considerations for sampling protocols.

Table 7: Recommended indicators of contaminant accumulation and associated sampling methods.

Indicator	Sampling method.
Contaminant-specific measures of concentration or activity	Laboratory analyses of water samples. Or analysis of contaminants that have accumulated onto appropriate time-integrating collection surfaces (specialised gel strips, etc.). Eco-toxicological studies may also be appropriate for some specific applications.

Table 8: High-level considerations for sampling protocols for baseline data gathering and monitoring of contaminant accumulation.

Phase of programme	Sampling locations and considerations for timing and duration of monitoring.
Baseline data gathering	If it is expected that baseline concentrations are above detection limits, baseline data should be collected at the farm and at areas sensitive to effects from the contaminant.
After farm operations have commenced	Monitoring locations should focus on areas where accumulation is expected to be greatest and on areas that are sensitive to effects from the contaminant. Timing and duration of monitoring is contaminant specific and should be informed based on how the substance is applied and when it used on the farm. One-off investigations may be sufficient instead of routine/ongoing monitoring.

8.3 Other considerations for water quality monitoring

8.3.1 Water quality baseline data

For water quality baseline data repeated sampling events will probably be needed due to high temporal variability in most marine systems. As a default, data collected by discrete water sampling should be collected at least monthly, and the baseline should extend over at least one year to reflect inter-seasonal variation. Other forms of monitoring, such as continuous *in situ* data collection from moored buoys, should also be considered (see discussion in section 8.3.4). Sufficient data should be assembled to characterise season-specific median conditions and elucidate at least some of the within-season variability and some of the natural spatial variability inside the anticipated farm footprint and in upstream areas and downstream areas outside the anticipated farm footprint if there are likely to be environmental gradients. In instances where there is high natural variability or temporal variability is not known, several years of baseline data may be required. This need not preclude farm installation or development of initial EQSs, though it is advisable to refine any initial EQS once natural variability has been established to a level that is meaningful for the intended EQS and associated monitoring indicator(s).

The baseline dataset should at least include all water quality indicators for which EQS have been (or are anticipated to be) defined. If areas of particular sensitivity to predicted effects have been identified, they should be included within baseline surveys.

Historical satellite data could be interrogated to determine whether it will be useful in establishing a baseline and as a part of ongoing monitoring (see discussion in Appendix 6).

8.3.2 Determining near-field and far-field perimeters

This section provides additional consideration regarding the glossary definitions of the terms near-field and far-field. The near-field is the region within which there has been insufficient time for meaningful quantities of farm-derived materials to be transformed into different forms by biogeochemical and biophysical processes (e.g., ammonium-N derived from the farm is still ammonium-N). Conversely, the far-field is the region in which the state(s) of farm-derived materials is overwhelmingly determined by biogeochemical and biophysical transformations that have taken place after release from the pens (e.g., ammonium-N derived from the farm has transformed into other forms, like nitrate-N, or been taken up by phytoplankton). In reality, there are no sharp distinctions between these areas because they blend into one another, but for the purposes of this discussion, we define the outer perimeter of the near-field as being the perimeter at which 'parcels' of farm-derived material retain a fraction (α%) of their original state. Conversely, we define the inner perimeter of the far-field as being when 'parcels' of farm-derived material retain (100-α)% of their original state. There is no formally agreed numerical value for α, but we suggest 80% may be a good estimate of when no meaningful quantities of farm-derived material have undergone transformation (and a figure of 20% is consistent with the expectation that the state of the farm-derived material in the far-field is dominated by transformations that have occurred after release).

Processes such as nutrient uptake by phytoplankton, microbial break-down of organic matter, and dilution by eddy mixing operate on timescales of hours-to-days. Thus, it is not unreasonable to adopt a characteristic specific transformation rate for parcels of farm-derived material of around 0.1-1 d^{-1}.[54]

Having selected 'degree of transformation' thresholds, it becomes comparatively easy to define the downstream distances corresponding to 'a few hours' and 'a few days' travel distance – at least when

[54] Choosing 0.5 d^{-1} implies that parcels of farm-derived material will be $T_{inner}=\log_e(0.8)/0.5= 0.45$ d old (i.e., a few hours old) when they reach the 80% of initial state threshold and $T_{outer}=\log_e(0.2)/0.5=3.2$ d old (i.e., a few days old) when they reach the 20% of initial state threshold.

(a) currents are constant in space and time[55] and (b) the magnitude of eddy mixing activity is well characterised. Fundamentally, eddy mixing tends to weaken the correlation between downstream distance and downstream age. The stronger the eddy mixing is relative to the larger-scale prevailing current, the weaker the distance/age correlation will become. Thus, it becomes more difficult to reliably distinguish between near-field and far-field.

Fluctuations in the direction and magnitude of the prevailing current (for example, those driven by tidal oscillations of fluctuating winds) will also serve to weaken the correlation between distance-from-farm and age-from-farm. For example, where tidal oscillations are sufficiently strong (relative to longer-term residual currents), some of the water that passes through a pen on one tide may re-enter it on one or more subsequent tides (such that it now contains a broader spread of material ages) before ultimately being persistently washed further afield. When tidal flows or wind-driven flows are not well aligned with the longer-term average flow, they will also serve to introduce the plume of farm-derived material into areas that it would not otherwise reach and further mix water parcels of differing ages.

Ultimately, the definition of the outer perimeter of the near-field and inner perimeter of the far-field will be a somewhat subjective choice. Numerical models (particle-tracking models incorporating large-scale persistent currents, tidal currents, and wind-driven flows, etc. and tracer-decay (tracer-transformation) could be used to identify these perimeters. As noted above, if one chooses to accept our guideline values for critical thresholds of transformation and specific transformation/decay rate, the near-field perimeter will be located at about 0.4–0.5 day's travel time from the pens whereas the inner perimeter of the far-field will be located at about 3–4 days' travel time. When transport is dominated by a current that is stable (in both time and space) across timescales substantially greater than the largest of these figures, it will be possible to reliably distinguish between the far-field and the near-field. When transport is dominated by currents that fluctuate on timescales shorter than several times 3–4 days (whether wind-driven flows or eddy mixing, etc.), it will be difficult to reliably distinguish the so-called near-field from the so-called far-field because materials/waters of differing age will be in close spatial proximity to one another.

Conceptually, the far-field wraps around the world. For practical reasons, it will be necessary to define an 'outer perimeter for monitoring' beyond which monitoring is deemed impractical or inappropriate. Like the other perimeters, this will need to be determined on a site-specific basis. Factors that influence the location of this perimeter might include:

a. the values people associate with habitats in the far-field;
b. the quantum of harm that habitats may suffer from farming activities;
c. the probability that this harm will arise;
d. instrument detection limits;
e. existence of other causes of spatio-temporal variability which will make it difficult to distinguish farm-induced change from 'natural fluctuation'; and
f. health and safety, etc. constraints associated with operating in regions that may be even more remote than the farms.

8.3.3 Identifying monitoring locations

Site-specific considerations and monitoring objectives will influence where (relative to the farm) monitoring should be undertaken. Monitoring should focus on areas where environmental change of

[55] Note that the outer perimeter of the near-field and the inner perimeter of the far-field both shrink towards zero as the current speed falls to zero.

greatest ecological importance is predicted.[56] Where there are multiple distinct areas predicted to experience similar environmental change (i.e., if the predicted farm footprint comprises several unconnected areas), it may be necessary to monitor them all or representative ones.

It will usually be appropriate to include reference sites at locations considered to be 'usually outside the footprint' to assist with distinguishing farm-induced change from natural variability. Sites located in the 'usually upstream of the residual flow' direction are likely to be particularly useful as reference sites. To be useful, upstream and downstream stations should be (at least) beyond the extent of the tidal ellipse and, preferably, a few days' travel time for the plume away from the farm along the axis of the long-term residual current (i.e., well into the far-field).

If monitoring is intended to confirm predictions, support iterative improvement of understanding of effects, or for development of EQSs, it may be appropriate to include monitoring stations located along the gradient of anticipated farm footprint intensity.

8.3.4 Continuous *in situ* monitoring

This section draws on considerations for the inclusion of continuous monitoring platforms provided in best management practice guidelines for water quality monitoring of salmon farms in the Marlborough Sounds (Elvines et al. 2019) and extends them to open ocean environments.

The integration of continuous *in situ* monitoring platforms increases the temporal resolution of sampling for parameters like chlorophyll *a* and dissolved oxygen. Such platforms are not a complete alternative to taking physical water samples because of the need for calibration of sensors. They also require moderately frequent servicing (e.g., cleaning, battery replacement). The inclusion of these types of platforms into monitoring programmes needs be done with careful consideration, including how the data will be analysed and used, particularly in relation to any EQS.

Continuous monitoring platforms require structures to keep them in place. For measurements near finfish pens, existing farm structures may be utilised; however, for most monitoring locations separate buoys are required. Installation and maintenance of buoys in open ocean environments is possible but can be challenging given the dynamic conditions and requires sound engineering of the mooring array. These challenges, and the expense of continuous monitoring platforms, may limit the number of monitoring sites that can be established and therefore the spatial resolution of these data.

Another limitation of continuous monitoring platforms is the availability of sensors for measuring indicators relevant to finfish aquaculture. Many sensors are still in their infancy (e.g., marine nitrate sensors and *in situ* cytometric phytoplankton cell count technologies) and there is a need for further research-led testing and validation before these sensors can be used in aquaculture monitoring programmes. As a consequence, at this time, only a limited number of relevant water quality parameters can be measured using continuous monitoring platforms. For example, although it is possible to measure temperature, salinity, pH, dissolved oxygen, turbidity, and chlorophyll *a*, it is not yet possible to reliably measure nitrogen or properties such as TN, PC, and PN.

[56] Care must be taken if simple tracer modelling (e.g., dispersal and decay of farm-derived nitrogen) is used to infer potential influences upon phytoplankton, chlorophyll, or incremental oxygen demand. It is inevitable that the tracer (nitrogen) concentration will be greatest close to the pens – but in that area, it is likely to be present as ammonium. It will take time for the ammonium to be incorporated into living matter and detritus that can generate an oxygen demand. If monitoring is to defend against e.g., algal blooms or excessive oxygen demand in high-value habitat, there may be situations where it is better to monitor more distant (several days' travel time) high-value areas in preference to closer ones that exhibit greater 'incremental nitrogen'.

9. REPORTING

9.1 Baseline data

Baseline data reports (whether collected during descriptive or dedicated baseline surveys) should contain all results from the survey(s), including sampling station coordinates, methodology, raw data, and any recommendations or considerations for ongoing monitoring at the site. Reports should be completed and disseminated to the council and other parties (as required) before the EMOP is finalised.

9.2 Routine monitoring

There should be regular preparation of monitoring reports that summarise the findings of routine monitoring performed at the site. We recommend the following information is included in the reports:

- Brief site history, farm installation date, feed history (ideally monthly feed data);
- Recommendations from the previous monitoring and compliance reports, where relevant, and discussion of these within the report as necessary;
- Comprehensive sampling details for the monitoring described in the report;
- A description of the seabed and water quality state at any monitoring stations;
- If required, an assessment of whether the seabed and/or water quality conditions met the associated EQS (if any) and environmental management goals (if any) and note any other important findings with respect to adverse effects (e.g., unexpected effects);
- A comparison of results to historical baseline surveys and (if available) earlier monitoring data;
- A timeline showing the monthly feed discharge against the survey timing, as well as any other relevant site-specific information;
- Recommendations for management practices to ensure compliance with the EQS, as necessary; and
- Recommendations or considerations for future reviews of the management or monitoring plan.

Reporting timeframes of monitoring results will depend on monitoring needs. Options include reporting on an annual basis or aligned with completion of production cycles, depending on the farm operational schedule and adaptive management regime. It may also be that different reporting schedules are required for different monitoring aspects within the farming area if monitoring components are not sampled on the same timescale or at the same time intervals. For example, broad-scale monitoring may be synthesised and more comprehensively analysed over a longer period (multiple years), rather than annually. Such a review could align with key milestones such as development staging reviews. Site-specific reporting timeframes would ideally be stipulated in the EMOPs or EMAPs, rather than in consent conditions.

9.3 Supplementary investigations

Supplementary investigations performed should be presented in standalone reports, shared with the council and other parties (as required) and the relevant outcomes should be considered (and integrated if appropriate) in subsequent reviews of the EMOP.

PART C: Environmental monitoring plans, quality standards and the role of other environmental information sources

10. ENVIRONMENTAL MONITORING PLANS

10.1 Developing an EMOP

An overview of the development process for an application-specific EMOP and the documents that inform the EMOP are shown in Figure 6. The key information sources feeding into the content of the first EMOP are the AEE, baseline survey report(s), the EMAP (if prepared separately), and technical monitoring guidance from these Guidelines. Other information may be available to inform the design of EMOPs, such as research findings or SOE monitoring results (not shown in Figure 6).

As illustrated in Figure 6, the decision-making process for determining whether monitoring is required in relation to a potential effect involves a wide range of factors outside the scope of the Guidelines, including the policy, operational and environmental context, community expectations, and application-specific information.

Although the EMOP should be drafted by independent suitably qualified scientists, its development should involve engagement with industry and resource management planners to ensure that monitoring is well integrated in the wider operational and resource management context to maximise its effectiveness.

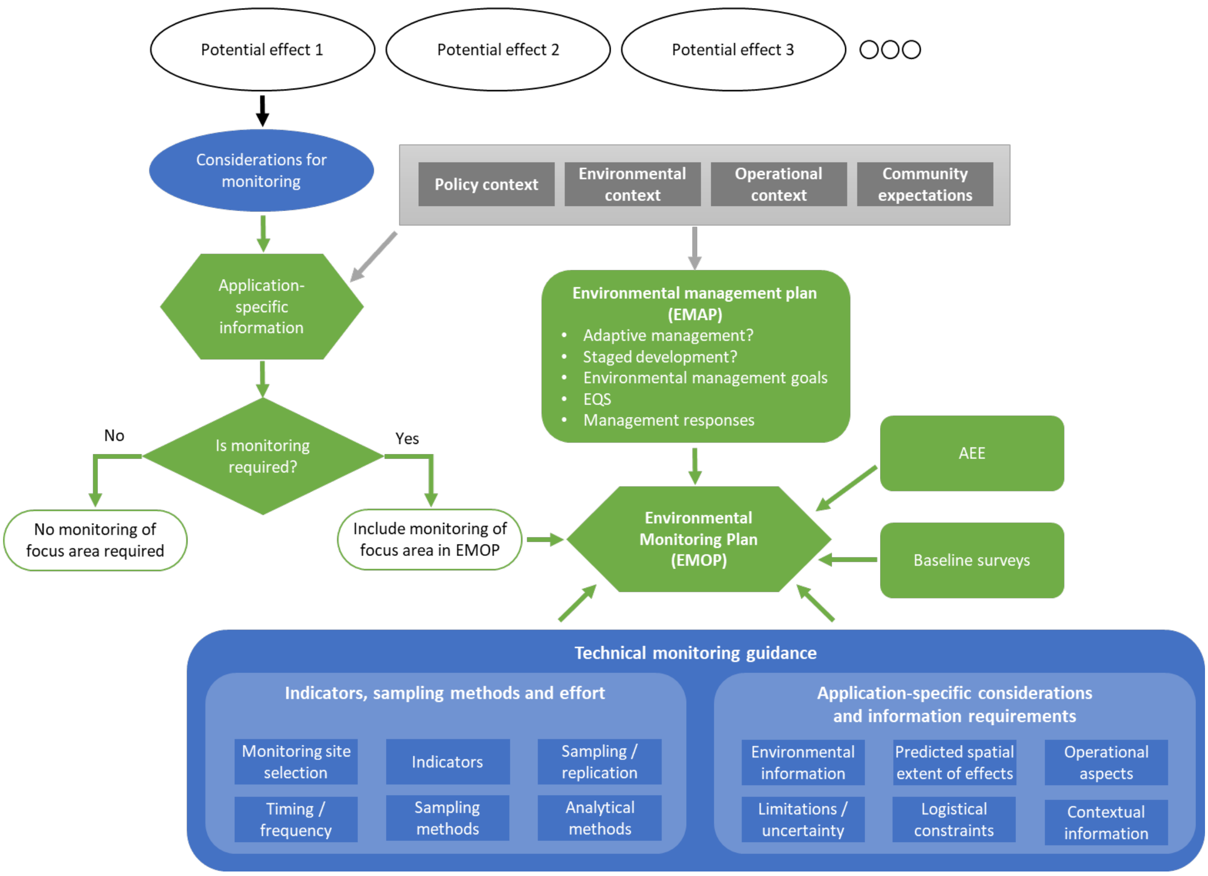

Figure 6: Illustration of how the technical guidance presented in the Guidelines (blue shapes) is intended to be implemented to inform the development of EMOPs. Green shapes represent components of the consenting and (adaptive) effects management processes. Grey shapes represent broader considerations that inform application-specific decision making.

10.2 Monitoring objectives

Monitoring objectives form a fundamental aspect of monitoring programme design, and it is important that these are set out clearly so that monitoring is targeted and effective. Developing monitoring objectives requires information about the practicability and resourcing requirements of methods and it is therefore envisaged that the technical guidance is used to inform the development of monitoring objectives. However, most importantly, monitoring objectives must be aligned with environmental management goals, EQSs, and (adaptive) farm management strategies.

The scope of the monitoring guidance provided in the Guidelines is relatively broad. It is primarily applicable to monitoring objectives related to:

- Detecting adverse effects of the marine farm and comparison with EQSs; and/or
- Confirming predictions of the nature, extent, and temporal scale of effects.

The guidance may also be useful for collecting environmental data that will allow EQSs to be developed or improved in the future.

10.3 Content of the EMOP document

As a minimum, the EMOP should include the following information:

- Site name, operator name, consent number;
- Relevant monitoring conditions from resource consent;
- Species of finfish, permitted biomass/feed discharge levels, permitted medicines/chemicals;
- Diagram showing pen configuration and locations, modelled impact area, allowable mixing zone, transects, and sampling stations;
- Rationale behind selection of transects and sampling stations;
- Proposed timing for sampling;
- Parameters sampled at each station and collection and analytical method(s) to be used;
- Detail of sample handling and preservation techniques;
- Data analysis approach (full data analysis methods should be provided with the monitoring results themselves);
- Reporting of monitoring information (timing, report content); and
- Review timeframe for the EMOP, and a description of the process for instigating and carrying out that review (see section 10.5).

10.4 EMOP Implementation

The implementation process of an EMOP will largely depend on consent requirements. We reiterate that, as discussed in Part A of the Guidelines, monitoring is an integral process in the management of environmental effects and EMOPs therefore need to be well-aligned with the relevant RMA processes as well as with any (adaptive) management frameworks used to manage effects of an open ocean finfish farm.

10.5 EMOP review

10.5.1 Establishing a process for EMOP review

A process for reviewing the EMOP should be established through collaborative engagement, and then defined in the EMOP itself. The appropriate review period for the EMOP may be stipulated in consent conditions or the EMAP and should be specified in the first version of the EMOP. Review of the EMOP should be aligned with a review of the EMAP, to ensure the related elements between these documents (EQSs, management response framework, etc.) remain aligned. The timeframe for review should reflect the level of uncertainty/risk associated with the site and farm operation. Ideally, the EMOP would not be reviewed until a reasonable amount of monitoring data has been gathered to inform an effective review. For example, an initial period of operation could encompass three production cycles at one site prior to a review being undertaken. Reviews may also be conducted out of schedule of the expected review period, for example (1) if there is a change or proposed change to farming operations that needs specific consideration within the EMOP; (2) if fundamental methodological changes are proposed; (3) if there is substantial new information that necessitates earlier review; or (4) if EQSs need revising.

Reviews should be performed through an independent science provider and should involve consultation with external technical experts and, depending on the scope of the changes, other stakeholders as necessary. External peer review of the EMOP that covers the initial operational period is also recommended; ideally this would include experts familiar with relevant monitoring programmes overseas to provide a broad range of experience and perspectives. EMOP reviews will generally require certification by the council but it may be advantageous (and possibly necessary) to engage with the council during the review process to seek their input earlier in the process.

We recognise that there are information gaps in some aspects of science that, if filled, would be useful to inform and improve monitoring approaches. Accordingly, reviews of the EMOP should involve reviewing new information of relevance to the monitoring programme. The revised EMOP should incorporate this new information, including research findings and technical innovations. Reviews should also take account of wider environmental changes (natural and resulting from other anthropogenic activities) that might warrant changes in the monitoring approach, EQSs, data interpretation, or other aspects.

10.5.2 First review recommendations

After the first operational period of the farm, a substantial amount of new knowledge will exist on the effects of the farm. These data will comprise robust information upon which to propose refinements of monitoring design and EQSs, but the review should not be limited to these aspects. The key considerations for the first review will be site-specific, depending on the uncertainties, risks, and information gaps relevant to the site. Each site-specific EMOP should explicitly list those key points to be considered in the first review.

Examples of what aspects may require review include:

- Refinement of monitoring design (sampling effort, replication, station placement);
- Evaluation of indicators and recommendation of new/substitute indicators (if comparative data are available);
- Refinement of key taxa to target for monitoring complex habitats (benthic monitoring);
- Development or refinement of EQSs;
- Evaluation of the effectiveness or monitoring in meeting monitoring objectives and environmental management goals; and
- Evaluation of the specifics of, and need for, ongoing monitoring (including deciding if the monitoring performed to date sufficiently characterises the risk/effect, such that monitoring can be discontinued).

We note that some of the above will be best informed using findings of the research or supplementary investigations.

10.5.3 Synthesis of results to support EMOP and EMAP reviews

To assist with the review process, particularly for the first review, we recommend a synthesis of all monitoring results is prepared to inform and support the review of the EMOP. The synthesis would determine environmental performance in relation to environmental management goals and the wider (adaptive) management framework and, if information is available, could also elucidate system-wide changes and variability that might feed into the (adaptive) management framework. The synthesis report should comprise trend analyses on key indicators for each habitat from all monitoring performed for the site. Feed inputs should be summarised in the report. The synthesis report should also include a summary of supplementary investigations and additional research performed at the site, and relevant findings (if available). The full results need not be presented within the synthesis report itself. The synthesis report could be combined with the last monitoring report before the EMOP is to be reviewed.

11. ENVIRONMENTAL QUALITY STANDARDS (EQSs)

11.1 Background and examples

An EQS is a specified threshold (or limit) for environmental change or disturbance. The main purpose of developing an EQS is to provide a measure of certain environmental conditions; usually corresponding to environmental management goals set for the operation of the farm. An EQS may, for example, reflect the lower limit of conditions that are deemed unacceptable or of concern. EQSs are therefore monitored for managing compliance and also setting EQSs is often a focal point for consultation with stakeholders.

In some instances, EQSs may be universal, published thresholds; like those set to guard against toxic effects on aquatic organisms from sediment or water contaminants. For example, based on eco-toxicological studies, sediment zinc concentrations at which toxic effects are 'possible' are set at 200 mg kg^{-1} in the ANZECC sediment quality guidelines (now ANZG 2018). In this example, an EQS can be developed reflecting an 'absolute' concentration that is not to be exceeded. It may be equally appropriate to focus an EQS on relative degree of change from the natural state of the ecosystem. For example, an EQS might stipulate that the average[57] concentration of total nitrogen at a location should not exceed the 70^{th}[58] percentile of concentration in the baseline data for that location. Such an approach with an element of conservatism may be useful for open ocean environments where relatively little is known of the response of the environment to finfish farming effects, and selection of absolute EQS values may be relatively more arbitrary (see discussion on initial EQSs below).

Examples of EQSs used to manage benthic effects of aquaculture at inshore sites in New Zealand and internationally are shown in Box 3. Because the EQSs in Box 3 were developed for inshore environments, they are not necessarily applicable to open ocean aquaculture. It is envisaged that EQSs for open ocean aquaculture sites will need to be developed on a site-specific basis, especially during the early phase of open ocean aquaculture development in New Zealand.

[57] Likely across at least a season at a single station. Depending upon the arrangement of stations, it might also be appropriate to average across stations.
[58] This percentile value was chosen purely for illustrative purposes. It should not be taken as a recommendation.

Box 3. Examples: Benthic EQSs for inshore soft-sediment environments

The example EQSs below are used to manage benthic effects of aquaculture at inshore sites in New Zealand and the international context. Because they were developed for inshore environments, it is not expected that these EQSs are directly applicable to open ocean aquaculture. However, the structural framework of these EQSs may be useful starting points for developing EQSs for open ocean aquaculture sites.

Marlborough Sounds

In the Marlborough Sounds, organic enrichment effects from existing salmon farming are managed by both a limit on the intensity of the deposition under the pens (enrichment stage (ES) 5; reflective of a 'peak-of-opportunist' infaunal state, where infaunal biomass and therefore waste assimilative capacity is theoretically maximal), and a limit on the spatial extent of the enrichment footprint (by specifying a specific ES value and/or the requirement to be comparable with unimpacted reference sites). The allowable spatial extent is managed using a zoning concept that includes an area of intense deposition (zone of maximum effects) and the outer boundary of effects (outer limit of effects). The size of the zones is typically site-specific, defined in the resource consent for each site. The approach is described in the Marlborough BMP (Keeley et al. 2019a).

In addition to the overall ES criteria, three readily assessible and widely established indicators of excessive enrichment and anaerobic conditions were also adopted. For these associated EQSs, compliance is achieved where: (1) not more than one replicate had macrofauna virtually absent; (2) bacteria mat (*Beggiatoa* sp.) coverage must be no more than localised/patchy in distribution; (3) there is no obvious spontaneous out-gassing (of H_2S or methane).

Big Glory Bay, Stewart Island

It is noted that Big Glory Bay on Stewart Island is a very sheltered bay with long residence time and low flow conditions, which are not reflective of conditions expected in open ocean finfish farming locations.

In Big Glory Bay, farms are required to meet the following EQSs for the seabed within 10 m of the edge of the pens:

- The benthic community retains a diversity and abundance of marine taxa (in addition to one or two opportunistic enrichment tolerant taxa such as Capitellid and *Dorvillea* worms, and nematodes) at levels which allow for sustained farm waste assimilative capacity and sufficient seabed recovery to support a farm rotation cycle with a fallowing period of not less than 5 years;
- No more than 1 of the 5 replicate cores collected have no taxa present. In any assessment under this condition, the effects of mussel shell substrate on benthic communities are to be ignored;
- No visually obvious, spontaneous out-gassing (H_2S/methane); and
- Bacteria mat (*Beggiatoa* sp.) coverage not greater than 50% of the sampled area.

Furthermore, the ANZECC sediment quality guidelines (now ANZG 2018) are used to measure the effects of metal contaminants.

Additional seabed monitoring requires measuring a range of parameters at 10 m from the pens, an outer zone of maximum impact (50 m downstream from the edge of pens) and outer zone of reduced effect (100 m downstream from edge of pens).

Note that Big Glory Bay on Stewart Island is in a relatively sheltered bay compared to proposed open ocean finfish farm locations.

International context

In the Norwegian Modelling–Ongrowing fish farms–Monitoring (MOM) programme, EQSs are applied to two zones: 'local', and 'intermediate and regional' (Ervik et al. 1997, Hansen et al. 2001), similar to the local zone and outer zone approaches utilised by other countries (e.g., Canada, Australia). The Norwegian MOM programme uses a multi-metric approach to measure seabed conditions against EQSs by combining the scores from groups of chemical, visual, and/or macro-infaunal indicators (similar to the Marlborough BMP).

In British Columbia, Canada, sulphides are the key focus indicator for compliance, but other indicators are sometimes also measured. For example, in Nova Scotia, if total free sulphides at > 50% of sampling stations exceed 3000 µM then additional monitoring must occur (Chang 2011).

In Tasmania, within the lease area there must not be any significant visual impacts including excessive feed dumping, extensive bacterial mats, and spontaneous gas bubbling. In the event that gas bubbles are present at the surface without seabed disturbance (e.g., methane or hydrogen sulphide), the farm is to be fallowed immediately (Department of Primary Industries 2018, Ross & Macleod 2013). Tasmania also uses the ANZECC sediment quality guidelines (now ANZG 2018) to assess the risk from sediment metal contamination. Readers are referred to the review by the Environment Protection Authority Tasmania (2019) for further details.

In Scotland, a strengthened regulatory framework was implemented in May 2019, resulting in the indicators and associated thresholds shown below[59]. Farms must be operated so that there is no significant adverse impact on the biodiversity of seabed life beyond the edge of a permitted mixing zone. Within the mixing zone, wastes must not accumulate to levels that would compromise the biological process needed to breakdown and assimilate them. Computer modelling is used to assess whether proposed developments will be able to operate without compromising these processes. Environmental monitoring results are used to check that the required standards are being maintained at operational sites. No quality standards for seabed habitats dominated by large stones, rock, or other hard materials are provided. Visual imagery survey results are used to assess the condition of these habitats if present within the mixing zone of a farm.

Scottish Environment Protection Agency (SEPA) environmental seabed standards for marine finfish farming

What the standard applies to	Where the standard applies	The type of standard	How the standard is measured	What the standard is
Condition of invertebrate animals living in soft sediments	At mixing zone limit & beyond	Good status standard	Infaunal quality index method as specified under 2014 Standards Directions[60]	0.64 as minimum value at any time
Most extreme permitted effect of waste deposition on seabed invertebrate animal communities	In mixing zone	Basic seabed functioning standard	Number of species, and abundance, of the re-worker polychaete worms: • all polychaete species listed as "AMBI Group V"[61] species; • *Ophryotrocha* species; and • *Boudemos* species	A minimum of 2 species with a combined abundance of more than 1000 individuals per m^{-2}
Maximum concentration of in-feed sea lice medicine, emamectin benzoate	At mixing zone limit & beyond	Good status standard	ng per kg of marine sediment (dry weight)	23.5
	In mixing zone	Basic seabed functioning standard	ng per kg of marine sediment (dry weight)	235

[59] https://www.sepa.org.uk/regulations/water/aquaculture/environmental-standards.
[60] http://www.wfduk.org/resources%20/coastal-and-transitional-waters-benthic-invertebrate-fauna.
[61] "AMBI Group V" species as listed by Borja et al. (2000).

11.2 Considerations for developing site-specific EQSs

The definition of an EQS depends on a range of factors, including farm operational aspects, the policy context, environmental characteristics, as well as the level of changes that are considered acceptable by stakeholders, and non-ecological considerations. EQSs therefore often need to be tailored to specific sites. Factors informing EQS development should include the following.

- The policy context, to ensure assessment of compliance with the EQS aligns with objectives and compliance requirements of regional planning instruments; for example, water quality standards or definitions of ecologically significant sites in coastal plans (these compliance requirements are typically also stipulated by consent conditions).
- The environmental context, to ensure that the EQS takes into consideration natural variability and wider environmental change.
- Environmental management goals, to ensure that assessment of compliance with the EQS enables assessment of whether environmental management goals have been met.
- Operational and practical constraints, to ensure that monitoring required to assess compliance with the EQS is feasible.

Importantly, our ability to develop scientifically robust EQSs depends on our understanding of the sensitivity and assimilative capacity of the environment, its functional response to effects from finfish farming, and the natural variability of the environment.

It is likely that there will be some level of uncertainty at the time of preparing an AEE and, particularly for open ocean aquaculture, probably also at the time of granting consent. Accordingly, 'initial' EQSs (iEQSs) will likely be appropriate starting points to provide initial indications of expected environmental change. Because of the likely need for adaptation, some EQSs (especially those with fixed defined limits) may be best specified in EMAPs and not in consent conditions. iEQSs assist in identifying initial monitoring indicators, and monitoring results will in turn assist in establishing and refining EQSs (that might become enforceable thresholds) over time. It is likely that EQSs will require ongoing review and refinement at least through the early stages of the consent.

In addition to changes in EQSs, it is likely that changes in monitoring indicators used to infer compliance with the EQSs will also be required or desirable to take advantage of new knowledge or technological advancements. This may also result in further development or refinement of EQSs.

It is important that this expected level of ongoing review and refinement is provided for in the consent framework, as emphasised earlier in section 3.

12. THE ROLE OF OTHER ENVIRONMENTAL INFORMATION SOURCES

12.1 The role of central government and research providers in targeting research

The Guidelines recognise that there is a need for research (rather than monitoring) to fill some information gaps on the effects of open ocean aquaculture[62]. It is not possible to provide specific guidance on where consent-related monitoring turns into research. The complexity of scientific investigations and monitoring that can, under the RMA, be expected to be carried out by consent

[62] For example, in November 2019 a meeting was convened by Fisheries New Zealand to identify and prioritise research needs for open ocean aquaculture. Representatives from government, iwi, Māori interest groups, research institutes, and industry were invited to participate.

applicants or holders depends on the scale and effects of the activity and the sensitivity of the environment.

The technical guidance provided is focused on informing monitoring in the context of consent-driven resource management and compliance. However, knowledge gaps may encourage central government and research support of the open ocean aquaculture industry through targeted funding or research toward these knowledge gaps, particularly those outside the scope of consent-related environmental monitoring.

The value of research that is geared toward supporting sustainable open ocean aquaculture will partially depend on communication among industry, councils, and science providers to ensure that it focuses on the areas of highest needs and generates optimal improvement within the resource management context.

12.2 The role of State of the Environment (SOE) monitoring and other information gathered by councils

SOE monitoring is carried out by councils to collect information on the state and condition of the environment to monitor trends over time. SOE monitoring has a wide range of target audiences, including council policy and planning staff, science, iwi, communities, industry and business, central government, environmental groups, and schools.

SOE monitoring in the Coastal Marine Area (CMA) is predominantly confined to locations near the coastline, in either the intertidal or shallow subtidal areas (for example, estuaries). This is because these locations are typically priority areas for regional or local resource management.

Councils also monitor the environment to assess the effectiveness of plans and policies and initiate research and investigations into key regional environmental management questions. Very few (if any) councils currently have monitoring programmes in areas that would be relevant to open ocean aquaculture.

12.3 Supplementary investigations by consent holders

In addition to 'routine' monitoring, supplementary investigations comprising one-off investigations, or 'additional' monitoring may be appropriate means of addressing uncertainty or knowledge gaps associated with the development of open ocean finfish farms. These investigations may not be strictly related to compliance with EQSs but instead could be triggered by the need to investigate unexpected monitoring results (see discussion on adaptive management in section 3.2), develop or review EQSs, or provide other contextual information for effects management. These types of supplementary investigations could be initiated voluntarily by consent holders or may sit within the management response framework.

The outcomes of these supplementary investigations may be used to inform operational and resource management decisions related to the consented activity, such as potential further development or improved mitigation, management, or monitoring of effects. For example, investigations could be undertaken to improve certainty around effects of organic enrichment on biogenic habitats found within the farm footprint. This could lead to improved EQSs and monitoring approaches, with a reduced monitoring effort reflecting the improved understanding of the effect.

Using supplementary investigations to fill uncertainties that are key to effects management likely requires collaboration between consent holders and councils. This is because such investigations can often only be undertaken once effects occur; i.e., once a farm is in operation. Consent holders and councils may need to explore strategic approaches to align and balance comprehensive research aimed at improvements in knowledge and understanding of environmental risks, with less intensive routine monitoring that provides the required safeguards and information for consent compliance until the environmental risk is better understood.

12.4 Resourcing implications

Within the New Zealand research funding model, it can be challenging to attract research funding for the types of research questions that would directly benefit resource management of open ocean aquaculture, especially when such investigations or research is site-specific. Similarly, funding supplementary investigations may pose challenges for individual consent holders.

All parties are encouraged to take a strategic approach to improving our level of knowledge and understanding related to benthic and water quality effects of open ocean aquaculture. This includes collaborations among scientists, central government, councils, and industry. In particular, collaboration within the industry and also with other complementary industries would add value to generating and collating data and knowledge. It is acknowledged that funding research and supplementary investigations requires substantial investment. However, the resulting reduction in uncertainty will likely create opportunities for improving efficiencies in farm management and monitoring, and thus cost reductions over time.

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15. GLOSSARY

Term	Definition/description
Adaptive management	A systematic, iterative process of decision making in the face of uncertainty, with the aim of meeting resource management objectives and reducing uncertainty over time through system monitoring and if required adaptation of operational management in response to learning about the system.
Area of interest	The region around a proposed/existing fish farm in which it is deemed possible that the farm's effect on benthic or water quality characteristics is measurable. In open ocean aquaculture, the area of interest may be considerably larger than for inshore aquaculture because of the greater likelihood of far-field effects in dispersive environments.
Assessment of environmental effects (AEE)	An assessment of the effects of an activity on the environment as described in Schedule 4 of the RMA.
Assimilative capacity	The capacity of the environment to assimilate a farm-derived discharge of potential environmental concern. This does not necessarily mean that a particular contaminant (and its ecological effect) is completely removed, absorbed, or converted to other chemical forms; rather it means that the concentration or load of the contaminant remains within some chosen acceptable level.
Baseline survey	A survey that aims to establish natural variability, provide fine-scale descriptions of the existing natural environment, or fill a specific information gap identified during the consenting process. Baseline data must be sufficiently robust in time and space to enable effective monitoring of environmental change.
Benthic environment	All biological, physical, and chemical aspects of the foreshore and seabed. For example, it includes infauna, epifauna, benthic algae, organic matter processing, nitrogen cycling, biogenic habitats, and soft sediments.
Biogenic habitat	Habitat formed by living or dead plants or animals in sufficient density and extent that their three-dimensional structure or interaction with the substrate provides substantive ecosystem services such as shelter, protection, and resources for at least one phase of the lifecycles of other marine macro-biota. Examples include kelp forests, rhodolith beds, coral structures, bryozoan thickets, sponge gardens, tubeworm fields, and horse mussel beds.
Chlorophyll	A photosynthetic pigment found within phytoplankton (and many other plants). In the Guidelines chlorophyll generally refers to chlorophyll *a*.
Coastal marine area (CMA)	Coastal marine area as defined in section 2 of the RMA
Complex habitat	Complex habitats are ones which are structurally diverse on spatial or temporal scales relevant to macro-biota that inhabit or make meaningful use of the habitat during at least one phase of their lifecycle. Examples include rocky reef and biogenic habitat. Complex habitats can promote species coexistence by providing a wide range of niches, thereby increasing diversity.
Consent	A resource consent as set out in section 87 of the RMA and includes all conditions to which the consent is subject.
Contaminant	Unless defined otherwise in the text, contaminant is used as defined in section 3 of the RMA, i.e., includes any substance (including gases, odorous compounds, liquids, solids, and micro-organisms) or energy (excluding noise) or heat, that either by itself or in combination with the same, similar, or other substances, energy, or heat

	(a) when discharged into water, changes or is likely to change the physical, chemical, or biological condition of water; or (b) when discharged onto or into land or into air, changes or is likely to change the physical, chemical, or biological condition of the land or air onto or into which it is discharged.
Council	A regional council or unitary authority as defined in the Local Government Act 2002.
Descriptive survey	A survey with emphasis on characterising and describing environmental conditions in the survey area, which typically forms the basis for assessments of environmental effects.
Effect	Effect is used as defined in section 3 of the RMA, i.e., includes (a) any positive or adverse effect; and (b) any temporary or permanent effect; and (c) any past, present, or future effect; and (d) any cumulative effect which arises over time or in combination with other effects—regardless of the scale, intensity, duration, or frequency of the effect, and also includes (e) any potential effect of high probability; and (f) any potential effect of low probability which has a high potential impact.
Environmental management goal	A measure or statement that describes a desired environmental outcome or a desired level of protection and reflects ecological and community values. An environmental management goal forms the basis for setting appropriate environmental quality standards and monitoring objectives, and for determining (adaptive) management strategies for achieving them. Other terms used elsewhere to describe environmental management goals as defined in the Guidelines include environmental management objectives or environmental outcomes.
Environmental management plan (EMAP)	For the purpose of the Guidelines, an environmental management plan (EMAP) sets out how the potential effects and environmental risks of the consented activity are to be managed as part of conducting the consented activity. EMAPs should include specific environmental management goals, environmental quality standards (EQSs), and management responses in the event that EQSs are breached. If a consent is managed under an adaptive management approach, the EMAP should include all necessary additional components required for effective adaptive management.
Environmental monitoring plan (EMOP)	For the purpose of the Guidelines, an environmental monitoring plan (EMOP) describes how environmental monitoring is to be conducted and how monitoring results are to be assessed. EMOPs should include monitoring objectives aimed at evaluating the effects of the consented activity and, specifically, whether environmental management goals and EQSs set in EMAPs are met. They may also include monitoring of the effectiveness of mitigation activities.
Epifauna	Animals living on the surface of the seabed or attached to submerged objects, aquatic animals, or aquatic plants.
Environmental quality standard (EQS)	A threshold (or limit) for environmental change or disturbance that reflects a specified level of environmental change, typically indicative of an adverse effect. An EQS may reflect a threshold below which no adverse effect on the environment occurs, a threshold above which effects are deemed unacceptable, or a threshold that triggers a specific management response. An EQS can be a single indicator value or comprise multiple indicators and can be numeric or descriptive. EQSs

	should be closely aligned with environmental management goals. EQSs are sometimes referred to elsewhere as 'performance standards', especially if they are qualitative descriptions of environmental change or conditions.
Eutrophication	Over-enrichment of a water body with nutrients, commonly resulting in excessive growth of phytoplankton and depletion of oxygen concentration.
Existing natural environment	Refers to the characteristics of the natural environment before the consented activity has taken place.
Far-field (and near-field)	**For benthic monitoring** **Near-field:** The seabed area comprising the 'primary footprint', typically defined as a continuous area/zone around a single block of pens, within which uneaten feed and faecal waste, or other deposited material, initially reaches (or is expected to reach) the seabed during the period of highest discharge intensity of the production cycle. In dispersive environments, this could be up to 1 km (but possibly more) in the primary flow direction/s. Distances may differ for different deposited substances. **Far-field:** The seabed outside the primary footprint.**For water quality monitoring** For water quality applications, the regions considered to be near- or far-field are related to distances and angles travelled by sequential 'parcels' of water over a timescale that is appropriate to the biogeochemical transformations influencing the water quality property in question. **Near-field:** The region within which little biogeochemical transformation of the original farm-derived material has yet had an opportunity to take place amongst recently released material. **Far-field:** The geographic region beyond a perimeter that is sufficiently distant (in travel time) to ensure that the biogeochemical form(s) across which the original discharge is now distributed is no longer dominated by the original release form (because there has been ample time for biogeochemical reactions to operate on the original material).
Farm footprint	The area of predicted effects (predicted farm footprint), or actual effects identified through monitoring (actual farm footprint). The farm footprint is determined by a specific value range of an indicator. The farm footprint may be one cohesive area or a group of separate areas. Note that the benthic farm footprint and water quality farm footprint are usually different. Also, individual benthic (or water quality) indicators may define the benthic (or water quality) farm footprint differently. For water quality effects, this area is also referred to as the 'plume' of the indicator (typically a specific discharge). The farm footprint is usually a subset of the area of interest.
Guidelines	This document.
Habitat	Habitat is the type of natural environment where organisms can find substrate, food, shelter, protection, and mates for reproduction. It is characterised by both physical and biological features.
Indicator	A characteristic of the environment that, when measured, quantifies the magnitude of stress, habitat characteristics, degree of exposure to a stressor, or degree of ecological response to the exposure.
Infauna	Animals living predominantly in the sediment.
Inshore aquaculture	Aquaculture taking place in inshore environments. For the purpose of the Guidelines, this includes all existing developed finfish farms in New Zealand as of 12 August 2020.

Inshore environment	The part of the CMA comprising semi-enclosed bays and harbours or other sheltered or semi-sheltered locations around mainland New Zealand and larger offshore islands.
Monitoring	Monitoring involves systematic repeat measurements and periodic analysis and reporting of information over time. For the purpose of the Guidelines, monitoring refers to consent-related monitoring, that is monitoring aimed at tracking changes in the environment in relation to specific predicted or potential adverse effects of the consented activity. Monitoring is typically used to: • Identify actual effects; • Assess whether effects predictions were correct; • Assess whether environmental management goals are met; • Assess whether EQSs are complied with; • Assess whether the EQSs established at the time of consent are fit for purpose; • Make recommendations for future monitoring; and/or • Provide feedback to stakeholders. The term SOE monitoring is used to specifically refer to State of the Environment (SOE) monitoring.
Monitoring objective	(Usually) quantitative statements that provide a means to evaluate whether environmental management goals or EQSs were achieved. Monitoring objectives should be specific, quantifiable, and practicably achievable based on environmental characteristics, resource availability, and sensitivity of methods.
Multi-trophic aquaculture (MTA)	Involves the farming of two or more species of different trophic levels in close proximity to one another, usually such that one utilises waste products of the other.
Near-field	See far-field,
Open ocean aquaculture	Aquaculture taking place in open ocean environments as defined in the Guidelines. This definition includes the existing applications in Marlborough, Southland, and Otago as at the publication date of this document.
Open ocean environment	Areas outside semi-enclosed bays and harbours or other sheltered or semi-sheltered locations around mainland New Zealand and larger offshore islands. For the purpose of the Guidelines, it is assumed that open ocean environments targeted by industry for open ocean aquaculture are highly dispersive environments. Open ocean environments do not include harbours and other areas listed in Schedule 3 of the NES-MA.
Operational management	Refers to operational aquaculture management activities of the operator, such as implementation of best practice environmental management or mitigation activities.
Operator	An individual or company responsible for the operation of an existing or proposed marine farm.
Performance standard	In the context of consent condition, this describes environmental change or environmental conditions that are required to be achieved. Performance standards are sometimes used interchangeable with EQSs. Often, the term performance standard is used to refer to high level qualitative descriptions of environmental change or conditions, whereas the term EQS typically refers to specific, commonly numeric, descriptions.
Pilot survey	A field component used not strictly for data collection, but may be necessary to develop, refine, and/or test sampling approaches to ensure they are fit for purpose for baseline surveying and ongoing monitoring.

Reference station	One or more monitoring stations chosen to represent environmental conditions in a given area outside the area of interest from the proposed or consented marine farm. Reference stations are sometimes also referred to as control stations in other documents.
Replicate samples	A series of samples collected in the same time frame, at the same sampling station, and in the same manner which enables statistical comparison.
Resource management	The management of natural and physical resources under the Resource Management Act. In the Guidelines, resource management typically refers to processes under the RMA, such as those related to resource consent applications, processing, compliance management, review of conditions, or policy or plan development. The term includes responsibilities of those administering the RMA, including regional councils, territorial authorities, unitary authorities, the EPA, MfE, and DOC.
Solute inorganic nutrients	The dissolved inorganic forms of the major macro-nutrients: ammonium, nitrate and nitrite, dissolved reactive phosphorus, and dissolved reactive silicon.
State of the environment (SOE) monitoring	Monitoring carried out by councils to collect information on the state and condition of the environment and monitor trends in condition over time. It involves repeated measurements of variables that can be used to quantify trajectories of temporal and/or spatial change in the environment to provide information on the state of, and pressures on, the environment, and changes in these. It can provide early warning of issues and broad-scale environmental change, and support decision-making on the effectiveness of regional policies or management actions.
Technical guidance	Refers to the detailed guidance on benthic and water quality monitoring provided in sections 7 and 8 of this report.
Therapeutants	Additives to a marine farm for the purpose of improving farmed stock health.
Water quality	Encompasses the chemical, physical, and biological characteristics of water with respect to its suitability for a specific purpose, e.g., drinking or providing suitable conditions for aquatic organisms. Water quality is a subjective term and ‘good’ versus ‘poor’ quality is defined by the properties (e.g., dissolved oxygen) and the levels (e.g., chemical concentration or saturation) of these properties that are set for the chosen purpose. In aquaculture, the water quality variables determined must be monitored to safeguard the cultivated organisms and/or the surrounding environment.
Working group	The Fisheries New Zealand-led working group that has prepared the Guidelines and will review them over time.

APPENDIX 1: SUMMARY OF RECENT REPORTS ON THE ENVIRONMENTAL EFFECTS OF OPEN OCEAN AQUACULTURE

This appendix provides summaries of two reports commissioned by Fisheries New Zealand to gather background information for the Guidelines. Bennett et al. (2020) reviewed a recent report by the Nature Conservancy, who assessed the open ocean aquaculture industry's potential to deliver healthy, sustainable seafood to satisfy its rapidly growing global demand (O'Shea et al. 2019) and discussed whether environmental effects associated with open ocean salmon farming outlined in the report apply to New Zealand, based on our current state of knowledge. Keeley (2020) provided a Norwegian perspective on the state of knowledge surrounding benthic and water column effects associated with open ocean aquaculture. This appendix also provides a brief overview of statements relating to the transition of aquaculture into open ocean environments made in a recent review of Tasmanian and international regulatory requirements for salmonid aquaculture (Environment Protection Authority Tasmania 2019). Most sections of this appendix are taken verbatim from the three reports.

Bennett et al. (2020)

A recent report by the Nature Conservancy (O'Shea et al. 2019) referred to as the 'Blue Revolution report') includes a review of potential ecological benefits and risks of environmental effects associated with open ocean salmon farming. Bennett et al. (2020) compared these findings to the New Zealand context and assessed how the expected environmental effects of aquaculture in open ocean environments compare with current inshore farming practices. It is noted that these comparative assessments are subject to the scale of the farming operation. In relation to effects on benthic habitats and water quality associated with salmon farming they derived at the following conclusions:

- **Benthic habitat impacts**

  <u>Conclusion on the difference between open ocean and inshore farming:</u> Worsened to Improved compared with current inshore farming practices.
  <u>Assessment:</u> The deeper, high-flow waters likely in open ocean environments may reduce/ dilute the amount of waste reaching the seabed. However, effects are unlikely to be negated (as per the Blue Revolution report) and would still need to be assessed on a case-by-case basis. We note that reductions in tidal flows and wave-induced benthic currents could arise at deep open ocean sites away from the coast, potentially exacerbating effects to the seabed. However, because there is the potential for larger areas to be considered in some open ocean environments, sensitive habitats can be more easily avoided and lower intensity farming (per unit area) may be able to mitigate effects. It is also likely to be important to consider far-field effects in areas with increased dispersion potential (i.e., high-water flows). We note that far-field depositional effects will be less intense than near-field deposition.

- **Water quality impacts**

  <u>Conclusion on the difference between open ocean and inshore farming:</u> Worsened to Improved compared with current inshore farming practices.
  <u>Assessment:</u> Depending on the characteristics of the site, dilution of farm discharges may be improved or worse compared with inshore sites in open ocean environments; e.g., if farms are larger and the currents are lower, then water quality effects are expected to be worse. However, because there is the potential for larger areas to be considered in some open ocean environments, lower intensity farming (per unit area) may be able to mitigate effects. Further to this, in New Zealand, open ocean conditions can be encountered near to the coast. Therefore, transport of farm nutrients into coastal embayments also remains possible

Overall, Bennett et al. (2020) conclude that, although open ocean salmon farming may provide environmental advantages over current inshore practices, environmental effects could still occur and must be managed appropriately on a case-by-case basis.

Bennett el al. (2020) recommend that open ocean farms are developed using a staged approach to enable monitoring and understanding of the environmental response before development is increased. Furthermore, environmental monitoring should be undertaken to ensure that expected advantages are realised. The authors consider it imperative that management plans are prepared prior to the installation of open ocean farming infrastructure to avoid unacceptable adverse effects and suggest that effects-based management strategies are adopted to reduce and mitigate potential environmental effects. They clarify that effects-based management involves monitoring of potential effects of concern and adapting farming practices to ensure unacceptable effects do not eventuate as the activity progresses.

Bennett et al. (2020) further note that, as with inshore sites, a range of flow environments are possible at open ocean sites. Because open ocean sites may be high-flow environments where wastes will be readily dispersed, far-field effects are also possible. Therefore, monitoring of effects in open ocean environments may need to consider wider ecological effects rather than just small-scale effects directly under the farm.

Keeley (2020)
Keeley (2020) provides an international perspective on the state of knowledge surrounding benthic and water column effects associated with aquaculture in some open ocean environments with a particular focus on the potential extent of farm footprints, the type and severity of under-farm impacts, and rates of recovery following farm retirement.

The main findings of this report are:

- Because the term ‘open ocean aquaculture’ is somewhat ambiguous, it is recommended sites be viewed in terms of their physical dispersion properties, which are determined primarily by three main factors: currents, wave exposure, and site depth. Possibly the most influential and pertinent to more exposed sites is wave action, yet it is also the least well understood.
- Experience from New Zealand and Norway shows high dispersion sites generally have a diffuse benthic effects footprint, whereby conditions directly beneath the cages are less likely to reach a point of extreme enrichment or benthic malfunctioning. Despite very high production levels and significant biodepositional fluxes, the enrichment state (ES) beneath high dispersion farms studied in Norway has not exceeded ES 5. However, the spatial extent of the effects footprint may be relatively large (e.g., 400–1000m in one direction) and traces of farm waste can be identified up to 1500 m away. The positive identification of farm waste (using tracers) does not necessarily infer measurable ecological effects, and far-field ecological impacts are difficult to detect.
- Although a good proportion of the waste at dispersive sites is processed in the near-field benthic environment, a significant portion (e.g., about 60%) is exported and processed further afield, where it is more dilute and presumably readily assimilated. The same dispersion process does, however, also elevate scope for interactions with other farms (overlapping effects) and neighbouring habitats, albeit at a very low level.
- Benthic recovery at high dispersion sites appears to be much more rapid than at non-dispersive sites, where cumulative and acute effects are more likely to manifest. In a case study provided, substantive recovery was observed in 7 months at a high dispersion site as opposed to 5–10 years at heavily impacted low dispersion sites.
- Highly dispersive sites have a greater tendency to occur in close proximity to mixed and hard bottom substrates, which are inhabited by epifauna and mobile demersal species for which little is known in terms of sensitivities to enrichment. In a case study provided, the anticipated high sensitivity of the selected epifauna species to enrichment was not immediately apparent; instead indicating some moderate resilience and more subtle and complex changes.
- Modelling of waste dispersion and associated benthic effects is relatively difficult at high dispersion sites due to greater model uncertainty, mainly relating to resuspension processes and

the effects of wave action. The report outlines some areas where existing modelling approaches could be refined to reduce this uncertainty.

- With respect to water column enrichment effects, the report describes three complementary Norwegian studies that examined potential nutrient enrichment in the water column surrounding dispersive sites. These studies failed to identify significant ecological impacts beyond the immediate vicinity of the farm.
- Some existing and new benthic monitoring and research tools relevant to New Zealand include:
  - A new method for measuring total free sulphides in sediments.
  - Advances with the use of microbial environmental DNA (eDNA) to determine benthic enrichment levels, which is being progressively implemented globally.
  - Modified video surveying techniques to evaluate effects on epifauna in mixed and hard bottom habitats.
  - A substrate-independent sampling device that, when coupled with microbial eDNA procedures, has potential as a widely applicable new tool for evaluating the spatial extent of benthic effects or fish farm waste dispersal.
  - Existing and evolving tracer techniques (i.e., terrestrial fatty acids, carbon and nitrogen isotopes, genetic signatures and microbial eDNA) for tracking and mapping the spatial influence of fish farm wastes.

Keeley (2020) also identified the following knowledge gaps:

- Farming in dispersive physical environments will likely encounter new habitats and species for which little is known about their tolerance to organic waste. This will likely require new species sensitivity knowledge and/or development of new environmental monitoring strategies. It is quite likely that microbial eDNA via SIBS will provide a solution to monitoring hard and mixed bottom environments in the near future.
- The role of wave action in the redistribution and 'cleansing' of sediments is not well understood and particularly pertinent in exposed locations, both in terms of understanding and predicting benthic effects and the time scales of recovery.
- There is a need for additional studies on benthic recovery at high dispersion sites to confirm, or otherwise, the relatively rapid recovery described in this report.
- Further refinement of predictive depositional models is required, because many of the relatively complex processes are only approximated at present.
- Although limited available evidence suggests that far-field effects in high dispersion sites are unlikely to manifest, the potential for ecological effects resulting from microbial or pathogen loads, as well as other subtle drivers of benthic ecological change, should not be disregarded, especially for large developments with multiple high production farming blocks.

Finally, Keeley (2020) provided the following guidance that he considered may be useful to mitigate and manage risks in the development of any new offshore sites, as follows:

- High dispersion sites have a less severe, more diffuse footprint than low dispersion areas. The assimilation capacities are less likely to be exceeded and the benthic processes remain more aerobic and functional and therefore recovery is likely to be more rapid. The trade-off is that the spatial extent of effects is greater, albeit at levels where change appears difficult to detect.
- Recommendations for optimal farm placement (within a multi-farm area) include the following.
  - Minimise overlapping effects (suggested > 2 km spacing where possible).
  - Single year class stocking to gradually ramp up feed use and provide scope for fallowing between year classes.

    - Maintain redundancy in space to facilitate responsive site management.

- Environmental management triggers should be based around effects indicators as opposed to tracers. Tracer studies can be used to confirm the potential farm 'influence field' but that needs to be balanced against measurements of ecological effects (appropriate to the species/habitat in question).
- Baseline surveys in 'new' regions should include the collection of microbial eDNA samples (using established methods) in conjunction with macrofauna samples for benthic enrichment monitoring.
- As part of the development or implementation of new monitoring approaches, it is suggested that:
    - Baseline surveys and preliminary monitoring in 'new' environments would ideally include the collection of microbial eDNA samples in conjunction with conventional (macrofauna) sampling. This approach should extend to mixed or hard bottom habitats as appropriate.
    - A process is developed for transferring different indicators and environmental thresholds to new environments. The conventionally accepted 'adaptive development' and staged development approach is not necessarily conducive to this.
    - Existing tracer tools should be used to better define the potential far-field footprint of farms in high dispersion sites and help target monitoring efforts where appropriate.

Environment Protection Authority Tasmania (2019)

Although mainly focused on 'traditional' inshore aquaculture, the Environment Protection Authority Tasmania (2019) report contains some references to environmental effects of open ocean salmon farming, which are discussed here.

- Recently, there has been a greater push to establish new fish farms away from sheltered coastal locations and into more exposed/offshore locations. A general perception when setting up new fish farms in more exposed and deeper locations is that there will be a reduction in environmental impacts through increased dispersal and dilution, reduced deposition, and limited accumulation of particulate wastes adjacent to net pens (Holmer 2010). A growing body of evidence suggests that such a shift in farming practices may be less harmful for marine ecosystems in many situations. Monitoring observations under salmon farms in New Zealand with contrasting current flow regimes (high vs. low flow), demonstrated that benthic sediment ecosystems had a greater tolerance to organic enrichment under higher flow conditions compared with low flow conditions. Studies in the Mediterranean and Norway (Bannister et al. 2014, Keeley et al. 2019a) suggest that intensive farming of finfish in more exposed locations significantly reduces localised impacts on the receiving environment.
- In offshore/exposed locations off the coast of Tasmania (e.g., Storm Bay), there is limited knowledge on the assimilative capacity of marine sediments to organic enrichment and associated elevated nutrients. Evidence presented to date (i.e., from benthic environmental monitoring reports) indicates that the effects of organic enrichment in the vicinity of fish cages is localised, but sporadic, with limited accumulation of organic waste under cages over time. Environmental forcing (e.g., irregular wave activity and current velocities) may increase the dispersal capacity of organic waste into the receiving environment. However, the transportation pathway for organic waste is unclear, and further studies are required to ascertain if any broader-scale effects of organic enrichment are occurring.

APPENDIX 2: POTENTIAL EFFECTS OF OPEN OCEAN FINFISH FARMING CONSIDERED IN THE GUIDELINES

The technical benthic and water quality guidance provided in sections 7 and 8 is structured around potential effects that may occur in open ocean finfish farming. The working group identified potential effects of open ocean finfish farming for the purpose of structuring the technical guidance by:

- Reviewing potential effects of inshore finfish farming on water quality and benthic environments primarily as described by Stenton-Dozey (2013) and Keeley (2013).
- Considering additional recent literature and expert knowledge held by the working group and using expert judgement to determine which effects are likely to be relevant, not relevant, or require modification when applied to open ocean finfish farming.
- Incorporating the findings from Bennett et al. (2020), Keeley (2020), and the Environmental Protection Agency Tasmania (2019) presented in Appendix 1.

The working group acknowledges the uncertainty inherent in the benthic and water quality effects of open ocean finfish farming resulting from the lack of New Zealand and international experience in operating such farms. Addressing this uncertainty has been considered in the preparation of the Guidelines. Furthermore, the relatively short initial review period compared to the length of the consents (see section 1.4) will provide for new information derived from our experience with open ocean finfish farms to be incorporated in the Guidelines.

Potential effects have been identified based on the state of knowledge at the time of writing the Guidelines (Table 9). Additional potential effects may be identified for specific open ocean finfish farms. Additional effects may be added in the future as our experience with open ocean finfish farming improves.

As a result of the state of knowledge and limited time available to produce the Guidelines, the working group had to prioritise which potential effects to provide guidance for. The working group chose those potential effects that were expected to most likely require monitoring (mainly because of the potential for ecologically important effects). As a result, no guidance is provided for some potential effects, but noting that effects not explicitly monitored may be detected by monitoring designed for other effects (Table 9). Furthermore, the level of detail of the technical guidance provided differs among potential effects, reflecting the state of knowledge and the working group's expert judgement on the transferability of monitoring approaches used for inshore finfish farming to open ocean environments.

Table 9: Potential benthic and water quality effects of open ocean finfish farming and identification of effects for which monitoring guidance is provided.

Environment	Potential effects	Monitoring guidance provided?
Benthic	Organic enrichment and associated changes in the benthic environment due to biodeposition or biofouling drop-off	Yes
	Smothering of benthic habitats and communities by biodeposits or biofouling drop-off	Yes
	Contamination of sediments due to the deposition of therapeutants, feed additives, antifoulants, or other potentially harmful substances used in finfish farm operations	Yes
	Changes in physical habitat due to biodeposition, biofouling drop-off, disturbance from mooring systems, or scouring due to the presence of the farm structure	No
	Shading of the seabed by farm structures	No
Water column	Increased concentrations of dissolved nutrients and subsequent enhanced phytoplankton biomass and growth	Yes
	Reduced concentrations of bottom water dissolved oxygen	Yes
	Reduced concentrations of surface water dissolved oxygen	No
	Increased concentrations of other contaminants such as therapeutants	Yes
	Effects from the presence of the farm upon waves, currents, water column stratification, and other hydrodynamic properties	No
	Effects from shading on water quality, or water column biology	No
	Effects of artificial lighting on water quality, or water column biology	No

APPENDIX 3: LEGAL REQUIREMENTS FOR CONDITIONS OF CONSENT AND EMPS (EMAPS/EMOPS) AS DIRECTED BY THE NEW ZEALAND COURTS

Giles & Barton (2020) reviewed direction provided in New Zealand court decisions on requirements for consent conditions and EMPs[63], implementation and enforceability of EMPs, and delegation of decisions to times after granting consent. This section presents a summary, including extracts, from Giles & Barton (2020). A summary of the case law analysis by Giles & Barton (2020) is shown in Table 10.

"The Environment Court has given consistent direction that conditions of consents must specify clear environmental outcomes and objectives for EMPs.[64] In addition, several judgments have required performance standards to be specified in the conditions of consent.[65] In *Auckland Volcanic Cones Society Inc v Transit New Zealand*, the Court considered it unsatisfactory that conditions designed to control the discharge of sediment left 'a significant discretion as to the acceptability of the details'.[66] While it acknowledged that the nature of the works required for the proposed extension of the motorway made the formulation of precise well-defined permits difficult, it requested that as an alternative a standard should be included as a condition 'for the control of sediment discharges that is effective in protecting the environment from adverse effects and which is clear, practicable and enforceable'.[67] In *Lower Waitaki River Management Society Inc v Canterbury Regional Council*, the Court redrafted conditions 'by moving the objectives so they are performance standards to be complied with, not only objectives for management plans'.[68] These strictures are similar to those of the Court of Appeal in *Trans-Tasman Resources* against vague language in a consent about avoiding adverse effects and leaving what it might really mean to management plans.[69] The Court of Appeal was equally insistent that if, at the time of granting the consent, there was uncertainty about likely environmental effects, the critical information gaps could not be filled by post-decision information gathering."

EMPs should stipulate how objectives specified in conditions will be met.[70] They can also be used to provide the methodology of how acceptable environmental limits are to be achieved.[71] Giles & Barton (2020) also examined the level of detail required in an EMP at the time of decision making on a consent, which has received varying direction from the Environment Court. The Environment Court has consistently stipulated that conditions of consents must specify implementation requirements for EMPs to ensure enforceability.[72] It also requires that there are no legal obstacles to the conduct of works required under an EMP.

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[63] In court decisions, EMAPs and EMOPs are generally referred to as EMPs.

[64] *Mount Field Ltd v Queenstown Lakes District Council* [2012] NZEnvC 262 at [7], [77] and [79]; *Royal Forest and Bird Protection Society Inc v Gisborne District Council* [2013] NZRMA 336 (EnvC) at 337 and [87]; *Lower Waitaki River Management Society Inc v Canterbury Regional Council* [2009] NZEnvC 242 at [463].

[65] *Wellington Fish and Game Council v Manawatu-Wanganui Regional Council* [2017] NZEnvC 37 at [175]; *Auckland Volcanic Cones Society Inc v Transit New Zealand Ltd* [2003] NZRMA 54 (EnvC) at [199]; *Lower Waitaki River Management Society Inc v Canterbury Regional Council*, above n 64, at [385].

[66] *Auckland Volcanic Cones Society Inc v Transit New Zealand Ltd*, above n 65, at [199].

[67] At [199].

[68] *Lower Waitaki River Management Society Inc v Canterbury Regional Council*, above n 64, at [385].

[69] *Trans-Tasman Resources Ltd v Taranaki-Whanganui Conservation Board* [2020] NZCA 86 at [12] and [227].

[70] *West Coast Environmental Network v West Coast Regional Council* [2013] NZEnvC 178 at [44]; *Wood v West Coast Regional Council* [2000] NZRMA 193 (EnvC) at 193; *Mount Field Ltd v Queenstown Lakes District Council*, above n 64, at [77].

[71] *Wellington Fish and Game Council v Manawatu-Wanganui Regional Council*, above n 65, at [175].

[72] *West Coast Environmental Network v West Coast Regional Council*, above n 70, at [54]; *Lower Waitaki River Management Society Inc v Canterbury Regional Council*, above n 64, at [554] and [408].

"The Environment Court and High Court have been consistent in holding that it is not lawful to delegate the making of substantive decisions, i.e., decisions that are sufficiently important to have a bearing on whether a consent should be granted or not.[73]"

"The courts have often followed *Turner v Allison*[74] allowing a consent authority to delegate to an official the role of certifying adherence to a standard, while not permitting the delegation of arbitral or judicial functions that the consent authority should have exercised itself.[75] In *Royal Forest and Bird Protection Society Inc v Gisborne District Council*, Judge Thompson discussed the difference between approval and certification. He explained that a condition that delegated substantive decision making was not acceptable and commented that the proposed conditions were so uncertain that 'it was difficult to see how the council will be acting in a certifier role, rather than making decisions that should have been made at first instance'.[76] Judge Skelton made a similar observation in *Wood v West Coast Regional Council,* saying:[77]

> '[18] It was generally accepted that it is not appropriate to provide for a management plan on the basis that it is to be approved by a consent authority or some delegated official at a later time, except to the extent that they may be regarded as certifiers.'

In *West Coast Environmental Network v West Coast Regional Council* Principal Judge Newhook expressed the importance of certification in situations where knowledge is incomplete and is expected to advance after granting resource consent.[78] He said that in such circumstances, conditions 'need to be flexible enough to allow the best possible environmental outcome to be achieved in the light of advancing knowledge and experience' and that:[79]

> '… [w]hat a management plan certifier is being asked to do is to confirm that the management plan concerned is the most appropriate means available at any given time to achieve the objectives stated in the conditions.'

In the High Court, MacKenzie J noted the need to examine the real nature of the decision that is to be delegated, to prevent unlawful delegation:[80]

> '… [t]he role of the delegate as certifier may conceal the fact that what is being delegated is the power to certify a matter which is an essential element of the decision which should be made by the tribunal. It is necessary to examine the real nature of the decision which the delegate is required to make, rather than the form in which the power to make that decision is conferred.'

This analysis revealed that the preparation or completion of EMPs after consent has been granted creates a risk of unlawful delegation. Managing this risk requires careful examination of the nature of the delegated decisions."

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73 *Director-General of Conservation v Marlborough District Council* [2004] 3 NZLR 127 (HC), at [28]; *Crest Energy Kaipara Ltd v Northland Regional Council* [2009] NZEnvC 374 at [222]; *Royal Forest and Bird Protection Society Inc v Gisborne District Council*, above n 64, at 337; *Mount Field Ltd v Queenstown Lakes District Council*, above n 64, at [77].

74 [1971] NZLR 833 (CA).

75 *Royal Forest and Bird Protection Society Inc v Gisborne District Council*, above n 64, at 337; *Mount Field Ltd v Queenstown Lakes District Council*, above n 64, at [77].

76 *Royal Forest and Bird Protection Society Inc v Gisborne District Council*, above n 64, at [88].

77 *Wood v West Coast Regional Council*, above n 70.

78 *West Coast Environmental Network v West Coast Regional Council*, above n 70, at [43].

79 At [43].

80 *Director-General of Conservation*, above n 73, at [27].

Table 10: Summary of case law analysis in Giles & Barton (2020) on requirements at the time of decision making on granting a resource consent under the RMA, focussing on (1) conditions of consent, (2) EMPs, (3) implementation and enforceability of EMPs and (4) delegation.

Cases reviewed	Requirements derived from case analyses
(1) Conditions of consent	
Auckland Volcanic Cones Society Inc v Transit New Zealand Ltd [2003] NZRMA 54 (EnvC) at [199]. *Lower Waitaki River Management Society Inc v Canterbury Regional Council* EnvC Christchurch C80/2009, 21 September 2009 at [385],[463]. *Mount Field Ltd v Queenstown Lakes District Council* [2012] NZEnvC 262 at [7], [77] and [79]. *Royal Forest and Bird Protection Society Inc v Gisborne District Council* [2013] NZRMA 336 (EnvC) at 337 and [87]. *Wellington Fish and Game Council v Manawatu-Wanganui Regional Council* [2017] NZEnvC 37 at [175].	• Conditions of consent must specify clear environmental outcomes and objectives for EMPs. • Objectives must not be too vague to be enforceable. • Acceptable levels of performance standards (for example for monitoring parameters) must be specified in conditions of consent. • Where the nature of the activity makes it difficult to formulate precise well-defined standards, clear, practicable, effective, and enforceable standard should be specified in conditions of consent to protect the environment from adverse effects.
(2) EMPs (purpose, content and level of completion)	

Re Canterbury Cricket Association Inc [2013] NZEnvC 184.
Crest Energy Kaipara Ltd v Northland Regional Council EnvC Auckland A132/2009, 22 December 2009 at [222].
Crest Energy Kaipara Ltd v Northland Regional Council [2011] NZEnvC 26, [2011] NZRMA 420 at [26].
Director-General of Conservation v Marlborough District Council EnvC Christchurch C113/2004, 17 August 2004 at [40].
Mount Field Ltd v Queenstown Lakes District Council [2012] NZEnvC 262 at [77] and [82].
Wellington Fish and Game Council v Manawatu-Wanganui Regional Council [2017] NZEnvC 37 at [175].
West Coast Environmental Network v West Coast Regional Council [2013] NZEnvC 178 at [44] and [45].
Wood v West Coast Regional Council [2000] NZRMA 193 (EnvC) at 193.
	• The purpose of EMPs is to specify how objectives specified in conditions will be met.
	• The methodology of how acceptable environmental limits are to be achieved can be provided in EMPs.
	• If the level of detail required in an EMP cannot be provided at the time of decision making on granting consent:
	  ○ As a minimum, the applicant must provide evidence demonstrating in broad terms how the objectives of the EMP are to be achieved; and
	  ○ The requirements for EMPs must be read in conjunction with other controls embedded in conditions.

(3) Implementation and enforceability of EMPs	
Lower Waitaki River Management Society Inc v Canterbury Regional Council [2009] NZEnvC 242 at [408] and [554]. *West Coast Environmental Network v West Coast Regional Council* [2013] NZEnvC 178 at [54].	• Consents must specify implementation requirements for EMPs in conditions to ensure they can be enforced. • There must be certainty that a consent holder has the legal right to perform all works required under an EMP.
(4) Delegation	
Crest Energy Kaipara Ltd v Northland Regional Council EnvC Auckland A132/2009, 22 December 2009 at [222]. *Director-General of Conservation v Marlborough District Council* [2004] 3 NZLR 127 (HC) at [28]. *Mount Field Ltd v Queenstown Lakes District Council* [2012] NZEnvC 262 at [77]. *Royal Forest and Bird Protection Society Inc v Gisborne District Council* [2013] NZRMA 336 (EnvC) at 337 and at [88]. *Turner v Allison* [1971] NZLR 833 (CA) at 856. *West Coast Environmental Network v West Coast Regional Council* [2013] NZEnvC 178 at [27] and [43]. *Wood v West Coast Regional Council* [2000] NZRMA 193 (EnvC).	• It is not lawful to delegate the making of substantive decisions. Substantive decisions are decisions that are sufficiently important to have a bearing on whether the consent should be granted or not. • Conditions of consent can delegate certification, but decision making must not constitute approval, arbitration, or a judicial function. • The preparation or completion of EMPs after consent has been granted creates a risk of unlawful delegation. Managing this risk requires careful examination of the nature of the delegated decisions.

APPENDIX 4: ENVIRONMENTAL DNA (eDNA) – OVERVIEW AND POTENTIAL APPLICATIONS IN MONITORING EFFECTS OF OPEN OCEAN AQUACULTURE

This appendix was prepared by Michael Bunce (EPA) with assistance from Xavier Pochon (Cawthron Institute) and subject to a review from Rich Ford (Fisheries New Zealand) with some input from other working group members. This is less scrutiny than the rest of the document (because this group does not hold the same level of expertise as the report authors in this area).

Consents for aquaculture issued under the Resource Management Act 1991 typically require the collection of ecological data to survey, baseline, and monitor potential impacts. The salient point with regard to collection of ecological data is that consents often do not specify which methods should be used, only that they are fit for purpose. Until recently, bio-surveys were largely restricted to indicator species which could be readily identifiable by morphology (e.g., infauna) or indirect measurements such as chlorophyll which is used as a proxy for phytoplankton biomass.

Microscopic assessment of macrofauna has been the mainstay bioindicator of aquaculture impacts. However, the characterisation of these communities is both time consuming and expensive. Over the last decade an alternative to these approaches has emerged—referred to as environmental DNA (or eDNA). eDNA refers to the genetic material that animals, plants, and bacteria shed as they move through any given environment. Within these ‘genetic breadcrumbs’ are DNA barcodes that, when scanned (akin to a supermarket barcodes found on food), provide an inventory of all the life that has deposited DNA into that environment. At the heart of eDNA is a technique called metabarcoding which refers to the ability of new DNA sequencing technologies to collate and read DNA ‘barcodes’ from many species at once. To provide an example: metabarcoding of fish DNA from a 9-litre seawater sample from an Australian coral reef yielded over 80 species of fish[81]. Rather than being restricted to indicator species that are ‘easy to spot’, eDNA can theoretically audit across the tree-of-life from microbes to megafauna, but in practice many studies restrict themselves to core groups, e.g., bacteria, fungi, vertebrates. There is a global challenge to adopt holistic ‘Ecosystem based management’ (EBM) approaches in the area of environmental stewardship[82] including aquaculture; eDNA has the potential to respond to this challenge.

There are now over 1300 papers that describe eDNA techniques and applications. eDNA approaches are used across the globe from biosecurity to rare species detection; while still in its infancy, eDNA is revolutionising global bioassessment efforts[83]. Figure 7, a 2019 eDNA survey of Wellington Harbour taken as part of the Open Waters Aotearoa programme, demonstrates the ability of eDNA to generate deep biotic audits across many trophic levels.

In a perfect scenario there would be a robust taxonomic framework for all life, and we would have DNA barcodes for all taxa. This is not the case, especially not for microbes. One key point with regard to eDNA approaches is that, while preferable, it is not absolutely necessary to link all barcodes to a specific species. In other words, it may be enough to say the assemblage of DNA barcodes between sites (or timepoints) is unchanged. It is likely that eDNA approaches in applications like aquaculture will employ a combination of taxonomy-based and taxonomy-free bio-surveys.

As New Zealand seeks to develop guidelines for open water aquaculture it is pertinent to ask what role eDNA could (or should) play in site surveys and biomonitoring. Indeed, work is underway to validate and implement an eDNA-based index for routine monitoring of nine salmon farms in the Marlborough

[81] https://www.nature.com/articles/s41598-017-12501-5

[82] https://www.sustainableseaschallenge.co.nz/about-us/why-do-we-need-ebm/

[83] Compson et al. (2020). Frontiers in *Ecology and Evolution 8*: 581835. doi: 10.3389/fevo.2020.581835

Sounds, New Zealand[84,85,86,87]. This adds to a wider body of published literature on what eDNA might be able to offer the aquaculture sector. Importantly, one key attribute of extracting DNA from environmental samples (sediment and/or water) is that these environmental samples act as a biological archive that can be queried years or decades later. It has been proposed that sample archiving (or biobanking) of environmental samples will be increasingly used in environmental consenting projects[88] and may be useful as evidence in enforcement actions.

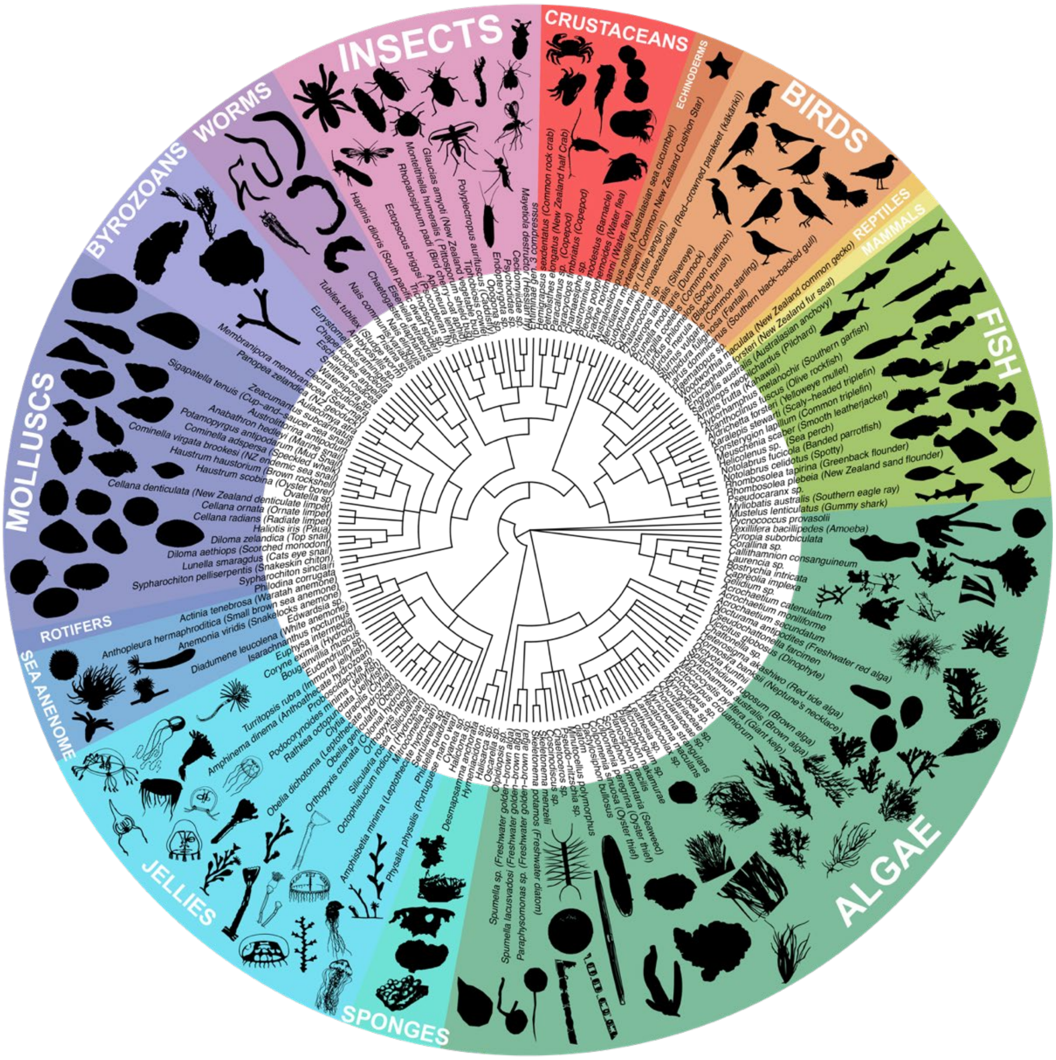

Figure 7: A biotic survey of Wellington Harbour using eDNA. Such methods could play a role in baselining the biota and then monitoring the impact (or lack thereof) of aquaculture farms. Image from the EPA's Open Waters Aotearoa initiative and Wilderlab.

[84] Pochon et al. (2015). *Marine Pollution Bulletin 100 (1)*: 370–382. doi.org/10.1016/j.marpolbul.2015.08.022

[85] Dowle et al. (2015). *FEMS Microbiology and Ecology 91(8)*: fiv089. doi.org/10.1093/femsec/fiv089

[86] Keeley et al. (2018) *Ecological Indicators 85*: 1044–1057. doi.org/10.1016/j.ecolind.2017.11.014

[87] Pochon et al. (2020) Cawthron Report 3400.

[88] Jarman et al. (2018) *Nature Ecology & Evolution (2018) 2 (8)*: 1192–1193.

Can eDNA detect impacts temporally and spatially?

A recurring question to advocates of eDNA is "Water and sediment move around, surely the DNA just gets all mixed up?". There are now numerous studies in New Zealand, and around the world, that demonstrate that water samples taken only a few hundred metres apart yield statistically different biotic assemblages[89,90,91,92]. This holds true when sampling different depths[93]. Although much of this work is in shallow environments (< 50 m), unpublished work has demonstrated eDNA-based assemblages can be easily differentiated at depths to 2400 m. Keeley et al. (2018)[86] and Pochon et al. (2020)[87] demonstrated that bacterial assemblages from fish farm environments in the Marlborough Sounds significantly differentiate along organic enrichment gradients (from fish cages to control sites), and that these patterns are repeatable and regionally transferable. They also demonstrated that the newly established eDNA-based bacterial index performed equally well to the 'gold standard' morphology-based approach that is currently employed in the Best Management Practice Guidelines[94] currently used for regulating benthic impact assessments of salmon farming in the Marlborough Sounds. Taken together, these studies demonstrate the ability of eDNA to discriminate biotic assemblages over much smaller spatial scales than one might first think.

The next logical question regarding eDNA is if the techniques can detect impacts. In a number of eDNA applications there is compelling evidence that the changes in biotic assemblages are easily detected by eDNA analyses; these have been trialled (mainly with bacteria) around fish farms and oil & gas operations[95,96,97,98]. Perhaps the more relevant question to aquaculture is not if there will be changes in biota, it is if the observed changes in biota are disturbing foodwebs and ecological networks in an unsustainable manner, or well beyond the boundaries of the predicted impact. The developing area of eDNA that may help to address these kinds of questions in future is using eDNA in network analyses. Such analyses seek to better understand the level of 'connectedness' between taxa. This kind of network/foodweb work feeds directly into the concept of EBM.

The field of eDNA is starting to generate temporal datasets that, like spatial analyses, allow us to characterise variation in eDNA over seasons and years. Berry et al. (2019)[99] track the biotic responses to a 5-year chrono-sequence in Western Australia and were able to map the impact of a marine heatwave event and found significant assemblage changes seasonally, but not month to month. Consistent with patterns seen using macrofauna around fish farms, Pochon et al. (2020)[87] found for both bacteria and eukaryotes analysed in eDNA samples, that the biotic fluctuations year-on-year were minor in comparison with the changes expressed due to organic enrichment. As longer time series of data are analysed, the results will improve our understanding and provide a measure of certainty of the use of eDNA analysis as a long-term biomonitoring technique.

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[89] Pochon X, Zaiko A, Fletcher LM, Laroche O, Wood SA. (2017). *PLoS ONE 12(11)*: e0187636. https://doi.org/10.1371/journal.pone.0187636

[90] Knapp M, Spencer HG, Lamare MD, Taylor HR, Stat M, Bunce M, Gemmell NJ. (2018). *Molecular Ecology Resources* https://doi.org/10.1111/1755-0998.12982

[91] West KM, Stat M, Harvey ES, Skepper CL, DiBattista JD, Richards ZT, et al. (2020). *Molecular Ecology 29 (6)*: 1069–1086.

[92] DiBattista JD, Reimer JD, Stat M, Masucci GD, Biondi P, De Brauwer M, et al. (2020). *Scientific Reports 10 (1)*: 1–15. https://doi.org/10.1038/s41598-020-64858-9

[93] Jeunen G-J, Lamare MD, Knapp M, Spencer HG, Taylor HR, Stat M, et al. (2020). *Environmental DNA 2 (1)*: 99–111.

[94] Keeley N, Gillard M, Broekhuizen N, Ford R, Schuckard R, Urlich S. (2019). *New Zealand Aquatic Environment and Biodiversity Report No. 219*. 48 p. https://fs.fish.govt.nz/Page.aspx?pk=113&dk=24708

[95] Dowle E, Pochon X, Keeley N, Wood SA. (2015). *FEMS Microbiology and Ecology 91(8)*: fiv089. DOI: 10.1093/femsec/fiv089

[96] Keeley N, Wood SA, Pochon X. (2018). *Ecological Indicators 85*: 1044–1057.

[97] Laroche O, Wood SA, Tremblay LA, Ellis JI, Lear G, Pochon X. (2018). *Marine Pollution Bulletin 127*: 97–107.

[98] Stoeck T, Frühe L, Forster D, Cordier T, Martins CI, Pawlowski J. (2018). *Marine Pollution Bulletin 127*: 139–149.

[99] Berry TE, Saunders BJ, Coghlan ML, Stat M, Jarman S, Richardson AJ. et al. (2019). *PLoS Genetics 15(2)*: e1007943.

In the context of aquaculture what would the advantages and limitations be when adopting an eDNA approach?

There are advantages and limitations to all bio-survey methods, eDNA is no exception. Most commonly, the use of eDNA analysis is compared with morphological methods. Inherently, some of the comparisons between eDNA and morphology are valid, and others are not. For example, there could never be a morphology-based bacterial index of aquaculture impacts due to inherent difficulty in identifying bacterial species based on morphology. Some of the advantages, limitations, and consideration of an eDNA approach relative to other bio-survey methods are given below.

1. Cost. It is estimated that eDNA analyses (per sample) could be done cheaper than morphological analysis, particularly where sites have diverse and abundant assemblages. eDNA analyses will likely get cheaper with scale and automation (Pochon et al. 2020)[87].
2. Time. eDNA is processed and analysed in about 4 weeks; this is historically quicker than for morphological analyses by providers in New Zealand. Furthermore, bottlenecks caused by limited available expertise in taxonomy (see next point) when large numbers of samples need to be analysed will be reduced when employing eDNA analyses, substantially increasing the turnaround time on results from large sampling campaigns.
3. Expertise. The digital nature of eDNA means that although it does require specialised analysis and bioinformatic expertise (which are transferrable to other areas), this form of analysis does not require the highly specialised taxonomic expertise needed for morphological methods.
4. EBM. eDNA analysis is able to satisfy the core concept of EBM in that it is able to look holistically at a broader range of biota within the ecosystem. In practice, studies will typically restrict their taxonomic scope (to bacteria, for example) there remains an option to explore a wide range of taxa.
5. Quantitativeness. There are differing opinions on how quantitative eDNA data can be. Because eDNA is shed at different rates by different organisms, it is likely that eDNA will be largely restricted to presence/absence or rank abundance. Experimental design will ultimately dictate the quantitative nature of eDNA data and how eDNA data might be integrated with morphological data.
6. Size and life history. eDNA is not able to determine the size or life history characteristics of a given animal; such data may be important in understanding impacts.
7. Ease of sampling. eDNA samples (sediment and water) can be collected rapidly. There are preservation methods that allow for room temperature storage and shipping.
8. Archiving. DNA extracts, filtered water, and small sediment samples can easily be archived for future study.
9. Data storage and visualisation. eDNA data can be easily updated when new taxonomy is established, this is less easily achieved for other forms of monitoring.

Environmental DNA is one of the most promising new tools in the bio-surveying 'toolkit'. The key decisions will be how best to integrate it into existing aquaculture practice and provide proof to decisionmakers and the broader community that it can deliver relevant information to enact best-practice stewardship[100].

[100] Darling JA (2020) Aquatic Ecosystem Health and Management https://doi.org/10.1080/14634988.2019.1682912

What eDNA approaches could be used in an open ocean aquaculture setting?

As guidelines are put in place around consenting and monitoring of aquaculture facilities in open ocean sites, it is perhaps useful to speculate on what an eDNA programme of work might look like. This builds on the inshore eDNA work already conducted in New Zealand and elsewhere.

Firstly, the collection of eDNA from sediment, water, and plankton samples from around the proposed open ocean aquaculture sites will potentially set up a record of all the biota that live in and around the consenting area. This biotic baseline will facilitate a powerful lens into the biota at proposed sites and associated reference sites. This sampling could occur every 3–4 months (for at least one year) to better understand how seasons impact on the biota.

If consents are issued, it may be prudent to collect 'foundation' samples that act as a permanent (genetic) record of any given site prior to development. Indeed, these environmental samples may never be analysed and may be stored pending an event which may trigger an investigation. Such archiving approaches are being used in the oil/gas sector as part of oil spill response plans.

Aquaculture facilities offshore act as artificial reefs and thus need to be monitored for invasive species and interactions with wildlife (e.g., sharks and birds). One of the benefits of eDNA is that a variety of questions can be asked of a single sample from the species of birds, fish, and/or marine mammals that are attracted to a farm, to the environmental microbes that may effectively act as barometers of change. This adaptability is useful because it is often difficult to predict *a priori* which environmental issue(s) may arise at a given location.

In open ocean sites which are high flow and deep, the biggest ecological concern may be diffuse nutrient wastes in the water column. In these cases, the use of eDNA transect lines may help to inform the impact area in the water column and help to inform/refine the monitoring strategies.

APPENDIX 5: GENERAL CONSIDERATIONS AND LOGISTICAL CHALLENGES FOR WATER QUALITY MONITORING IN OPEN OCEAN ENVIRONMENTS

General considerations for water quality monitoring in open ocean environments:

- We acknowledge that understanding the regional, broader-scale environment is of wider interest than just aquaculture and will require efforts from outside the aquaculture industry. However, as open ocean aquaculture is entering a new type of environment, the industry will have to manage their effects in this environment.
- In regions where residual transport speeds are high, it will not be possible to undertake regular mapping surveys across the full extent of the area in which any phytoplankton response might occur.

Likely state of existing information on open ocean environmental conditions:

- Existing historical/background water quality data (e.g., research or SOE monitoring data) will be limited in the region. Preliminary survey data collected to support the consent application are likely to be temporally and spatially sparse, which may pose challenges for characterising the mean state and the scale of natural spatio-temporal fluctuations.
- Existing historical/background data concerning any contaminants of concern are likely to be limited. However, in the absence of other nearby sources of those contaminants (or precursor molecules), the limited data available indicate that ambient concentrations will be negligible and undetectable.

General logistical challenges for water quality monitoring in open ocean environments that will make monitoring more difficult than in inshore environments are:

- It is unlikely that the plume (i.e., farm footprint) can be reliably predicted at any point in time from sampling more than a few tens of metres from the perimeter of a pen if sampling is restricted to collecting discrete water samples due to the high natural variability inherent in water quality indicators. Thus, there is a high probability that individual samples may be gathered from outside the plume and that the plume will be missed[101], perhaps even if sampling within a few hundred metres of the pens. Remote sensed data (e.g., satellite imagery, also see Appendix 6) may assist with identifying the plume for some water properties (but even then, will likely be restricted to identifying time-average plumes).
- Some water samples must be quickly chilled, filtered, and frozen. The transfer times between collection and delivery to the laboratory may be long in comparison with those associated with some inshore sites. Filtration and freezing may need to be undertaken on the vessel (or, perhaps, as soon as the vessel returns to shore) rather than at the laboratory.
- Biofouling is an issue whenever instruments are left at sea for more than a few days-to-weeks. We anticipate that it will be a lesser problem in open ocean environments than in some of the inshore farming regions. Nonetheless, deployed instruments will need to be cleaned and serviced regularly. The initial servicing interval should be set conservatively, possibly monthly. It may be possible to extend this to several months (subject to battery life and other factors).

[101] Because the plume can be episodic, and shifts according to water flows, a grid-type sampling design to collect water grab samples is considered highly impractical in an open ocean environment.

APPENDIX 6: MONITORING OPEN OCEAN AQUACULTURE WITH SATELLITE-BASED SENSORS – A TECHNICAL GUIDE

This appendix was prepared by Ben Knight and Dana Briscoe (Cawthron) with external review from Moritz Lehmann (Xerra Earth Observation Institute Ltd) and subject to a review from Rich Ford (Fisheries New Zealand) with some input from other working group members.

The purpose of this appendix is to provide a brief background on satellite remote sensing of oceanic environments around open ocean aquaculture (OOA) sites, specifically those concerned with finfish farming. We focus on passive sensing from optical satellite-based sensors for the purposes of environmental monitoring for effects from such OOA sites and their environs. As noted in the body of this report (sections 8.1, 8.2.1.2, 8.2.2.2, Table 5, Table 6) and Appendices 5 and 7, single monthly samples for water column properties like chlorophyll *a* can be of very limited use on their own for detecting effects from finfish farms. These limitations can be due to the patchy nature of the phytoplankton communities associated with the chlorophyll *a* pigments and the relatively small volumes analysed through traditional sampling techniques. To improve future monitoring of OOA activities and their potential effects on water properties like chlorophyll *a*, some form of regular, wide-scale monitoring could be used. Satellite-sensed measurements provide a cost effective and accessible solution to this problem.

In comparison to the ongoing expense associated with *in situ* monitoring, remotely sensed products are often freely available and more easily accessible. However, although the data may be available for free, there are still substantial costs in developing and ensuring that such data are valid for use in a monitoring programme. Despite such costs, there are additional benefits from satellite monitoring, including: access to long series of records of measurements, the ability to monitor at large spatial scales, and temporal sampling at frequencies that may be difficult or impossible for distant sites.

Here we focus on surface chlorophyll *a* estimates as a proxy for phytoplankton biomass. This is because excessive increases in nutrients (eutrophication) from open ocean fish farms can cause increases in phytoplankton biomass and primary productivity including potential increases the magnitude and frequency of harmful algal blooms (HABs). Although chlorophyll *a* is a focus, there is also the potential for other key marine properties of relevance to OOA monitoring to be investigated by optical satellite sensors, namely:

- sea surface temperature (SST);
- surface turbidity;
- phytoplankton taxa classification (e.g., harmful algal species detection);
- large wildlife interactions (e.g., whales).[102]

Although sea surface temperature and surface turbidity can be influenced by OOA operations, monitoring of these properties would be more usefully directed to detect events that are potentially harmful to the OOA operations and the health of farmed fish than the reverse, i.e., effects on the wider environment. Therefore, we do not focus on these properties, but note that changes in SST or turbidity could act as a cumulative stressor, potentially exacerbating OOA impacts to the OOA activities and the environment.

[102] Information on the spatial and temporal distribution of highly mobile marine species may be possible using high-resolution satellite imagery; however, such information is episodic and often difficult to obtain. Therefore, satellite-derived information may only be useful to supplement other information on water monitoring methods.

The effects of some harmful phytoplankton species can also present risks to both wild species (e.g., Jessup et al. 2009, Rountos et al. 2014) and cultured fish at a site (Chang et al. 1990, MacKenzie et al. 2011), even at low concentrations (Montes et al. 2018). Therefore, identification of phytoplankton species (or taxa), which is not currently routinely possible using remote sensing, may be valuable for both monitoring potential effects from OOA and ensuring the health of the cultured species.

Background to ocean colour sensing with optical sensors

Optical satellite sensors are passive in that they simply measure incoming electro-magnetic radiation from their field of view and operate in a similar way to a camera but are able to measure a greater range of colours (i.e., they are multi- or hyper- spectral). Optical sensors differ from active sensors such as synthetic aperture (SAR) or laser-based (LiDAR) sensors, which also emit the energy that they are sensing. In this section we only consider passive optical satellite sensing.

Optically-sensed satellite products (e.g., chlorophyll *a* estimates) typically use estimates of remote sensing reflectance, which is defined as the ratio of surface-leaving upwelling radiance to downwelling irradiance at the water surface. The colour of water[103], often called ocean colour, is defined as the spectrum of remote sensing reflectance in the visible wavelengths. The colour of water at the water surface is a function of the optical properties of water and its constituents, namely adsorption, absorption, and scattering of light.

Pure water absorbs light most strongly at long visible wavelengths (e.g., red light) and scatters most strongly light at shorter visible wavelengths (blue). In clear ocean water these two processes dominate, making the water appear blue. Other constituents in seawater, including phytoplankton and their photosynthetic pigments, suspended sediments, inorganic particulates, and coloured detrital matter have different scattering and absorption properties, giving water hues from blue-green to yellow, brown, and red (Figure 8). Red tide events represent an example of an extreme event where the absorption of light by high concentrations of phytoplankton make the water appear red.

The observation of ocean colour from a remote sensor is also affected by its transmission from the water surface through the atmosphere; that is, the sensor measurements are top-of-atmosphere measurements. Therefore, atmospheric conditions, the angles of solar reflectance and sensor viewing, and solar irradiance can all affect the colours observed at the satellite-based sensor that originated at the water surface. These atmospheric effects can create uncertainty in the reconstruction of the remote sensing reflectance at the water surface. Therefore, atmospheric correction of sensor collected data presents an important step in the production of accurate remote sensing reflectance estimates.

[103] We use the term colour to refer to the wavelengths of visible light with blue light having a wavelength of around 400 nm and red having longer wavelengths of about 700 nm. However, in the context of satellite sensors, here we also use the term colour to refer to wavelengths outside the human-visible spectrum which could include ultra-violet (<300 nm) and infra-red (>750 nm) wavelengths.

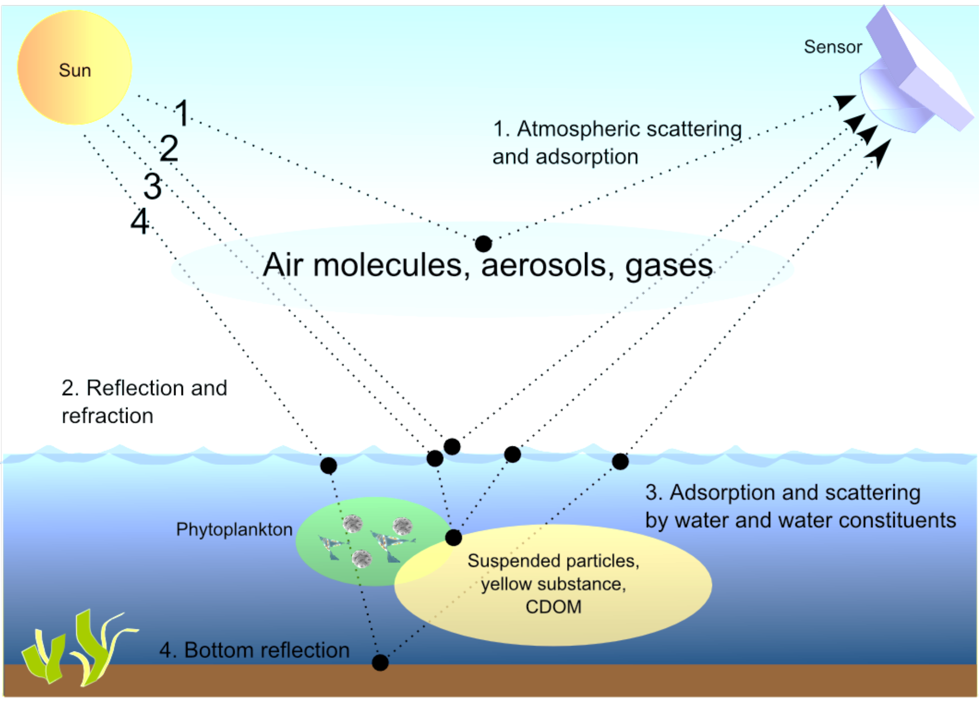

Figure 8: Satellite remote sensing in a marine environment. Pathways of light detected by the remote sensor: 1) Light scattered and reflected in the atmosphere; 2) Reflection and refraction from the sea surface; 3) Surface water-leaving irradiance[104] (dependent on inherent optical properties (IOPs) of seawater); 4) In shallow coastal environments, light signal altered by reflectance of seabed. Adapted from Sathyendranath et al. (2000) by Jones et al. (2013).

When considering the estimation of biological properties in water, a continuum exists between two categories of water: Case 1 and Case 2 waters (Mobley et al. 2004). Case 1 waters are defined as waters where the optical signatures of phytoplankton dominate. These waters tend to be offshore areas, located a long way from the coast. The optical signatures in Case 2 waters are dominated by non-biological material (e.g., sediment backscatter) and are generally coastal waters affected by river and land inputs. Localised algorithms may need to be derived for these more optically-complex waters to calculate chlorophyll *a*.

Satellite products can vary from raw top-of-atmosphere information (level 1 products) to high-level gridded products of water properties, such as chlorophyll *a* or sea surface temperature (referred to as level 2, or higher, products). In Case 1 waters, global high level satellite products are often available for key water column properties. Although level 1 products are available for all satellites, higher level products may not always be available. Open ocean aquaculture sites are generally removed from coastal influences and may not always meet the definition of Case 1 waters. Therefore, even if high level Case 1 products are available, local validation is still recommended.

If preproduced data products are not available for a particular satellite sensor, the locations and widths of wavelength bands from the sensor may be important to consider before deciding which sensor data will be useful for OOA monitoring. For example, the calculation of specific water properties such as chlorophyll *a* often rely on the ratio of absorbed light from particular colours (Figure 9). This information can be used to construct a locally-validated model for the estimation of chlorophyll *a*, provided a suitable quantity of *in situ* data is available to train it (e.g., see Jiang et al. 2017).

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[104] Note that the water-leaving signal received by the sensor is composed of up to 80% contribution from the top-of-atmosphere signal, therefore atmospheric correction is important (Loisel et al. 2009).

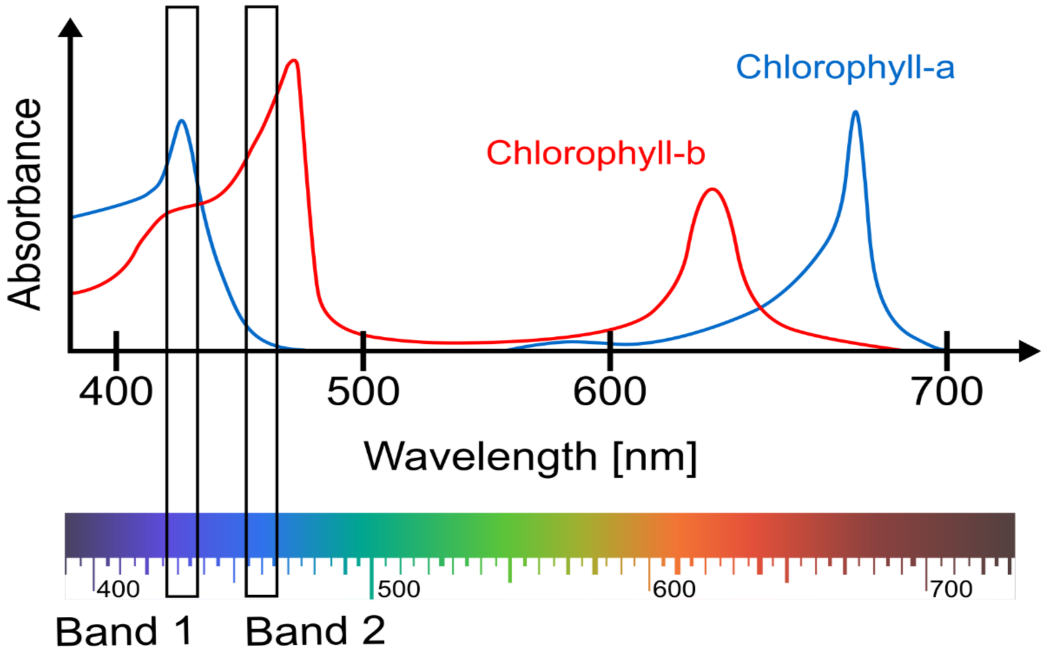

Figure 9: **Visible wavelengths (390 nm to 720 nm) (bottom) in comparison to the absorbance spectra for phytoplankton pigments, chlorophyll *a* and *b* (top). Black boxes labelled Band 1 and 2 (B1 and B2) indicate example wavelengths and spectral widths of satellite sensor measurements that could be used to infer chlorophyll *a* concentrations. In this case, the ratio of B2 to B1 would increase under increasing chlorophyll *a* concentrations (where more B1 is absorbed). Modified from Creative Commons image on Wikipedia[105].**

As well as the spectral band considerations, there are several other factors that are relevant to the use of satellite data products for aquaculture monitoring. These factors, along with examples of the types of questions that could be asked, are:

1. Length of mission. Has the sensor operated for at least a year or more (so any initial faults are rectified and baseline data from that satellite exist)?
2. Availability of high-level products. Are there existing products available for chlorophyll *a*, or will they need to be developed?
3. Temporal resolution. Will the data be provided at least daily?
4. Cost. Is it freely available?
5. Spatial resolution. Will the user be able to resolve the OOA site in the data?
6. Signal to noise ratio of measurements. Are the data precise enough to detect potential levels of expected change associated with aquaculture?
7. Accessibility and use. How will the data be accessed and processed?

These are not discussed in detail here, but we consider that the first five considerations in this list are probably the most important for the purposes of aquaculture monitoring.
New Zealand/Aotearoa is the "land of the long white cloud" and cloud cover can be particularly problematic in some areas. Therefore, cloud cover at the site of interest and revisit interval of the

[105] https://en.wikipedia.org/wiki/Chlorophyll_a#/media/File:Chlorophyll_ab_spectra-en.svg (Accessed 28/9/2020)

satellite are important to consider before selecting a sensor for use in OOA monitoring. For instance, a site that has significant cloud cover for 50% of the year will require sensor data with a frequent revisit period (a day or less) to ensure that a sufficient quantity of data can be provided for the purposes of monitoring.

Types of satellites for monitoring Open Ocean Aquaculture

There are several satellite platforms that meet the criteria posed above. Broadly, we consider four main types of satellites: those with moderate resolution sensors, high resolution sensors, hyperspectral sensors, and geostationary satellites.

Moderate resolution sensors
Moderate resolution refers to sensors with an on-water spatial resolution of between about 250 m and 1 km. Such instruments often provide data for several (e.g., 10 or more) narrow ocean colour bands and often include far-infrared thermal bands for SST estimation. These sensors are typically designed for ocean sensing applications and therefore typically present the most promising candidates for OOA monitoring.

Examples of moderate resolutions sensors are the Moderate Resolution Imaging Spectroradiometer (MODIS) sensors on the NASA Aqua and Terra satellites and the Sentinel 3 Ocean and Land Colour Instrument (OCLI). The MODIS sensors have collected daily data since 2002, with many high-level global products for Case 1 waters produced by the NASA Ocean Biology Processing Group (OBPG). These products are available at coarse 4 km and 9 km spatial resolutions through the Ocean Color website[106].

High resolution sensors
The NASA Landsat series of satellites represent an example of high-resolution sensors and have a long history of use for remote sensing on land. They also have full coverage over coastal areas of New Zealand and can extend tens of kilometres offshore, with continuous and regular coverage available since 1990. Their revisit intervals for consecutive scans are quite long, at 16 days, which can limit their use for monitoring during periods with high cloud cover. However, the recent arrival of the European Space Agency (ESA) Sentinel 2 (A and B) satellites has added a much-improved revisit frequency to freely available high-resolution data, with updated imagery provided every 3–5 days at 10 m resolution.

The main benefit of such high-resolution images is the potential to elucidate fine-scale changes close to aquaculture operations. However, in these sensors often the number and width of spectral bands are limited, which limits both the number of products and the accuracy that may be able to be achieved for ocean applications. In addition, because these high-resolution images generally have a terrestrial focus their spatial coverage may not cover distant OOA sites.

Hyperspectral sensors
Hyperspectral sensors measure light at hundreds of regularly-spaced spectral bands (e.g., 5 nm) in the visible and near-infrared part of the spectrum, rather than the tens of discrete bands common to most current multispectral satellite sensors. This additional spectral information offers the potential to investigate subtle optical changes that could be associated with specific phytoplankton species or groups (e.g., novel pigment absorption or fluorescent emissions). Such information is a relatively new use of the technology and, although some examples exist internationally (e.g., Bracher et al. 2017, Mouw et al. 2017), some research effort would be required to validate this for New Zealand. However, this may enable elucidation of phytoplankton species or genera of interest in future.

[106] https://oceancolor.gsfc.nasa.gov/ (Accessed 28/9/2020)

The HICO mission[107] is an example of a hyperspectral sensor that operated from 2009 to 2014. The Italian Space Agency has also recently launched a hyperspectral sensor called PRISMA[108], which has an on-ground resolution of 30 m, a 30-km wide swath and a revisit frequency of a week to a month. The upcoming PACE mission[109] (planned launch in November 2022) also represents an opportunity to collect additional hyperspectral data.

Geostationary satellites
Geostationary satellites are stationary with respect to their location on the earth surface. This enables multiple measurements per day over large areas and can potentially provide information from areas where cloud cover may have prevented observation from rapidly moving satellites. Geostationary satellites are typically focused on infra-red wavelengths for weather observations. Although the use of such data for ocean sensing purposes is not common, the far-infrared wavelength data have now been used for some years to supplement other sea surface temperature information, including the 'cloud-free' SST products produced by the Group for High Resolution SST (GHRSST)[110].

The new generation of such satellites (e.g., Himawari 8, GOES-17) also includes sensors with the capability to measure visible colour spectra. The Japan Aerospace Exploration Agency (JAXA) P-Tree system[111] shows that such data from the Himawari 8 satellite can be converted into useful ocean products, such as chlorophyll *a*, at resolutions of 1 km and up to 10-minute intervals. The value of near continuous near-real time temperature and chlorophyll *a* data may become very useful for future monitoring of OOA sites.

We summarise the potential value of each of these potential satellite sensor types for some exemplar sensor platforms in Table 11.

107 http://hico.coas.oregonstate.edu/ (Accessed 28/9/2020)
108 http://www.prisma-i.it/index.php/en/ (Accessed 8/10/2020)
109 https://pace.gsfc.nasa.gov/ (Accessed 28/9/2020)
110 https://www.ghrsst.org/ (Accessed 28/9/2020)
111 https://www.eorc.jaxa.jp/ptree/index.html (Accessed 28/9/2020)

Table 11: Qualitative assessment of suitability of a range of sensors for deriving water properties (from best * to worst *) and their potential accuracy (box shading).**

Satellite sensor type	Example sensor	Spatial resolution (m)	Revisit interval (days)*	Operational period	Sea surface temperature	Surface turbidity	Surface chlorophyll *a*	Surface phytoplankton taxa[b]
Moderate resolution	MODIS	250–4000	1 day	2002– Present	***	***	***	*
Moderate resolution	VIIRS	375–1000	1 day	2011–Present	***	***	***	**
Moderate resolution	Sentinel 3	300–1200	1–3 days	2016 Present	***	***	***	**
High resolution	Sentinel 2	10–90	5 days	2015–Present	*NA*	***	**	*
High resolution	LandSat 8	10-90	16 days	2013–Present	***	***	**	*
Hyperspectral	PRISM	30	7–30 days	2019–Present	*NA*	***	***	***
Geostationary	Himawari 8	1000	10 min	2014–Present	***	***	**	*

High accuracy possible after validation	Moderate accuracy possible after validation and algorithm development	Low accuracy likely; improvement possible with research and algorithm development	Poor accuracy likely, even with validation

* Although the revisit interval is given here, some sensors appear on multiple satellites (e.g., the Sentinel 2 a and b, or the MODIS Aqua and Terra satellites); therefore images may be available at a greater frequency than would be determined from the revisit interval.

Data for most of the sensors presented in Table 11 can be obtained through several sources, such as:

- NASA Ocean Color (https://oceancolor.gsfc.nasa.gov/ - MODIS, VIIRS)
- PO.DAAC (https://podaac.jpl.nasa.gov - various datasets, including GHRSST products)
- USGS Earth Explorer (https://earthexplorer.usgs.gov/ - LandSat, Sentinel 2)
- Copernicus Open Access Hub (https://scihub.copernicus.eu/ - Sentinel 1, 2 and 3).

However, regardless of the source of the satellite data or derived product, comparison with *in situ* measurements is strongly recommended to quantify the accuracy and utility of such products. Therefore, some technical guidance on the collection of *in situ* measurements is also provided in the following section.

An overview of the science of optical satellite remote sensing, potential sources of data, and the requirements to ensure its use in monitoring of OOA is provided here. However, there is potentially much more detail that could be provided. For those wishing to know more, we recommend the freely available Earth Observation series (Harrison et al. 2016) published by the Australian and New Zealand Cooperative Research Centre for Spatial Information (CRCSI) and which can be found here: https://www.crcsi.com.au/history-2/earth-observation-series-2/.

Collecting data for the validation or development of new satellite products

Validated satellite data can reduce ongoing costs of, and improve, OOA monitoring and are typically used to supplement a wider monitoring programme. However, if a greater reliance is placed on the accuracy of satellite data over other methods of monitoring, then ensuring the accuracy of satellite data becomes very important. Validation and checking of the data can be a resource-intensive undertaking, therefore careful planning to maximise the value of collected *in situ* data for validation is required.

There are many aspects to consider when collecting *in situ* data for comparison with satellites; broadly, these include:

- The optical properties of oceanic water can change throughout the year due to coastal water influences, sun angles, and biological changes; therefore initial *in situ* validation data should be collected for at least a year and would ideally be available on an ongoing basis for future confirmatory checks.
- Data collection within the water is often undertaken at much smaller spatial scales than can be resolved by satellites, so the use of continuously sensing instruments (e.g., logging fluorometers for chlorophyll *a*) is recommended to remove artifacts of patchiness in the property of interest (e.g., phytoplankton).
- Long deployments of continuously running equipment can be prone to fouling and such instruments will require regular cleaning and checking to independent lab measurements.
- Satellites measure surface water properties, therefore *in situ* measurements should ideally be located at the surface or be representative of surface conditions where this is not possible (i.e., be located less than 10 m from the surface and above any strong density gradients).
- Conductivity temperature depth (CTD) instrument measurements should be undertaken beside any sensors to ensure that a large density change (i.e., a pycnocline) is not located between the submerged instrument and the surface[112].

[112] Where temperature is the main driver of density differences, temperature information (e.g., from a thermistor) at the depth of the sensor may provide suitable information to infer the location of any density changes, reducing the need for CTD measurements.

- Fluorescent-derived chlorophyll *a* measurements are prone to non-photochemical quenching whereby high light levels (associated with surface waters) can reduce the fluorescence response of chlorophyll *a* to excitation by the sensor[113]. Therefore, careful consideration should be given to the deployment depth to reduce the effect of quenching while still resolving surface water properties. Typically, depths of 5 m to 10 m will be sufficient, but this depends on the clarity of the water, latitude, and the time of year.

Once satisfactory *in situ* data have been collected, there are several tools and techniques available for data processing. Selecting the appropriate level of processing can save considerable time and resources. For some purposes, the use of quality-controlled datasets such as those available from NASA's Ocean Color website provide a suite of pre-processed and off-the-shelf products[114]. Many of these data are readily available to use through commonly-used programming languages and software packages such as Matlab, R, Python, IDL, ArcGIS, QGIS, WorldView (Alpha), SNAP, and SeaDas. Subsetting for specific locations from large global datasets (to reduce the amount of data downloaded) is also possible through the use of OpenDAP services[115] from providers such as NASA's Ocean Color[116] or Physical Oceanography Distributed Active Archive Center (PO.DAAC)[117] websites.

Although pre-processed products can provide robust estimates for the marine environment (i.e., Case 1 waters), remotely sensed products used in Case 2 waters may require more effort to process, due to the interference of sediment and other non-algal particulates. In New Zealand, previous research suggests that even in the oceanic areas of the continental shelf, the accuracy of Case 1 chlorophyll *a* algorithms can still be affected (Pinkerton et al. 2005). However, recent improvements to include the colour index approach of Hu et al. (2012) for global chlorophyll *a* estimation by NASA's OBPG has occurred since the Pinkerton et al. (2005) study was published[118]. Regardless, some form of validation will still be required. At the most basic level, trusted *in situ* data can be used to validate, or empirically adjust, preproduced data products to ensure they are fit for the purposes of monitoring OOA sites.

The next level of analysis is to consider retraining a new algorithm from satellite reflectance data to collected *in situ* data, as was undertaken for chlorophyll *a* estimates by Jiang et al. (2017). The benefit of this approach is that it allows a wider range of satellite platforms to be used (e.g., fine-scale Sentinel 2 data) or even improved products from an existing satellite platform (e.g., 1-km resolution data, rather than preproduced level 3 data for MODIS that has 4-km resolution).

Advancements in the development of analytical or semi-analytical approaches also have the potential to provide a better understanding of the inherent optical properties and can improve estimates of biological properties, such as chlorophyll *a* (e.g., Aurin & Dierssen 2012). However, this approach has some challenges in marine environments (e.g., Werdell et al. 2018) and requires a greater level of measurement to validate *in situ* optical and biogeochemical observations, before they can be broadly applied (Jones et al. 2013). Alternative approaches using machine learning, though a relatively recent focus of research, also appear promising for estimation of properties such as chlorophyll *a* (Pahlevan et al. 2020).

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[113] Typical florescence sensors work by exciting chlorophyll *a* with a high energy light source (e.g. blue or green light) and measuring the resulting fluorescent emission of lower energy (red) light.
[114] https://oceancolor.gsfc.nasa.gov/l3/ (Accessed 30/9/2020)
[115] https://www.opendap.org/ (Accessed 30/9/2020)
[116] https://oceandata.sci.gsfc.nasa.gov/opendap/ (Accessed 30/9/2020)
[117] https://podaac.jpl.nasa.gov/dataaccess (Accessed 30/9/2020)
[118] The approach of Hu et al. 2012 is applied to low nutrient areas of the ocean (covering about 75% of the earth's ocean surface). See https://oceancolor.gsfc.nasa.gov/atbd/chlor_a/ for more detail (Accessed 15/10/2020).

Summary

Satellite remote sensing can provide a cost-effective tool for the long-term detection and monitoring of the oceanic environments around OOA sites more frequently and over greater areas than could be achieved by traditional water monitoring. An initial investment is required to ensure the accuracy of satellite sensed information through validation with *in situ* measurements. Such investments include framework development, expertise for data processing, and an operational budget. Satellite monitoring is not meant to replace traditional *in situ* monitoring and ideally will require ongoing collection of traditional measurements to ensure its accuracy. A hybrid approach to such monitoring is recommended, whereby continuously logging instruments are checked against lab measurements to ensure that the *in situ* data are accurate, but sufficiently resolved, for comparison with satellite sensor data.

Provided the satellite data can be validated, their potential for assessing impacts and placing them in the context of historical changes provides an opportunity for the monitoring of future OOA activities. This can provide information to stakeholders concerned that traditional monitoring is not sufficient to monitor the potentially large and remote areas that could be developed from OOA applications. Similarly, the use of the technology also provides an opportunity for the industry to more widely monitor their growing environments to enable better management (e.g., avoid mortality through the early detection of harmful algal blooms). Consequently, although some investment will be required to utilise this technology, the potential benefits for the environment and industry make the use of satellite data valuable.

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APPENDIX 7: BIOGEOCHEMICAL PARAMETERS AND ANALYTICAL PROCEDURES FOR WATER QUALITY MONITORING

Total nitrogen (TN)

Total Nitrogen is the sum of all molecular species of nitrogen other than N_2 (which is gaseous). The specific laboratory test for TN amounts to summing ammoniacal N, nitrate and nitrite N, and dissolved organic N and particulate N.

Particulate nitrogen (PN)

Particulate Nitrogen is the nitrogen bound within particulates. Strictly, there are different laboratory tests for PN and PON (particulate organic N). In practice, the two measures are very tightly correlated with a near 1:1 relationship. That is, they are measuring the same quantity. The NIWA laboratory considers the PN test to be a little more reliable because it involves fewer manual steps (M. Crump, NIWA, pers. comm.).

The test for particulate nitrogen also yields particulate carbon. The particulates are filtered out of the water to be measured. Particulate concentrations measured in different samples will only be comparable if they are all measured using filters with the same pore-size distribution. In coastal monitoring it is common to use glass fibre filter with a nominal pore size of 1.2 µm (GF-C) or 0.7 µm (GF-F).

If monitoring of particulates is required, a pore size should be specified so results are consistently analysed, and therefore comparable. This is especially important if there are associated compliance thresholds.

Chlorophyll

Chlorophyll is a photosynthetic pigment found within phytoplankton (and many other plants). Strictly, there are several forms of the chlorophyll molecule. Laboratory analyses usually measure chlorophyll *a*. Chlorophyll is a protein that contains nitrogen. Thus, chlorophyll concentrations are usually somewhat correlated with particulate nitrogen.

Cell specific chlorophyll content varies across taxa and even within individual cells across time (depending upon temperature, nutrient, and light availability, etc.). Water samples collected for laboratory determinations of chlorophyll must be chilled and stored in the dark until they can be filtered. The filters should be frozen until they can be analysed. As with other filtered particulates, it is important that standardised filters be used across all samples, so the filter size should also be stipulated for monitoring if chlorophyll measurements are required.

Chlorophyll concentration can be inferred from the intensity of fluorescence at particular wavelengths. This can be done in the laboratory or the field, though different methodologies and instruments are used for each. Before data from the laboratory method can be meaningfully compared with data from the field instrument, the two systems must be cross-calibrated on common water samples.

Although the water samples collected for laboratory analysis are necessarily small volume ‘spot samples’ (e.g., a bottle sample or a hose sample), the fluorimeter devices used in field determinations of chlorophyll can be used in profiling mode to map the vertical distribution of chlorophyll at a given location. Similarly, flow-through systems that permit continuous monitoring of near-surface chlorophyll whilst vessels steam at near-normal speeds can also be used to increase sampling coverage across an area. Such flow-through systems are becoming more readily available.

Some satellites carry instruments that measure relevant wavelengths of light emitted from the sea surface. See Appendix 6 for further information.

Dissolved oxygen

Oxygen sensors are readily available. Automated, logging instruments can be lowered over the side of the vessel to sample the vertical distribution of oxygen in the water column or left moored in place for periods of weeks to months (subject to biofouling). Hand-held sensors can also be used to measure oxygen *in-situ* for surface waters (spot measurements), or in water that has been brought aboard a vessel. For the latter, care must be taken to minimise loss/introduction of oxygen to/from the air between the time that the sample is collected and the time the measurements are made.