Aquatic
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Assessment of ecological effects of expanding salmon farming in Big Glory Bay, Stewart Island – Part 2 Assessment of effects

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Table of Contents (Part Two)

4. EFFECTS OF THE PROPOSED CHANGE TO SALMON FARMS ON THE RECEIVING ENVIRONMENT IN BIG GLORY BAY

4.1 History of farms

The observed and predicted effects of salmon farming on the receiving environment depend very much on the farm's history. In summary:

- Farm 249 was a mussel farm for 12 years before the pens previously at Farm 320 were shifted to the site of Farm 249 and it was converted to a salmon on-growing facility in 2011. In 2016 the site was vacated and left to fallow;
- Farm 246 was historically a mussel farm and since September 2016 has been Sanford's main on-growing farm;
- Farm 338 was a salmon smolt site from 1985 to 2012. Brood stock were moved to the site in December 2015;
- Farm 250 shifted to Farm site 339 in about 1987 and the farm became a smolt site for about 6 years until 1993. It was converted to a mussel farm from 1999 to 2013 and then back to a smolt farm from 2013 to 2018. The pens have recently been moved to Farm 340; and
- Farm 320 (sometimes referred to as the 47 South Site) was vacated after use for 2-3 years in September 2011 and shifted to Farm 249. The farm is now vacant.

Other sites may be used for salmon farming later but only once they have been modelled and it is demonstrated that they are sustainable.

4.2 Introduction to effects

Farming of Chinook salmon has been undertaken on a commercial scale in BGB since the 1980s.

The effects of aquaculture on the water and the benthic environment (from the Marlborough Sounds, Akaroa Harbour and Big Glory Bay) have been summarised in a number of comprehensive and recent reviews and reports including Forrest et al. (2007, 2015), MPI (2013b) and Keeley et al. (2009). This report presents a summary of the information contained in these comprehensive reviews, focusing on potential adverse effects. While most information presented in the previous reviews is relevant for all finfish aquaculture, the importance of specific effects differs among farms as a consequence of many factors, including farm location, farmed species, farming practices and the nature of the environment. The description of effects in this report assumes a standard of good farm management practice.

The main effects on the water column are the release of ammonia-N, potential enhancement of phytoplankton biomass, consumption of oxygen and effects of the structures on water movement.

Effects of caged salmon farming on the seabed are typically localised and reversible. Seabed enrichment from deposition of faecal material and uneaten food can result in low to highly impacted areas under farms, but little, if any, effect is observed beyond, at most, a few hundred meters of the farm boundary. The degree of impact will depend on the history, location, physical and ecological characteristics of the site (Forrest et al 2007). The general effects of finfish farming are illustrated in **Figure 19**. These effects are described and explained in more detail in Sections 4.5 to 4.10 in the following order:

- Water quality
- Seabed effects
- Biosecurity (marine pests and disease)
- Wild fisheries
- Marine mammals
- Seabirds

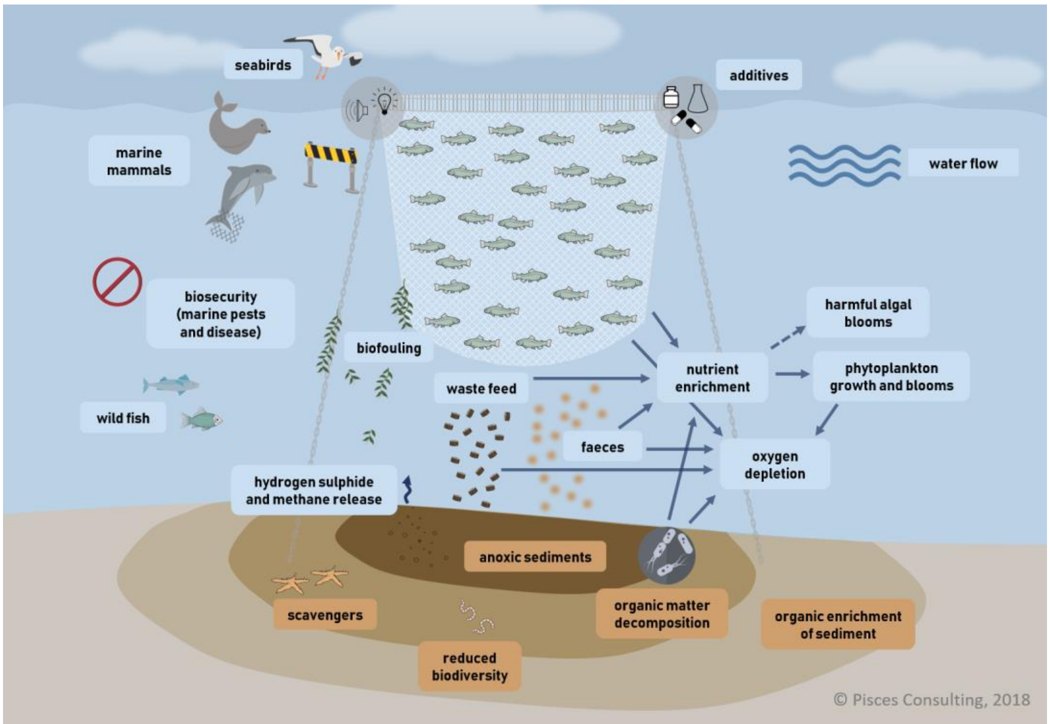


Figure 19: Illustration of the environmental effects of finfish aquaculture (provided by Hilke Giles).

In the following sections we expand on a description of potential effects, and then put them in the context of the proposed expansion in BGB. The existing environment described earlier in this report (Part One) provides information on what is found under and at varying distances from farms and the changes due to effects of the expansion of the farms is summarised in the relevant sections.

4.3 Relative importance and ecologically significant risk of effects

It is important to understand the relative importance and ecological significance of effects to ensure that activities are put into context for the assessment of effects. This report includes reference to some farm- or site-specific factors and, in some cases, mitigation options. While this provides some indication of relative importance, it is imperative that all effects are considered on a case-by-case basis.

The potential for, and severity of, environmental effects from a specific finfish farm depend on a wide range of factors, for example the species farmed, farm management practices, the scale and intensity of farming, the nature and resilience of the receiving environment, the level of knowledge of ecological issues, the scope to mitigate effects (for example by implementation of best management practices) and the scale and reversibility or recovery from effects. Note that for the purposes of this report recovery is to a state that the area can be farmed again.

The Ministry for Primary Industries convened an expert panel to rank the relative importance of potential effects of finfish farming in New Zealand (MPI 2013). The results (described as preliminary) indicate that the two most important effects (in decreasing order of importance) were biosecurity and pelagic effects. These effects were followed by marine mammal interactions, benthic effects, seabird interactions, additive effects, escapee effects, wild fish interactions and hydrodynamic alteration of flow. Clearly the relative importance of these effects will be site specific.

Forrest *et al*. (2015) took a risk-based approach to develop a monitoring framework for aquaculture in the Waikato region. This assessment indicated that biosecurity issues relating to marine pests was of high relative importance, largely because of the irreversibility of effects that may occur across regional scales. Water column nutrient enrichment was identified as a similarly significant issue because of its potential to contribute to broad-scale effects. The remaining effects were assessed as lower risk but Forrest *et al*. (2015) emphasised that they may nonetheless be regionally important.

These two assessments reveal that, in general, the two most important effects of finfish farming in New Zealand are biosecurity (marine pests) and water column nutrient enrichment. However, in the case of Stewart Island marine farming is only allowed in Big Glory Bay and, therefore, nutrient enrichment and sediment effects are likely to be the key effects but other effects, such as biosecurity and effects on mammals, also need to be taken into account.

4.4 Hydrodynamic model

As part of the assessment for the further development of salmon farming in BGB a hydrodynamic model was developed by ADS to assess the hydrodynamic regime as well as its flushing time (also referred to as “retention time”) (ADS 2017c). The results of these simulations were then used to drive water movement within the water quality module (see Section 4.5.1 – Water quality modelling).

The hydrodynamics model that was developed simulates water movement within BGB and across the entire model domain (ADS 2017c, **Figure 20**). The regional model is a two-dimensional model, while the local model is a three-dimensional model. Both models were run for one year to cover all four seasons.

To calculate residence times, a conservative (i.e. non-decaying) tracer was added evenly throughout the entire water column of BGB. The hydrodynamic model was then run to simulate how long it takes for the tracer to be flushed out of the Bay through water exchange with the ocean. Separate calculations were made for each season.

The hydrodynamic model was calibrated against ADCP data (current speed and direction) collected by the Cawthron Institute between 6 and 22 September 2010 at two locations within BGB (**Figure 20**) and water level from tidal harmonics in Paterson Inlet. The model was validated against another 14-day period of ADCP data (22 September to 6 October 2010).

At the mouth of BGB (location ADCP 1), the model matched current speed well, though it tended to slightly under-predict the flow. At the second location in the Bay (ADCP 2), the model simulated current speed well but performed relatively poorly in simulating current direction (ADS 2017 c). ADS ascribe the relatively poor current direction comparison for ADCP 2 to the flow being influenced by the numerous mussel farm and fish farm structures within BGB and it is beyond the capabilities of the model to take these into account.

The main findings of the modelling are:

- Depth averaged current speed is generally less than 5 cm/s, which is considered weak. Flow is stronger towards the mouth of the Bay;

- Current direction is variable and there are several eddies within the Bay. More eddies are likely to be generated by the mussel farms that the model could not identify;
- There is a general mouth-ward flow along the northern and southern banks of the embayment;
- The model demonstrated similar results for each season. In all four scenarios (seasons), the tracer concentration was less than 70% at all points in BGB at the end of day seven. By day 14, the tracer concentrations were approximately 40%. After 28 days, approximately 85-90% of the tracer concentration had been flushed out of the bay and the remainder disappeared over the next 20-30 days or so; and
- Flushing times are longer than those for Big Glory Bay calculated by Rutherford et al. (1988) who estimated them as 10-13 days during light winds and 7-9 days during moderate winds.

4.5 Effects on water quality

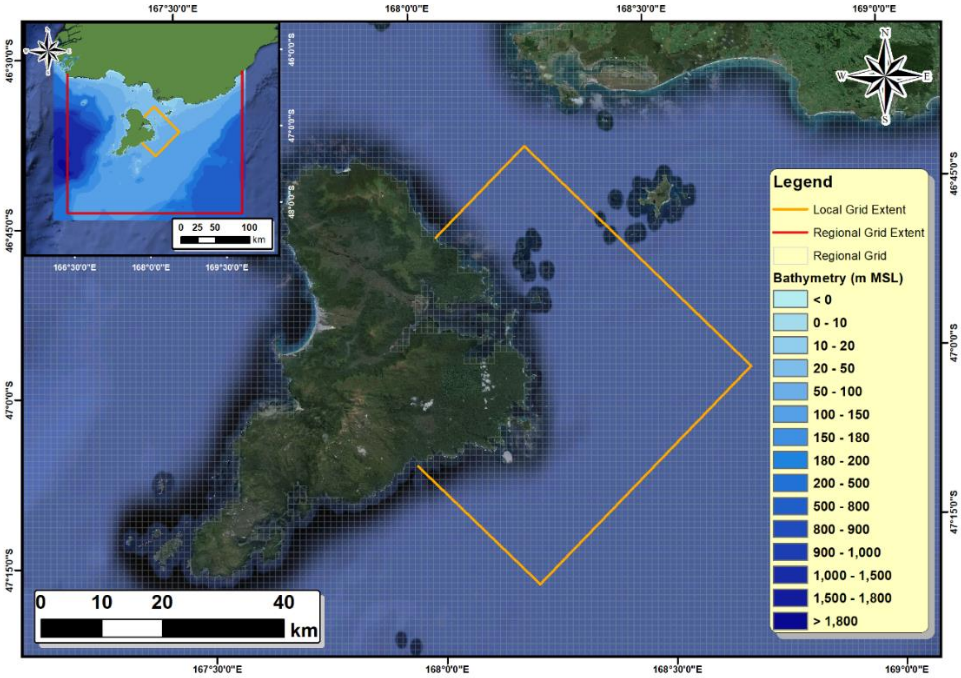
Finfish aquaculture affects water quality in several ways and depending on environmental and hydrodynamic conditions, water quality effects can potentially extend over large areas.

A particularly important issue is nutrient enrichment. Farmed fish excrete dissolved inorganic nutrients such as ammonium (NH_4). Nutrients (primarily nitrogen and phosphorus) are also released from waste feed, fish faeces and sediments (see benthic effects section).

Nutrient enrichment in the water column can stimulate phytoplankton growth (as a result of increased nutrient levels) and change the composition of phytoplankton species (as a result of changes in the ratios of nutrients), and potentially stimulating phytoplankton blooms. If nutrient levels are very high, the water body can become eutrophic. Eutrophic waters have high primary productivity due to excessive nutrients and are characterised by low water quality and frequent algal blooms.

Another important issue is the reduction of dissolved oxygen in the water column near finfish farms. Dissolved oxygen depletion is a consequence of increased biomass and microbial degradation that consume oxygen. Microbial degradation occurs in the water column (fish respiratory activities, degradation of phytoplankton, waste feed or fish faeces) and in the sediments (decomposition of deposited organic matter). Oxygen consumed in the sediments is replenished (if available) from the water column, thus leading to reduction of dissolved oxygen in the water column.

(A)



(B)

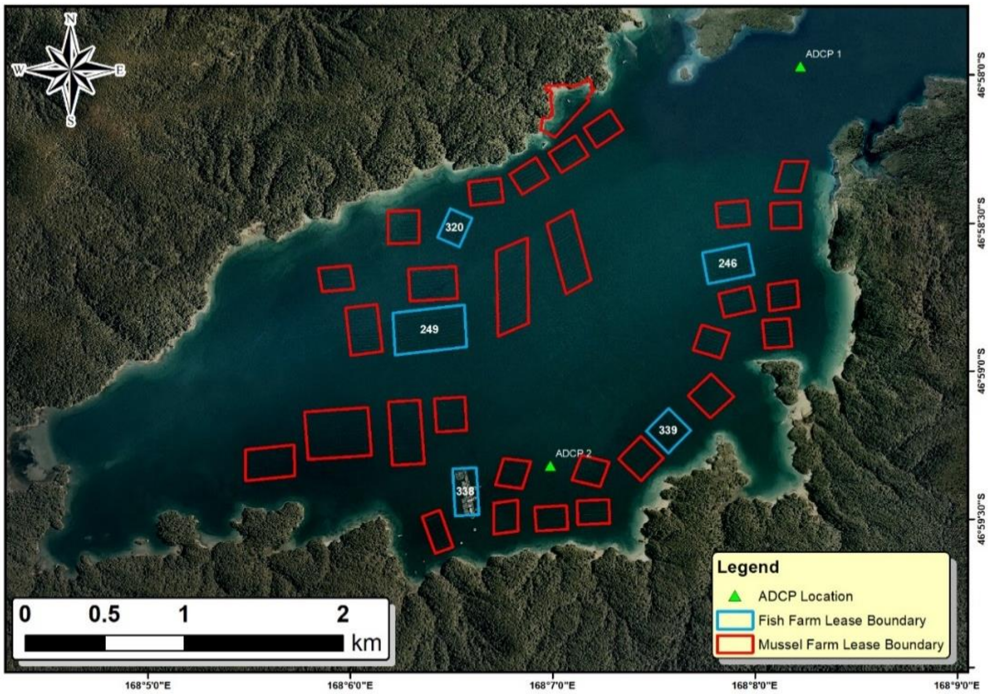


Figure 20. (A) Extent of regional and local grids, with regional bathymetry, (B) Location of the two ADCP deployments conducted by the Cawthron Institute from the 6th of September to the 6th of October 2010. (ADS 2017c).

Poor water quality can detrimentally impact the health and growth of farmed fish, particularly oxygen depletion and elevated ammonia-N concentrations. Oxygenated waters are critical for the survival and performance of farmed fish. If ammonia-N concentrations are very high close and in the fish farm, they can become toxic to fish. Excessive oxygen depletion and ammonia-N toxicity in the fish farm would stress and potentially cause fish mortality. However, these issues can be mitigated through appropriate farm location and management, and by protecting the farmed fish, so too fish species beyond the farm will be protected.

Another concern related to nutrient enrichment is the potential for increased occurrences of harmful algal blooms (HABs), including blooms of species that produce biotoxins (MPI 2013). However, in New Zealand, no known HABs have been linked to finfish farming (MPI 2013.

4.5.1 Nutrient enrichment and algal biomass

The existing state of the water column environment was described earlier in this report and changes observed as a result of salmon farming can be summarised as:

- DO levels were lowest in summer and only showed occasional reductions at sites near salmon farms and were always above 6 mg/L; and
- Chl-*a* varied between sites in some months with higher levels at sites near the entrance at times (eg. July 2014, Dec 2016), and the middle of the Bay at other times or were similar at all sites, and there was no consistent seasonal pattern (2014-17). There was no evidence of a spatial pattern related to marine farms, although a site near the head of the Bay was generally higher in 2012

Water quality modelling

ADS (2017d) reports on the modelling aimed at simulating the release of farm-derived nitrogen (in the form of total ammonia nitrogen or "TAN") and its subsequent effects on chl-*a* as a result of expanding salmon production in BGB. ADS (2017d) also examines the potential impacts of the proposed farm expansion on dissolved oxygen levels within BGB as a result of increased fish respiration (see next section). Readers are referred to ADS (2017d) for details of the modelling.

The latest modelling by ADS used new water column data collected monthly in BGB by Sanford since February 2015. The data includes dissolved nutrient and particulate information from four stations within BGB. This model was used to update the carrying capacity calculations made earlier (2010) with a more complex model of BGB. The 2010 model utilised data from an extensive literature review and *in-situ* datasets dating back to 1999-2000.

Three scenarios of TAN release were modelled by ADS based on feed data supplied by Skretting and Sanford:

- Baseline Scenario: Estimated existing releases of N as TAN from lease 246, 390 tons of N (in the feed);
- Scenario 1 (referred to as a “Conservative” scenario in ADS (2017d)): Corresponding to a feed input of 621.4 tons of N; and
- Scenario 2 or the maximum proposed production scenario (referred to as “Maximum” Scenario in ADS (2017d)) and a maximum feed input of 659 tons of N.

The proposal is to vary the consents to allow for an increase in feed to 659 t therefore the following sections focus on this scenario. Key results from the latest modelling reported in ADS (2017d) were:

- The modelled monthly averaged TAN concentrations were relatively stable throughout the year. They were slightly elevated from October to December and ranged from approximately 0.040 g/m^3 in August to 0.058 g/m^3 in November. The modelled annual average was approximately 0.050 g/m^3. Because of processes that were not modelled, such as phytoplankton uptake, the match is poor over the spring/summer period but levels were similar to those measured over autumn/winter;
- Measured total nitrogen concentrations as described in ADS (2017d) illustrate that total nitrogen concentrations follow the seasonality of measured ammonia concentrations but are up to approximately 8 times higher than ammonia concentrations. This reflects the contributions of nitrate and particulate nitrogen to the total water column nitrogen in BGB.
- For the Baseline Scenario modelled TAN concentrations showed little difference between the surface and bottom waters, except within cage locations where nutrients were released only in the surface waters. In BGB, the model predicts TAN concentrations would range from 30 to 60 µg/L and chl-*a* concentrations from 4 to 8 µg/L. The modelled chl-*a* concentrations are significantly higher than concentrations measured as part of the Sanford monitoring programme (ADS 2017d). ADS explain that this difference is a result of processes (especially mussel consumption of phytoplankton) that are not considered in the model.
- For the maximum proposed production scenario (this application) (referred to as “Maximum” in ADS (2017d)) TAN was released from Leases 246, 320 and 339 with a maximum feed input of 659 tons of nitrogen. TAN concentrations will increase by

between 20 to 30 µg/L in BGB. Chl-*a* in the spring and summer scenarios will increase by a maximum of 2.5 to 4.0 µg/L above baseline.

- Total N levels are predicted to increase by approximately 10% when compared to Total N observations made during Sanford's monthly water quality monitoring program.

ADS (ADS 2017d) have estimated that the total inputs of N from the full proposed level of production would be 1043 kg/d which compares with earlier estimates of rainfall, catchment runoff, and sediment release of 252 kg/d, 220 kg/d from excretion by salmon (**Table 1**, Rutherford et al. 1988) and an estimated exchange in and out of the Bay of 388 kg/d (Neil Hartstein, pers. comm.). However, despite the increase from salmon farms there is no evidence that phytoplankton biomass has increased or changed over the last 20 years, although as noted above there were some periods of low levels in 1997-2005. From **Figure 21** we can see that although there may be a slight decline over time there is no significant trends at either control site. For Site 3 at the mouth of the Bay there is no significant upward or downward trend ($P = 0.97$, Mann-Kendall test, $P<0.05$ is significant). For Site 4 there appears to be a very minor downward trend (reduction of Chl-*a* of -0.006 µg/L/year), however this isn't significant ($P = 0.66$, Mann-Kendall test). This suggest that the increase in salmon farming has not resulted in an overall increase in phytoplankton biomass in BGB.

4.5.2 Water quality standards for nutrients and phytoplankton biomass

Nutrient "thresholds" and guidelines have been applied in New Zealand and overseas to assess estuarine, harbour and coastal trophic status. There are no widely applied and used guidelines for bays like BGB in New Zealand, however ANZECC (2000) and the pilot National Objectives Framework (NOF) for Estuaries (reported in Green & Cornelisen (2016) do provide some guidance.

Although there are limitations with the present ANZECC guidelines they are still commonly applied to New Zealand coastal waters and estuaries as 'default trigger values" (i.e. where there is no better location specific information) for unmodified or slightly modified systems. Thresholds are to be compared with median concentrations for the system in question.

Nitrogen is considered the main limiting nutrient for plant/phytoplankton growth in New Zealand coastal systems, including the likes of BGB. A high concentration of phytoplankton is recognized as the major symptom or sign of eutrophication which can be followed by reductions in water clarity, depletion of DO and increased organic matter and potential deoxygenation in sediments along with increase in sulphide and other chemical changes. The high concentrations can lead to toxic effects on the food web if there are harmful algae blooms (HABs) or in the case of sulphate reduction the end product sulphide is toxic. In the case of

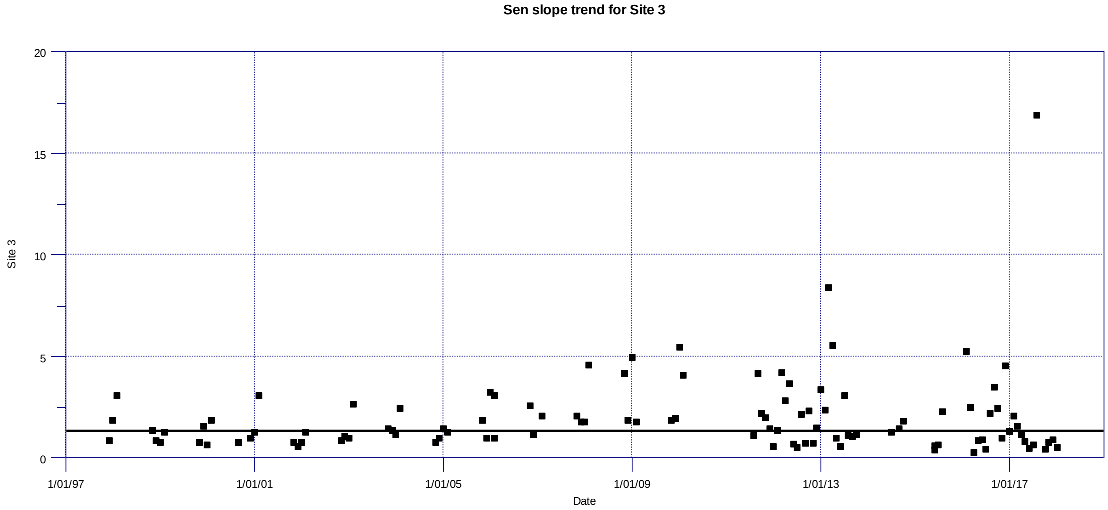
BGB, algal blooms are likely to be the main issue but in other coastal systems, especially shallow harbours macroalgal blooms can be an issue. Green & Cornelisen (2016) provide a review of marine water quality standards in New Zealand and internationally, including nutrients, chl-*a* as an indicator of phytoplankton biomass and DO. These are summarised below along with other standards and compared with levels predicted to occur in BGB.

Based on data from 2016-17 the annual median concentrations of DIN (nitrate plus ammonia-N [1]) at the two control sites in BGB was 0.055 g/m^3 with peaks in winter of up to 0.130 g/m^3 and lower levels in spring/summer (up to 0.050 g/m^3) (**Figure 22**). The median annual ammonia-N and nitrate-N concentrations for the year Mar 2016-Feb 2017 were 0.024 and 0.016 g/m^3 respectively, which is higher than the ANZECC guidelines for estuaries (0.015 mg/m^3 for both N species) and marine systems (0.005 g/m^3 for nitrate-N and 0.015 g/m^3 for ammonia-N). Median TN however was 0.048 g/m^3 in BGB for Mar 2016 to Feb 2017 which is considerably lower than the guidelines for marine waters (0.300 g/m^3 for estuaries and 0.120 g/m^3 for "marine' waters). Thus, while the dissolved nutrients were above the guidelines, this didn't translate into particulate material. Note that these guidelines are default trigger values for "slight-moderate" disturbed systems to avoid further significant adverse effects and are based on South-east Australian waters thus must be treated with caution.

The Auckland Council have set water quality objectives, which they consider to be acceptable for harbours and estuaries, at < 0.089 g/m^3 for ammonia-N and <0.105 g/m^3 for nitrate and nitrite-N. Levels for BGB would fall well within this range for both N species.

The pilot National Objectives Framework (NOF) for Estuaries discussed the use of "potential" nitrogen concentration as an attribute, which is, loosely, the water-column nitrogen concentration when primary production is zero (and therefore nutrients are not being drawn out of the water column). This might occur during periods of light limitation or during mid-winter. The NOF panel reasoned that potential nitrogen concentration could be compared with nitrogen concentration at which plant growth maximises under conditions of light saturation to develop attribute bands.

[1] No nitrite data is provided but concentrations will be negligible



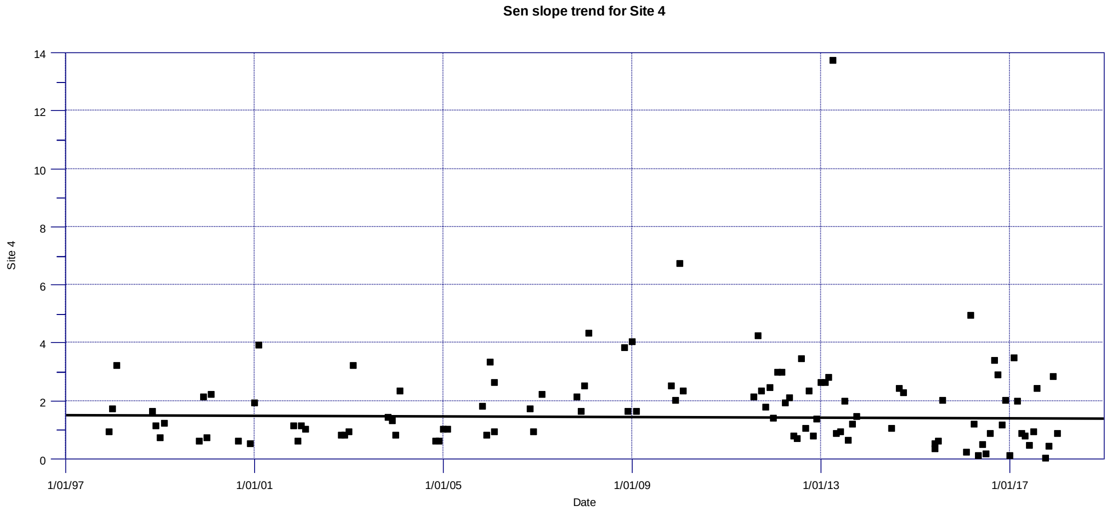


Figure 21. Chl-*a* concentrations for period 1997-2018. Trend analysed using Mann-Kendall test.

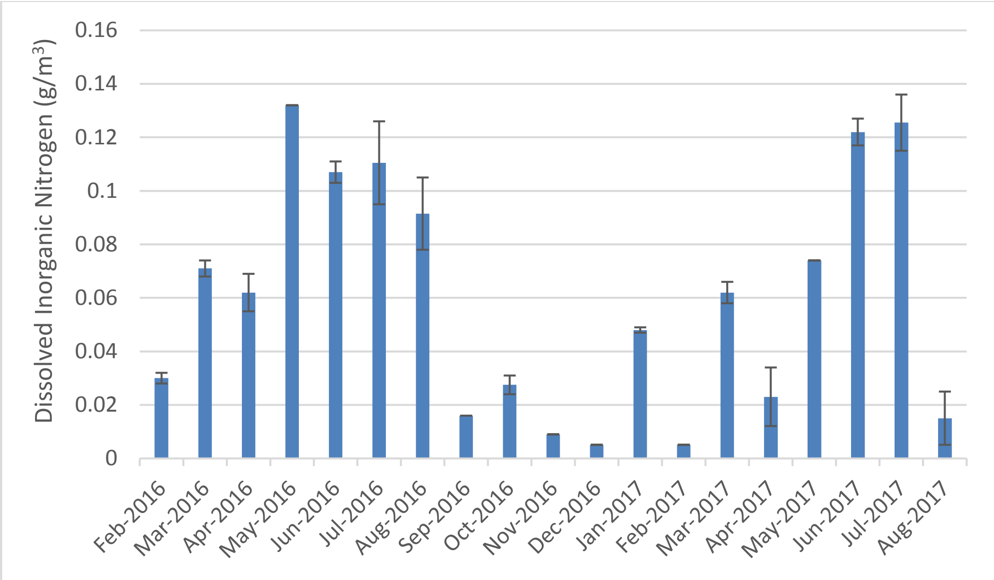


Figure 22. DIN (sum of nitrate and nitrite-N and ammonia-N) at control sites for 2016/17.

The panel considered that, "for nuisance algae such as *Ulva*, a reasonable value for this potential concentration is 0.070 g DIN m^{-3}, where DIN is dissolved inorganic nitrogen (sum of nitrate and ammonia-N). For coastal phytoplankton, values are quite variable among taxa but typically lie between about 0.014 and 0.070 g DIN m^{-3}. A 0.025 g m^{-3} threshold therefore constitutes a useful one with respect to identifying conditions where phytoplankton growth is likely to maximise". Following that reasoning, the NOF panel suggested attribute bands for potential nitrogen concentration as:

- Excellent: < 0.025 g DIN/ m^3
- Good: 0.025–0.070 g DIN/m^3
- Poor: >0.070 g DIN/m^3

The median DIN for Mar 2016-Feb 2017 in BGB at 0.055 mg/m^3 which would place it in the "good" band for potential nitrogen levels that may cause eutrophication.

Chang et al. (1990) attributed the naturally occurring bloom in 1989 that caused large scale mortality to salmon in BGB (see plankton section below) to a mixture of calm, warm weather (part of La Nina pattern) and relatively high levels of nutrients over summer (DIN recorded as 0.048 g/m^3), as well as the ability of the *Heterosigma* species to migrate down to nutrient-rich waters lower in the water column. Interestingly summer DIN concentrations in 2016/2017 reached similar levels to 1989 (0.047 g/m^3 in Jan 2017), but farmed fish were not affected at all. However, chl-*a* in BGB was 1.1 µg/L in Feb 1988 and 9.0 µg/L Jan 1989 which compares with summer maximums of up to 4.9 µg/L in 2009, 5.1 µg/L in Feb 2016 and 4.5 µg/L in Dec 2016. Very high concentrations of chl-*a* in August 2017 (up to 25 µg/L) would indicate that a large bloom occurred. However, no bloom was actually observed and there was no real pattern to the higher levels as very high concentrations were recorded at the control site at the mouth of the Bay and at farms near the north-west and south-east shores.

ANZECC (2000) provide default triggers for chl-*a* for natural to slightly disturbed systems of 4 µg/L and 1 µg/L for "estuaries" and "marine" respectively. Green & Cornelisen (2016) note that the median value for chl-*a* should be compared with the "eightieth percentile" for a nearby reference site or if this is not available then with the ANZECC guideline.

The pilot National Objectives Framework (NOF) for Estuaries (unpublished but referenced in Green & Cornelisen 2016) proposes the following bands for chl-*a* as indicative for phytoplankton:

- Excellent: < 2 µg/L
- Good: 2-5 µg/L
- Fair: 5-15 µg/L
- Poor: >15 µg/L

In the review by Green & Cornelisen (2016) they noted that ANZECC (2000) was developed for SE Australian waters and that Williamson et al. (2015) consider 10 µg/L is a more appropriate trigger for New Zealand waters as they are more productive naturally but with median levels typically below 5 µg/L. Green & Cornelisen also document a range of other standards that have been applied including the New Zealand Estuary Trophic Index Toolbox with a range from <3 µg/L for Band A with healthy ecological communities to >12 µg/L for Band D with a degraded state. The median chl-*a* in BGB in 2016 was 1.65 µg/L and 0.99 µg/L for control Sites 3 and 4 respectively and in 2017 was 0.74 and 0.85 µg/L respectively.

In the case of salmon farms in the Marlborough Sounds the Environmental Monitoring Plan for farms in Ngamahau, Kopaua and Waitata have set a chl-*a* standard for monthly sampling of a maximum of 3.5 µg/L in 3 successive months at any station in the Sounds. If this is exceeded and is shown to potentially be related to salmon farms then a second level response is required. Resource consent conditions for those farms also require (among other requirements) that farms do not cause:

- Increase in frequency, intensity, or duration of phytoplankton blooms (i.e chl-*a* concentrations >5 µg/L);

- Increased frequency of harmful algal blooms (HABs);

- Reduction in DO to levels potentially harmful to marine biota;

- Elevation of nutrients outside natural variation for a location and time of year beyond 250 m from the edge of pens; and

- Cause a statistically significant shift, beyond what is likely to occur naturally, from a oligotrophic/mesotrophic to eutrophic state.

Based on work carried out during the algal bloom of 1989 in BGB, Rutherford et al (1990) suggested 15 µg/L of chl-*a* as an initial upper limit which could result in nuisance levels of phytoplankton and is indicative of eutrophic conditions (based on their work and references contained in Rutherford et al (1988)). At the time, and based on existing levels of nutrients in BGB, they suggested the critical nitrogen concentration which could cause these chl-*a* levels would be 0.290 g/m^3. The nitrogen levels at the time were 0.035 g/m^3. DIN in summer of 2016/17 was up to 0.055 g/m^3 and residence time has been reassessed at more like 28-30 days rather than 10-13 days (Rutherford et al. 1990). It is considered that 10-15 µg/L of chl-*a* may be a reasonable indication that a bloom could be occurring at nuisance levels and could take the water body into a lower trophic state.

Predictions are that with the expansion in salmon farming in BGB the increases would be 2.5 to 4 µg chl-*a*/L, which would mean levels would be still well below the 15 µg/L indicator levels and in fact such a level would only have been exceeded once at each control site in BGB over the period 1997-2017. It should also be noted that no bloom was actually observed during the peak in August 2017. The model predictions are that only very small increases in TAN and chl-*a* would result in Paterson Inlet (0.005 g/m^3 and 0.5 µg chl-*a*/L respectively) and these would not affect water quality or health of the Inlet.

4.5.3 Dissolved oxygen

Salmon farming in BGB is largely self-regulating in that farmers will lose stock or fish will lose condition if they are stressed by low oxygen levels in the pens. Similarly if oxygen levels in the farm sustain the farmed fish, so too would oxygen levels in the wider environment be sufficient to sustain natural fish populations. Reduced levels of DO can affect the growth and survival of salmonids in different ways at different stages of life. Decreasing feeding activity in adult salmonids is usually observed under low DO concentrations (<6mg/L) and may also cause mortalities in “extreme” conditions (<3 mg/L) (Carter, 2005). Bjornn & Reiser (1991) also stated that DO concentrations <5 mg/L can adversely affect growth, food conversion efficiency, and swimming performance in salmonids.

USEPA (1986) calculated the median percent reduction in growth rate (as a result of lowering the DO concentration) of Chinook salmon (based on a number of previous studies) fed with full rations and exposed to an average temperature of 15°C. The calculations indicate that the growth rates were reduced by 7% at 6 mg/L, 16% at 5 mg/L, 29% at 4 mg/L and 47% at 3 mg/L. A similar study on Coho salmon and Sockeye salmon by Brett & Blackburn (1981) also indicates strong dependence of growth on the DO concentrations where DO levels were below 5 mg/L.

Ultimately, it is temperature that controls metabolic rates and the value of the limiting oxygen saturation (LOS) shifts exponentially with temperatures (Barnes *et al. 2011*). Atlantic Salmon have been shown to position themselves vertically in the water column primarily in response to temperature, even if it positions them in a layer of relatively DO poor water (Dempster *et al*. 2016, Johansson *et al.* 2006). Both Dempster *et al.* (2016) and Johansson *et al.* (2006) have shown salmon to actively avoid temperatures greater than 18° - 19°C by swimming to deeper areas of the cage where there are cooler temperatures. Numerous personal observations by Aquadynamic Solutions corroborate these same phenomena at farm sites in New Zealand and Australia for both King/Chinook and Atlantic salmon.

Dissolved oxygen concentrations are correlated to stocking densities and ADS (2017d) concludes that fish can be sustained and not be impacted if DO levels are kept above 6 mg/L.

Green and Cornelison (2016) have reviewed standards for DO in marine waters for a variety of biota, including ANZECC (2000) guidelines and various trigger values applied internationally and elsewhere in New Zealand. Triggers are based on percentage saturation or absolute DO concentrations and vary between 3-4 mg/L for instantaneous and up to 6 mg/L for 7–day measurements. Bands recently developed as part of the pilot National Objectives Framework (NOF) for estuaries suggested the following DO levels as providing for “ecological health”:

Excellent: > 10 mg/L

Good: 8-10 mg/L

Fair: 5-8 mg/L

Poor: > 5 mg/L

A level of 4.6 mg/L was recommended as a level that would avoid catastrophic mortality of marine biota and the NZ Estuary Trophic Index Toolbox (Robertson et al. 2016) classifies waters with 1 day minimum of > 5.5 mg/L as in the band A where there is no stress on marine species. Predictions are that DO levels will remain > 7 mg/L overall and > 6 mg/L within farms. The proposal for BGB that ensures DO of at least 6 mg/L is therefore considered to be protecting both farmed and wild fisheries.

4.5.4 Toxicity

Elevated nitrogen levels can potentially cause near-field toxicity effects in the immediate vicinity of fish cages, before any significant dilution of nitrogen has taken place.

Toxicity is based on a combination of two parameters: the duration of exposure and level of exposure. For example, "acute" toxicity levels are where pronounced effects occur over relatively short time frames (i.e hours to days), while "chronic" toxicity levels are those that relate to longer time frames (i.e. weeks to months). Chronic thresholds are generally set at much lower levels than acute toxicity values. Chronic toxicity can lead to a general decline in the health of farmed and natural biota, higher feed conversion ratios and ultimately increased levels of mortality in farmed and naturally occurring communities and populations. Elevated concentrations that only occur for very short durations may not necessarily cause harm to a particular organism. The actual processes involved in the toxicity relate to species assimilation and metabolism rates.

Ammonia exists in two forms in solution: the abundant ammonium ion or ionized ammonia (NH_4^+) and the toxic un-ionized or free ammonia (NH_3). The sum of both added together is the parameter referred to as TAN. The ammonium ion is almost innocuous whereas the un-ionized ammonia is extremely toxic and poses a threat to health of farmed and naturally occurring biota even at relatively low levels.

Trout are considered to be particularly sensitive to ammonia and the most sensitive of all salmonid species farmed (and hence a good conservative indicator). As shown in Fuller *et al.* (2003), Gila trout (*Oncorhynchus gilae*) survived 0.36 mg/L of unionized ammonia nitrogen (NH_3), but half of the population died after a 96 hours exposure to 0.47 g-NH_3/m^3.

The criteria for protection of aquatic life established by the US EPA agency are based on tables relating the concentrations of total ammonium nitrogen to pH and temperature. These can be used as guidelines for the maximum acute and chronic allowable concentrations. For seawater with salinity of 30 PSU, temperature of 15°C and pH of 8.4 (typical worst case observed in BGB), the maximum acute concentration allowed by EPA criteria is 4.2 g-TAN-N/

m^3 while the chronic concentration is 0.62 g-TAN-N/ m^3. In BGB the predictions that TAN would only reach 0.09 g/ m^3 which is well below the EPA chronic criterion.

Hartstein & Oldman (2015) have summarised the dissolved oxygen and nitrogen related adverse effects in estuarine and marine waters. The values of acute and chronic toxicity of each trigger parameters are shown in **Table 5** and again the levels in BGB are well below thse concentrations.

Table 5: Trigger parameters and their values for acute and chronic toxicity in estuarine and marine waters as cited in Hartstein & Oldman (2015).

	Acute Concentration	Chronic Concentration
Dissolved Oxygen	< 2.3 mg/L DO (EPA, 2000)	<4.8 mg/L DO (Vaquer-Sunyer & Duarte, 2008; EPA, 2000)
Nitrate	>339 mg NO_3^--N/L (CCME, 2012)	> 3.7 mg NO_3^--N/L (Meays, 2009) > 45 mg NO_3^--N/L (CCME, 2012)
Nitrite	Acute levels specific to marine systems are not yet defined.	> 0.06 mg NO_2^--N/L (EPA) > 5 mg NO_2^--N/L (EPA warm water fish)
Unionized Ammonia	Worst case: > 1.4 mg NH_3-N/L (EPA) (T=25°C, pH=8.6 and salinity=30 PSU)	Worst case: > 0.22 mg NH_3-N/L (EPA) (T=25°C, pH=8.6 and salinity=30 PSU)

4.6 Sedimentation and effects on the seabed

The effects of finfish farming on the sediment and seabed are primarily caused by the deposition of organic waste (waste feed, fish faeces, biofouling material) which in turn can lead to increasing anoxia, release of hydrogen sulphide and nutrients, and changes to biological communities.

4.6.1 Chemistry and nutrient release

In the sediment, organic matter is decomposed (broken down) by complex microbial processes. The decomposition of organic matter releases nutrients, particularly ammonia (NH_3), ammonium (NH_4, if dissolved oxygen is present) and phosphate (PO_4). These nutrients are released into the water column, exacerbating nutrient enrichment.

Sediment microbes use different sources of energy, which results in sediment oxygen consumption and potential release of hydrogen sulphide and methane into the water column. The preferred source of energy for microbes is oxygen. This generates a flux of oxygen into the sediment, contributing to oxygen depletion in the water column. Sediments can become anoxic (oxygen-depleted) under fish farms and, if organic matter enrichment is severe, even azoic (devoid of any living organism). When dissolved oxygen is depleted in the sediments, microbes use other sources of energy to decompose organic matter, following a well-defined sequence of diminishing energy gain for microbes. This can result in the release of hydrogen sulphide (H_2S) and methane (CH_4).

Microbial organic matter decomposition processes are so-called redox (oxidation-reduction) reactions. The redox conditions in the sediments depend on the prevalent microbial processes taking place and thus represent the degree of organic enrichment. Redox conditions can be assessed by measuring the sediment redox potential, Eh (expressed in mV). In the surface sediment (if oxygen is available), the redox potential is positive. It decreases with depth related to the decrease in the dissolved oxygen concentration. Negative Eh values are associated with anoxic conditions.

The redox potential discontinuity layer (RPD) is the point at which the redox potential drops abruptly from positive or slightly negative to highly negative values. This depth has been used as an indicator of the depth of the sediment where oxygen is depleted. However, while the depth of the RPD can be a good indicator of relative oxygen content in the sediment, it needs to be applied with caution because the relationship between oxygen and RPD depth is influenced by many factors that are situation-specific and vary greatly (for example, see Gerwing *et al*. 2015).

As described earlier, the history of the farm sites is that Farm 338 in the south of BGB has been a salmon farm for most of its time; Farm 339 in the south-east was a salmon farm, then was converted to a mussel farm and then back to a salmon farm; while the farm in the middle of the Bay (Farm 249) is a mussel farm that has been converted to a salmon farm and is now vacant. Farm 246 is the on-going farm site now, Farm 338 a brood farm and 339 a smolt farm (recently vacated).

The effects that can be attributed to salmon farming in BGB based on monitoring surveys to date can be summarised as:

- Farm 339 – The sediment properties appear to have remained similar from 2013 when the site was converted again to a smolt farm, through to 2015. Measurements since 2015 show the farm was organically enriched but sediments 50 and 100 m from the pens appear to be only moderately changed and still retain functioning and diverse communities with indications of a healthy, well oxygenated sediment with minimal impact from the farm at 50 and 100 m, similar to those at control sites (Stenton–Dozey 2015, ADS 2017a). Burrows and worm holes were evident at all sites at this farm in 2015. Metal concentrations did not exceed the ISQG-low threshold. By 2017 (ADS 2017a) metal levels were above ISQG-high at the pens themselves. *Beggiatoa* mats were present under the farm in 2015 but had gone by 2017 although there was still a strong sulphide smell in samples from the farm itself. The number of species at the edge of the pens was still very low in 2017 (lowest of all salmon farm sites). Metals were low at all sites 50 and 100 m from the boundary of this farm in 2017. Note this farm has recently been vacated and moved to Lease 340 (March 2018) to allow the benthic environment to recover;

- Farm 249 – In 2015 sediments were similar under pens, 50 and 100 m away with *Beggiatoa* mats present at all sites. Poor benthic conditions in 2015, 5 years after salmon farming began, had reduced species diversity but still retained high numbers of opportunistic species. There were no burrow or worm holes at the site surveyed in 2015. Species richness indicated a deterioration in the benthic environment between 2014 and 2015 (Stenton–Dozey 2015). Copper and zinc levels were high under the farm but dropped away from the edge of the farm. Zinc levels were below the guidelines. This site has been fallowed since 2016 and recovery will be followed. *Beggiatoa* mats were present in 2016 when the site was vacated but none were observed in 2017. Seabed conditions at all sites surveyed in 2017 appeared to have improved compared with 2016;
- Farm 338 – In 2015 species richness was low but there was some evidence of recovery at this site (reduced organics, increase in DO, very high numbers of some bio-turbators – nematodes and polychaetes). In 2014 and 2015 copper and zinc were higher at the 100 m point and were all above the ISQG-high level at all sites. By 2017 (ADS 2017a) levels were only above ISQG-high at the pens themselves even though the brood stock was moved back there in 2015, there was no sulphide smell in 2017. This is to be expected given the low stocking densities; and
- Farm 246 – This site was organically enriched at the farm itself in 2017 but with no sulphide smell. The yearly survey completed in early March 2018 (Neil Hartstein, pers. comm.) indicates that the new farm at 246, stocked in September 2016, is well managed and an abundant benthic invertebrate community was observed at the edge of the cages (see **Figure 23** for photos).

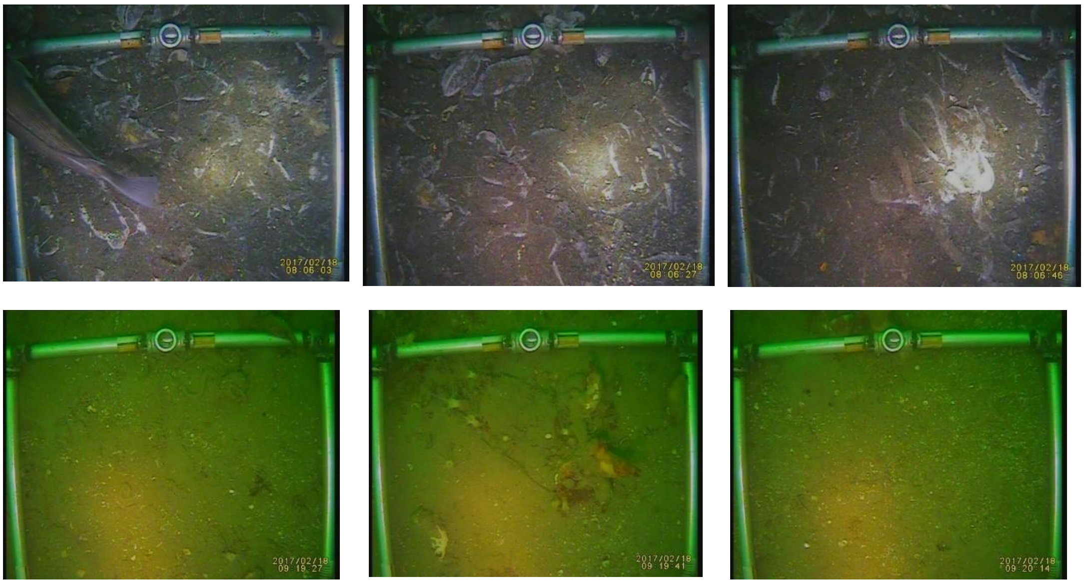


Figure 23. Photos beneath Farm 246 (top) and at 100 m away (bottom). Farm 246 has been operated as a salmon farming since September 2016.

4.6.2 Ecology

Sediment enrichment under fish farms can have detrimental effects on ecology. In anoxic sediments species diversity is reduced as only few sediment-dwelling organism species can tolerate the degraded conditions. In cases of extreme organic enrichment, conditions can become completely devoid of living organism (azoic). If deposition is very high (typically only directly under a farm and in low flow conditions), it can also lead to smothering of sediment dwelling organisms.

Organic enrichment and deposition of biofouling organisms (see biosecurity section) can attract scavenging organisms such as sea cucumbers, sea stars and sea-lice (isopods), which often aggregate around the perimeter of the farm (MPI 2013).

Two key indicator species for enriched sediments are the presence of Capitellida and *Dorvelleid* polychaete worm species. In the 2017 surveys these were relatively common under mussel farms and at the edge of under salmon pens at Farms 338 and 246 but were only in low abundance, when present away from the pens.

4.6.3 Deposition of additives

Currently, the New Zealand finfish aquaculture industry uses minimal chemicals such as antibiotics, parasiticides and other therapeutants and the risk of ecological effects from therapeutants is therefore considered very low (MPI 2013) and none of these are used by Sanford. In the case of Big Glory Bay with the change to mechanical cleaning of the nets instead of using antifoulants the concentrations of contaminants such as metals (in particularly copper) will have started to reduce (also see Section 4.8). In addition Sanford in recent years has moved to using organic zinc in feeds.

4.6.4 Farm footprint

ADS (2017e) used depositional modelling to define the footprint of the proposed changes to increased feed inputs in terms of sedimentation of faecal material and waste feed within BGB.

Deposition from fish cages was modelled using the software New DEPOMOD, a widely used particle tracking model designed for predicting salmon farm deposition. The model simulates depositional farm footprints based on bathymetry, local currents and farming practices, such as pen layouts, feed input, and stocking density.

A range of feed input scenarios were modelled for Leases 246, 320 and 339:

- Lease 246, Scenario 1: typical feed properties (feed without the use of binding agents) at a daily feeding rate of 18.6 ton, stocking density 15 kg/m^3.
- Lease 246, Scenario 2: included binding agent in the feed. The total yearly Nitrogen input is 415.1 tons which is representing the maximum potential production being applied for.
- Lease 320, Scenario 1: conservative stocking density scenario (12 kg/m3) with a daily feeding rate of 7.76 tons
- Lease 320, Scenario 2: high stocking density scenario (14 kg/m3) with a daily feeding rate of 9.05 tons.
- Lease 339, Scenario 1: conservative scenario for growing out smolts with a stocking density of 3.0 kg/m3
- Lease 339, Scenario 2: maximum scenario for growing out smolts with a stocking density of 4.3 kg/m3

A binding agent was introduced in scenario 2 of Lease 246 to manage the resuspension and wide dispersion of deposited material generated by the strong currents observed at this site. ADS (2017e) explained that because neither lease 320 nor lease 339 displayed any significant resuspension, there was no need to include the use of feed binders in those simulations.

Current data to drive the model were obtained from ADCP measurements:

- Data collected with ADCP 1, at the mouth of the bay, was used to simulate Lease 249, which is located slightly west of the centre of the bay.
- Data collected with ADCP 2, in the southern part of the bay, was used to simulate Leases 320 and 339.

Pen set up, inputs for the depositional model, farming inputs and waste scenarios are described in ADS (2017e). Based on international literature values, ADS (2017e) identified a deposition of 0.73 kg/m2/y (2 g C/m2/ day) as the threshold above which ecological impacts on the seabed are assumed to occur.

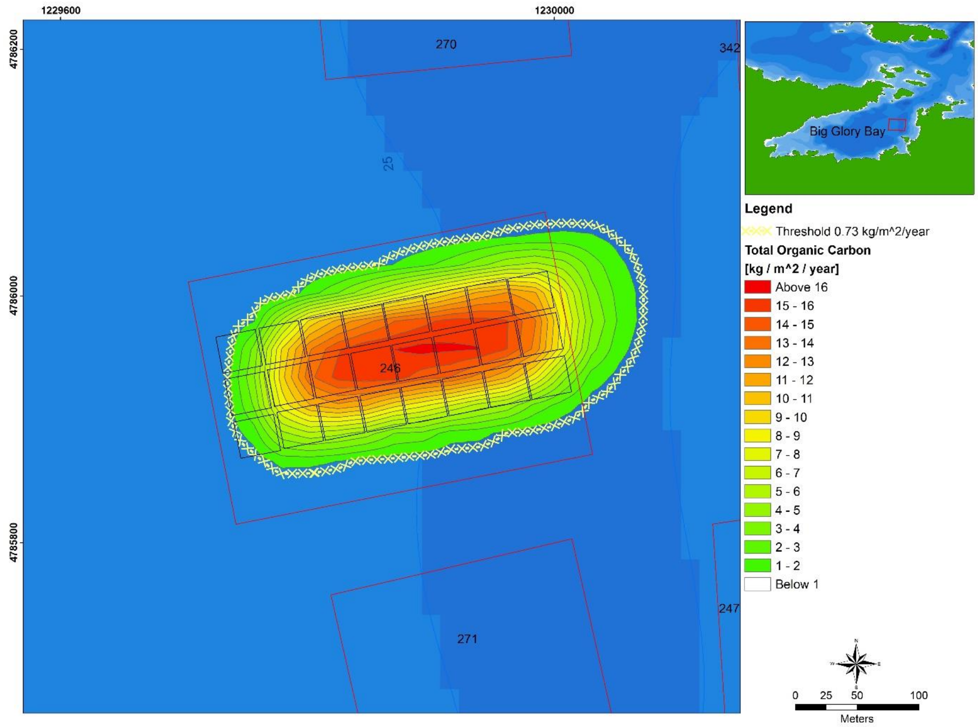
The simulated depositional footprints are mapped for all scenarios in ADS (2017e). They indicate that deposition of material from the farms generally remains within the boundaries of the lease (providing binding agents are used for feed in Lease 246). The maximum extent of the depositional footprint outside the Lease boundary was 100m. The maximum deposition under a Lease was over 16 kg C/m2/y.

Predictions are that deposition greater than 2 kg TOC/m^2/yr or 5 kg/m^2/yr total faeces and waste solid feeds will cover most of the available area within the boundaries of leases 246 (see **Figure 24** for Farm 246 as an example) and 320 but extend no more than 50 m or 25 m beyond the lease boundaries for Farm 246 (with binding agent) and 320 respectively.

In regard to Lease 339, ADS (2017e) concludes that “Fortunately, the relatively small size of the pens on lease 339 compared to the area of the lease will allow for the relocation of those pens above unused areas of the lease as a fallowing strategy, as this scenario yielded smaller depositional footprint (approx. 4.71 ha) than Lease 320 or 249. Several other fish farm leases are also currently being fallowed and could be utilised to allow for existing leases to be fallowed in the future”.

An important aspect of this application is that it is only to extend existing farms within license areas. The license areas have been carefully chosen to avoid sensitive habitats of inshore reefs. As discussed earlier the areas in BGB that have been identified as having more diversity and more of a “pristine” benthic environment are near the entrance around Bravo Island where there are red algae and brachiopods of conservation interest. The deposition of material is very localised around the salmon farms and would not impact on these more sensitive communities.

(A)



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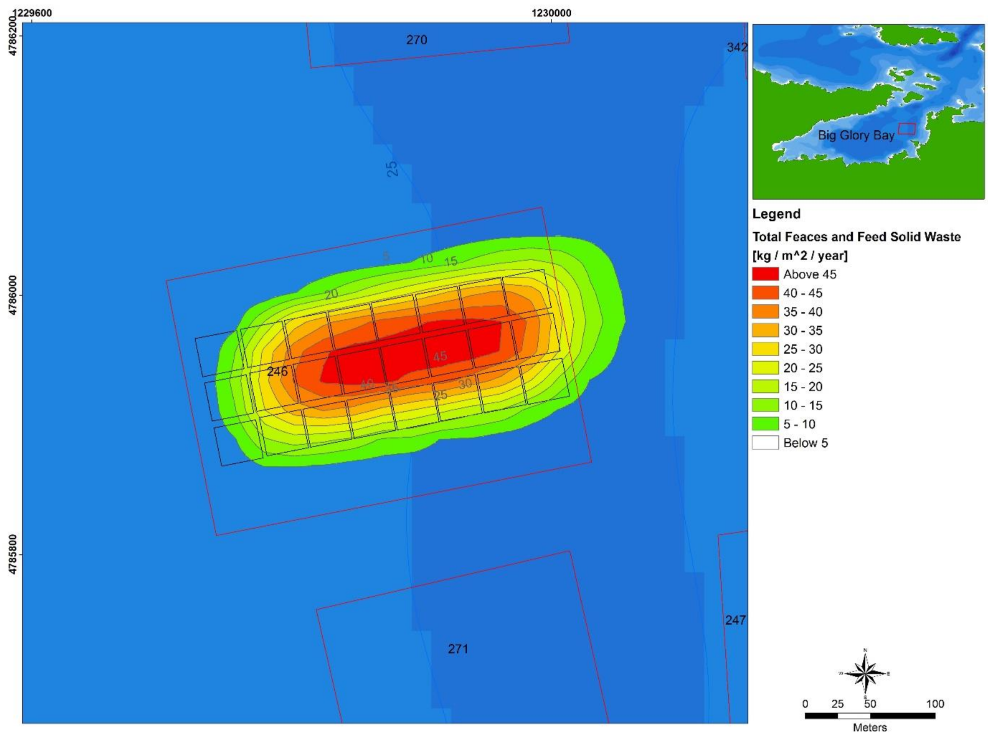


Figure 24. Lease 246 (A) total carbon deposition and (B) total faeces and solids deposition with a 12.0 kg m^{-3} stocking density. A binding agent is assumed to be in the feed. This scenario assumes an annual feed mass of 6,798 tons. (ADS 2017e)

ADS also carried out some additional deposition modeling of Farm 246 which is where the main increase in feed is proposed to occur initially. The results of that modelling are shown in **Figure 25** as an example of how the farms would overlap when moved within the lease as part of a fallowing plan**.** The model was run for a peak stocking density of 15 kg/m^3 and 14 cages, using binder in the feed. Results show that the deposition would still extend less than 50 m from the lease boundary. There would be some overlap as the new farm came into operation but most of the previous farmed area would recover prior to it being restocked after fallow.

4.6.5 Recovery

Comprehensive recovery studies have been carried out in Canada and recovery rates have been observed from a few weeks to over 6 years (Brooks et al. 2003, 2004). There are two important aspects to recovery defined in the Canadian studies 1) chemical – defined as the reduction of accumulated organic matter with a corresponding decrease in free sulfide in the sediments and an increase in sediment redox potential under and adjacent to salmon farms to levels at which more than half the reference area taxa can recruit and survive (free sulfides < 960 µM), and 2) biological – defined as the restructuring of the infaunal community to include those taxa whose individual abundance equals or exceeds 1% of the total invertebrate abundance at a local reference station.

MacLeod et al. (2004) studied recovery processes at salmon farms in Tasmania over several years and concluded that macrobenthic recovery was slower than chemical recovery and thus both need to be taken into account.

Keeley et al. (2014) documented 8 years of benthic recovery at a highly impacted salmon farm and concluded that although substantial recovery had occurred after 2 years, recovery was assessed as complete after ~ 5 years.

Keeley at al. (2015) tracked recovery at a salmon farm in a sheltered embayment in the Marlborough Sounds. Significant recovery was observed after 6 months, however the macrofaunal community still had not fully recovered after the two year study was completed.

Morrisey et al. (2000) used data from BGB to test a predictive model for impacts and recovery of benthic environments from salmon farming. Based on 2-hour average velocities at the sites (low velocity sites) recovery ranged from 3.4-4.5 years which corresponded well with observed rates of decreasing waste layers. They also noted that the presence of heavy metals can impact on rate of recovery. As noted earlier for the purposes of this report recovery is to a stage that will allow the area to be farmed again.

Monitoring of the recovery at Farm 249 is now underway but the literature review provides enough to determine appropriate mitigation measures (see Section 4.11).

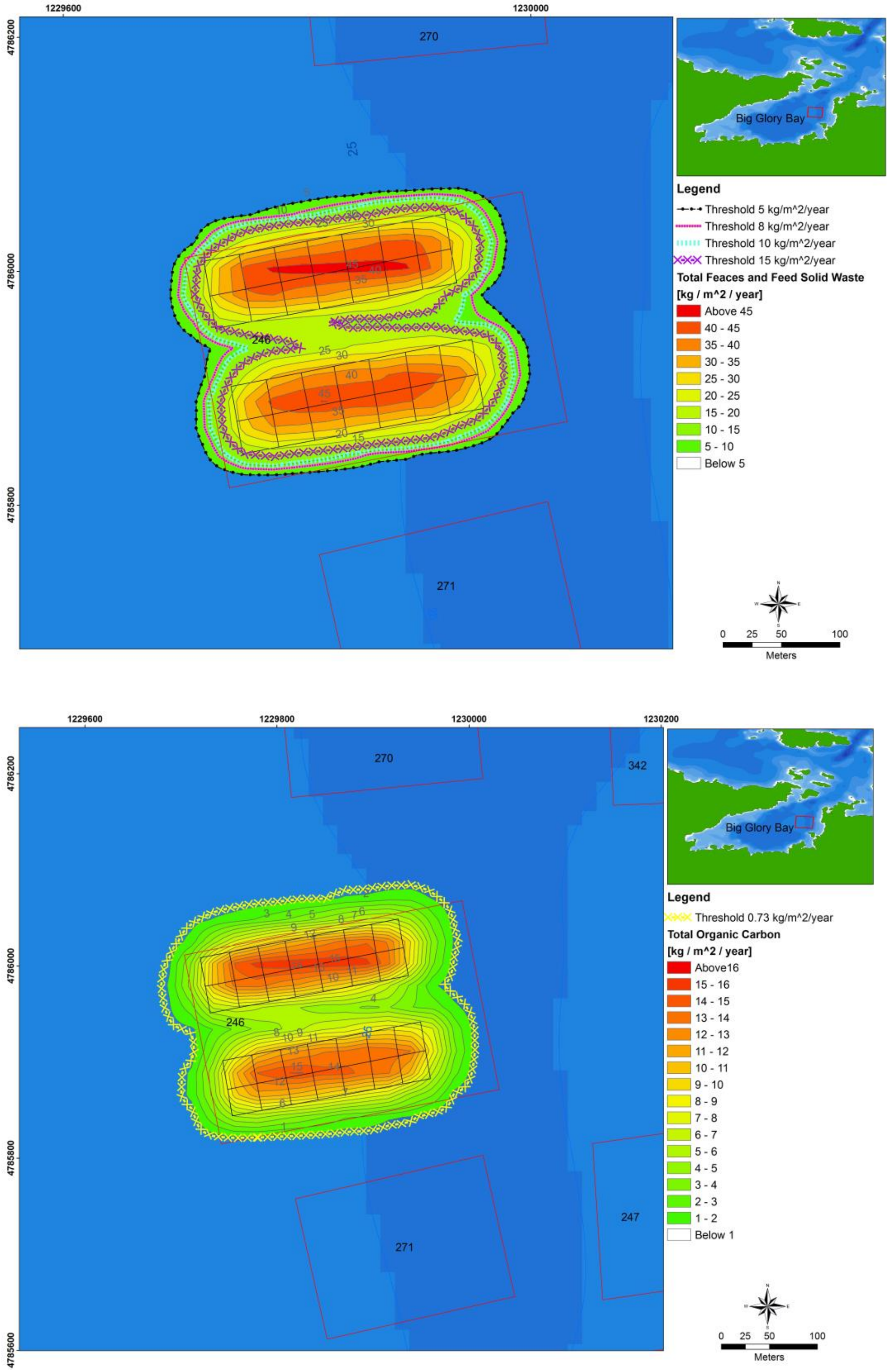


Figure 25. Predicted deposition rate and overlap for moving pens within lease for Farm 246.

4.6.6 Standards and guidelines

The monitoring of benthic impacts is mandatory in most countries and for salmon farms in New Zealand, with standards set for a number of indicators, which if breached will lead to further investigations or eventually management actions if necessary. These standards or indicators are aimed at retaining a functional benthos around salmon cages.

A number of individual parameters and indicators have been applied to describing the effects of marine farms on the benthic environment. One that has received considerable attention in New Zealand in recent years is the Enrichment Scale (ES) which provides a single measure or index for managing resource consents and integrates a number of measures including sediment organic content, sulphides, redox, species richness and infaunal abundance. The scale goes from what is considered a pristine natural condition (ES=1) to one that is extremely enriched and impacted (ES=7) (Pearson and Rosenberg 1978, Keeley et al 2012) (**Figure 26**). While this approach has merit, it relies on one integrated unit and it is very important that key drivers of ES are not lost sight of when integrated across multiple measures. The application of ES also needs to take into account the type of environment and whether the ES is applicable to that environment.

The extent of ES predictions for the proposed expansion of feed inputs in BGB are shown in **Figure 27**. It can be seen here that only a very small area has an ES of >5 and it extends less than 50 m from the farm boundary. In the recent King Salmon developments in the Marlborough Sounds the criteria applied was that immediately beside and beneath the pens (Zones 1 and 2) ES should be < 5.0. The predicted area with an ES >5 for farm 246 in BGB would meet this criterion (based on solids deposition converted to an ES score as reported in Keeley et al. (2012)).

Associated Environmental Quality Standards (EQS) used in the Marlborough Sounds are that in Zones 1 and 2 there is:

- No more than one replicate core with no taxa;
- No obvious spontaneous outgassing (H_2S/methane); and
- *Beggiatoa mats* coverage no greater than localised/patchy distribution.

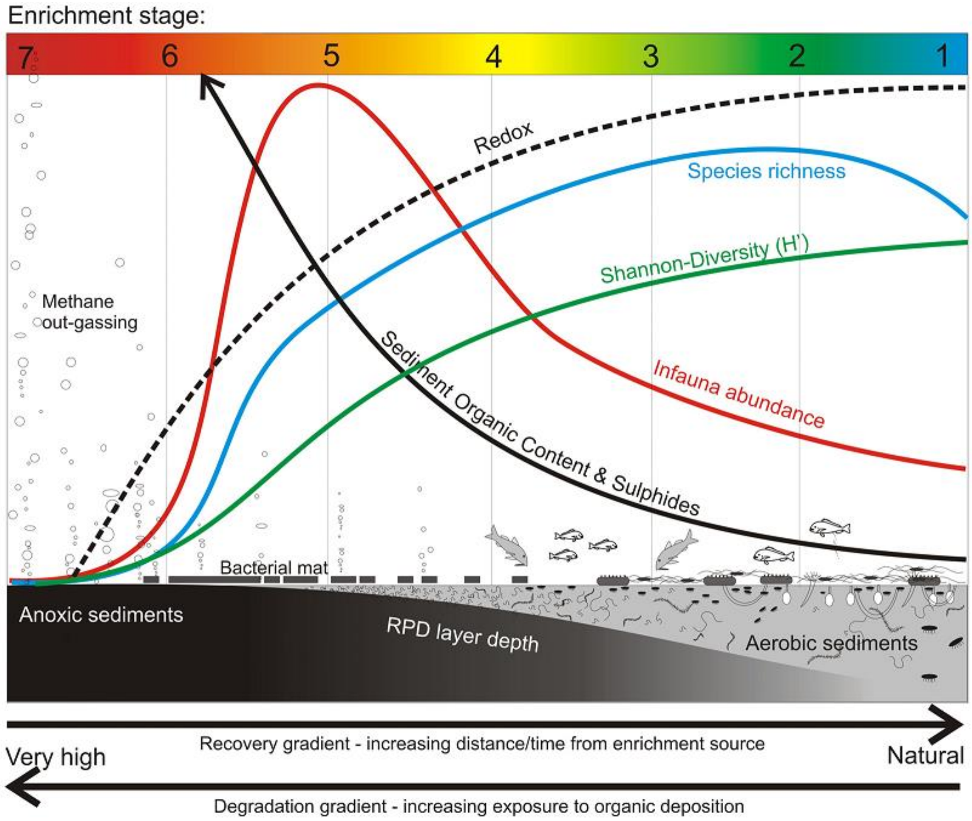


Figure 26. Seabed effects with increasing organic enrichment. (MPI 2013a).

The Benthic Working Group Report by Black et al. (2008) provide sediment quality criteria derived for the Northern Hemisphere at which regulators need to take remedial action. The criteria include that within the farm there are no more than two replicates with no taxa present, TOC is < 9% and outside the farm no *Beggiatoa* mats. These criteria would all be met with the proposed expansion of Farm 246 in BGB.

For this consent ADS (2017e) and the AEE recommend triggers for TOC deposition of no greater than 0.73 kg/m^2/y and total faeces and solid waste no greater than 5 kg/m^{2}yr. These are consistent with Hargreave (1994) and Gillebrand et al. (2002) who suggested that loadings of < 2 gC/m^2/d (=0.73 kg/m^2/y) on the seabed will have little impact on the environment. The same trigger has been used recently in an application for a marine farm development in Storm Bay Tasmania along with:

- A corrected redox value which differs significantly from the reference site or is <0 mV at 3 cm;
- A corrected sulphide level which differs significantly from the reference site or is > 250 µM at 3 cm depth;

- A 20 x increase in total abundance of any individual taxonomic family relative to reference site or > 50 times the total Annelida abundance at reference sites; and
- A reduction in the number of families by 50% or more relative to reference site.

These various studies, triggers and standards should be taken into account when setting limits for the proposed expansion in BGB and included in the Big Glory Bay Salmon Farm Environmental Management Plan ("BGBSFEMP"). The recommended limits for the benthic environment are included in the proposed conditions with the focus on the total organic carbon and total faeces and solid waste deposition which are the key parameters of concern.

4.7 Biosecurity

4.7.1 Marine pests

Organisms that have adverse effects on the environment, including some biofouling organisms, are referred to as marine pests or harmful aquatic organisms[2]. Adverse effects on the environment include over-growing of high value biogenic habitats, invasion of seagrass habitats (not relevant to BGB), and localised fouling of structures, including marine farms (Forrest *et al*. 2011). These effects are potentially irreversible depending on the species.

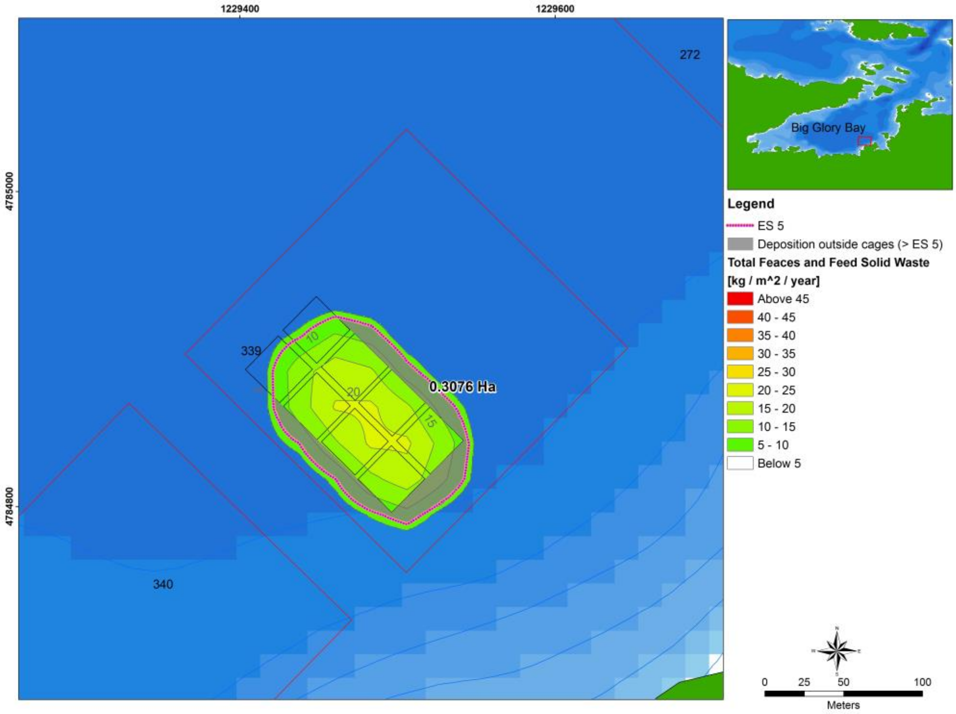
Some marine pests spread naturally, either via release of microscopic life stages (e.g. seaweed spores or animal larvae) to the water column (Forrest *et al*. 2007), or via release of viable fragments (Forrest *et al*. 2000; Bullard *et al*. 2007). Marine pests may also spread through anthropogenic pathways among farms, to other structures or to land via transfer of aquaculture gear or vessel movement.

While finfish aquaculture may potentially exacerbate the spread of marine pests, it is unlikely to significantly increase the level of biosecurity risk because of the many other sources of biosecurity risk in the coastal marine area (Forrest *et al*. 2015). Finfish farms thus predominantly act as "stepping stones", enhancing spread of marine pests, even if they were not the original source of the marine pest in the region.

In the case of this application in BGB the proposal is for an extension of existing farms and within existing leases. Thus we would not expect an increased risk of marine pests. Sanford also have strict biosecurity plans which represent best practice and are aimed at eliminating or minimizing such risk.

[2] The NZCPS 2010 defines harmful aquatic organisms as "Aquatic organisms which, if introduced into coastal water, may adversely affect the environment or biological diversity, pose a threat to human health, or interfere with legitimate use or protection of natural and physical resources in the coastal environment."

(A)



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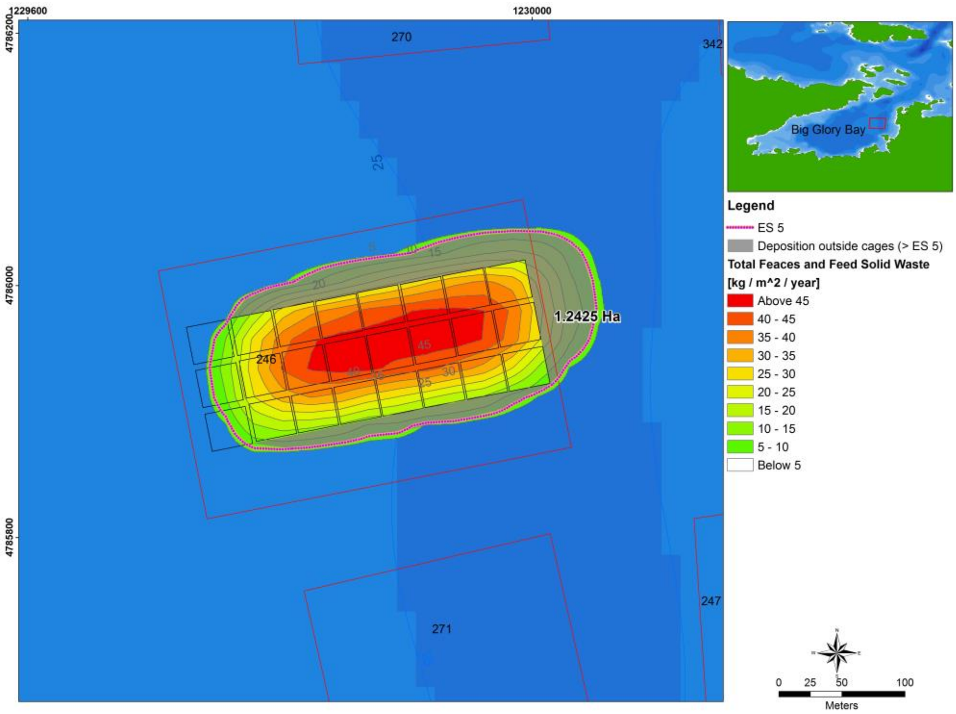

Figure 27. Predicted deposition of total faeces and feed solid waste for (A) Farm 339 and (B) Farm 246 (Model output from ADS)

4.7.2 Diseases

There are a range of known diseases and parasites associated with finfish aquaculture overseas but no diseases have been reported farmed fish in the South Island of New Zealand and no anti-biotics are used (Sanford Ltd, Alison Undorf-Lay pers. comm.).

4.8 Effects on wild fisheries

Fish farm structures and waste feed may attract wild fish. This might make wild fish more vulnerable to fishing pressure as marine farms are often popular recreational fishing locations. Wild fish aggregations may attract other predators such as seals, dolphins or sharks. These effects are considered to be of minimal importance in New Zealand (Forrest *et al*. 2015).

There is a risk that farmed fish may escape (MPI 2013). Escapees may compete for resources with wild fish, predate on wild species and alter the genetic structure of wild fish populations (change in fitness, adaptability, diversity or reduced survival). Escapees may also transfer pathogens to wild fish (see section Biosecurity – disease).

The potential effect and risk of escapees depends on a range of factors, including the farmed species (especially whether fish are native or introduced), the number of escapees, the proximity of the farm to wild fish populations and the ability of escapees to survive and reproduce. At present, the likelihood of adverse escapee effects is considered low in New Zealand because of the small size of the industry, and the limited overlap of farmed and wild salmon populations. This would particularly apply to BGB and Paterson Inlet where there are no significant wild salmon fisheries, also noting that salmon spawn then die and farmed fish are all females/neo-males which mean they can’t reproduce (Sanford Ltd, Alison Undorf-Lay pers. comm.).

4.9 Effects on mammals

Interactions between marine wildlife and aquaculture result from an overlap between the spatial location of the facilities and important habitats and / or main migration routes of the species. Interactions can lead to either direct effects, such as habitat exclusion, incidental entanglements or localised attraction / disturbance from artificial lighting and underwater noise generation, to indirect effects such as possible flow-on effects through the food web or changes to the habitat (MPI 2013).

In the case of BGB, there appeared to be little overlap initially between the farms in Big Glory Bay with any critical marine mammal habitats at the time of the first farming in the 1980s. However, interactions and predation incidents between the farms and both pinnipeds and bottlenose dolphins have not been considered uncommon over the years[3]. With the recent shift in one farm's location closer to the mouth of the Bay, and the establishment and continued growth of a sea lion colony in the 'Neck' area nearby, such incidents have increased and will likely continue to do so, particular if there are any further changes in farm size or activity.

Such interactions do not constitute habitat exclusion as this Bay is not considered ecologically significant habitat for any species. Nonetheless, the attraction of individuals could, if not well managed, be a potential issue from a potential entanglement perspective (fatal or non-fatal) as well as an impact on farm itself (e.g. fish losses). Dolphins are often more attracted to an increase in wild fish around finfish farms (due to excess feed or sheltering) while pinnipeds tend to be strongly attracted to the farmed fish as a food source. As a result, these species can become entangled in farms that are not properly installed and maintained or in predator nets (Tanner 2007).

Sanford does not use predator nets around the outside of the pens to reduce or dissuade predator attacks in Big Glory Bay, unlike other salmon farm companies in other parts of New Zealand. The lack of predator nets, along with proper cage maintenance, should reduce the risk of possible marine mammal entanglement significantly (see Tanner 2007). This factor likely accounts for the zero marine mammal entanglement record of salmon farms in this Bay to date (A. Undorf-Lay, Sanford pers.com.).

Despite this record, the current increase in marine mammal occurrences around the farms indicates that Sanford should prepare for the possibility of a marine mammal entanglement by developing and putting in place appropriate management practices, such as an entanglement avoidance protocol, along with a marine mammal management plan (see mitigation section below). This consideration is particularly important given the number of internationally recognised endangered or threatened marine mammal species within these waters.

4.10 Effects on birds

Seabirds potentially affected by finfish farms in BGB include shags, gulls, terns and penguins. Diving seabirds may be attracted to the fish and feed pellets. The main potential effect of finfish farms on seabirds is entanglement and subsequent drowning in predator nets or nets used to contain the farmed fish. However, there have been very few reported incidents of seabird

[3] Sanford – pers. Com – Bottlenose dolphins frequently visit the Bay, about once every two weeks. They have been known to swim in between nets and under cages. Both fur seals and NZ seal lion visit the farms and push nets in to grab fish.

deaths as a result of entanglement and this risk can be managed through best management practices (MPI 2013).

Other interactions of seabirds with finfish farms have the potential to be positive, neutral or adverse; however, adverse effects are not presently considered significant (MPI 2013). Potential positive effects include roost sites for seabirds close to foraging areas created by farm structures (**Figure 28**) and additional sources of prey through aggregations of small fish near the farm. Potential adverse effects include disturbance of nesting and feeding birds by noise, boat traffic or artificial lighting.

Effects on seabirds can be mitigated by avoiding placing fish farms near ecologically significant shorebird and wading bird habitats. In the case of BGB there are no ecologically significant areas for these birds.

Figure 28. Shags resting on mussel buoys in BGB.

4.11 Mitigation and monitoring

As discussed above the severity and spatial extent of water column and sediment effects depend on site-specific factors such as water depth and current speed. Environmental effects can be mitigated by managing stocking densities, minimising feed waste and optimising feed conversion ratios. Rotating fish cages provides opportunities for sediments to recover from organic enrichment. It should also be noted that this application is to extend the pens and stocking but within existing leases i.e no more water space is required.

BGB is recognised as an Aquaculture Management Area and Sanford want a world-class facility and sustainable environment that meets best practice. An important aspect of the long term plan should be keeping pace with best practice and new technology. A technological review is recommended every 3-5 years to ensure Sanford continue to use best practice.

4.11.1 Monitoring

The existing monitoring and management plans are appropriate for detecting changes to the water and column, benthic environment and other ecological effects (including birds and mammals) as a result of salmon farming in BGB and are covered by the existing and proposed conditions (including preparation of a monitoring plan). Additional conditions have been recommended regarding annual reporting and adaptive management, as well as a three yearly technology update report, which I support.

In addition to the proposed conditions and existing BGBSFEMP I would recommend a further reference site outside the Bay in outer Paterson Inlet is added to the sampling for water quality. This would provide a reference site for changes at the larger scale that may be impacting on BGB (eg upwelling and nutrient inputs from Foveaux Strait).

4.11.2 Mitigation

Considerable advancements have been made over the last 40 years including improved feeds with better FCRs, use of physical net cleaning instead of antifoulants, improved netting and seabed management.

Fallowing

A robust fallowing plan is critical to the ongoing sustainability of salmon farming in BGB. The additional monitoring conditions for this proposal for the BGBSFEMP include a requirement for reporting on how the results of the monitoring required will be utilised to adapt operational farming practices, including but not limited to the fallowing of individual sites, in the event that monitoring indicates that unforeseen environmental effects may arise. The existing fallowing plan attached to the consent for Farm 246 sets out details including a 7 year fallowing rotational plan (2 years on and 5 years off for fallowing).

This should include the periods after which a farm site should be fallowed and how long they should be left vacant to recover.

Antifoulants and contaminants

Further developments in feed and operations, such as net cleaning and types of nets and net material, should be assessed regularly as part of the technological review included in the recommended new conditions.

Best management practices for mammals

There are a range of additional BMPs regarding the set-up and operation of marine farms that can reduce risks of entanglement and other adverse effects to marine mammals. Many of these practices are already reflected in the Finfish Aquaculture ECOP developed by the New Zealand Salmon Farmers Association (NZSFA 2007) and / or already being undertaken by the farms. The requirements of the additional conditions to be included in the BGBSFEMP include procedures and practices to be implemented to minimise, to the extent practicable, the interactions of marine mammals and seabirds with the farm site.

Triggers and standards

Triggers and standards to be met are set out in the proposed conditions and will be included in the new BGBSFEMP to be submitted. The proposed limits for the water column monitoring in the proposed conditions are for increases in TAN and chl-*a* above those recorded in recent years. The data set to be sued as a baseline should be clarified in the conditions.

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6. REFERENCES

ADS (2016). Big Glory Bay benthic and water quality sampling 2016. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, September 2016.

ADS (2017a). Big Glory Bay benthic and water quality sampling 2016/2017. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, May 2017.

ADS (2017b). Big Glory Bay carrying capacity update, Stewart Island, New Zealand. Volume 1 - Summary of findings. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, May 2017.

ADS (2017c). Big Glory Bay hydrodynamics report. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, May 2017.

ADS (2017d). Big Glory Bay water quality modelling report. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, May 2017.

ADS (2017e). Big Glory Bay benthic footprint report. Report by Aquadynamic Solutions Sdn Bhd to Sanford Ltd, May 2017.

ANZECC (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality.

Baker, C.S.; Chilvers, B.L.; Childerhouse, S.; Constantine, R.; Currey, R.; Mattlin, R.; van Helden, A.; Hitchmough, R.; Rolfe, J. (2013). Conservation status of New Zealand marine mammals, 2013, Department of Conservation.

Barnes, R. K., King, H.; Carter, C. G. (2011). Hypoxia tolerance and oxygen regulation in Atlantic salmon, *Salmo salar* from a Tasmanian population. *Aquaculture*, 318(3), pp.397-401.

Bergheim, A.; Gausen, M.; Næss, A.; Hølland, P. M.; Krogedal, P.; Crampton, V. (2006). A newly developed oxygen injection system for cage farms. *Aquacultural Engineering*, 34(1), pp.40-46.

Bjornn, T. C.; Reiser, D. W. (1991). Habitat requirements of salmonids in streams. American Fisheries Society Special Publication, 19(837), 138.

Black, K.D.; Hansen, P.K; Holmer, M. (2008). Salmon aquaculture dialogue: Working Group Report on benthic impacts and farm siting. (www.worldwildlife.org/aquadialogues)

Bloecher, N.; Olsen, Y.; Guenther, J. (2013). Variability of biofouling communities on fish cage nets: A 1-year field study at a Norwegian salmon farm. Aquaculture, 416, pp.302-309.

Bradford-Grieve, J. M.; Murdoch, R. C.; James, M. R.; Oliver, M.; Hall, J. (1996). Vertical distribution of zooplankton > 39 µm in relation to the physical environment off the west coast of South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 30*:285-300.

Braithwaite, R.A.; Carrascosa, M.C.C.; McEvoy, L. A. (2007). Biofouling of salmon cage netting and the efficacy of a typical copper-based antifoulant. Aquaculture, 262(2-4), pp.219-226.

Brett, J. R.; Blackburn, J. M. (1981). Oxygen requirements for growth of young coho (*Oncorhynchus kisutch*) and sockeye (*O. nerka*) salmon at 15°C. *Canadian Journal of Fisheries and Aquatic Sciences*, *38*(4), 399-404.

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

Brooks, K. M., Stierns, A. R., Mahnken, C. V. W. and Blackburn, D. B. (2003). Chemical and biological remediation of the benthos near Atlantic salmon farms. *Aquaculture 219*, 355-377

Brough TE, Guerra M, Dawson, SM. 2015. Photo-identification of bottlenose dolphins in the far south of New Zealand indicates a ‘new’, previously unstudied population. *New Zealand Journal of Marine and Freshwater Research* 49 (1): 150-158.

Bullard, S. G.; Lambert, G.; Carman, M. R.; Byrnes, J.; Whitlatch, R. B.; Ruiz, G. M.; Miller, R. J.; Harris, L. G.; Valentine, P. C.; Collie, J. S.; Pederson, J.; McNaught, D. C.; Cohen, A. N.; Asch, R. G.; Dijkstra, J. (2007). The colonial ascidian Didemnum sp. A: current distribution, basic biology and potential threat to marine communities of the northeast and west coasts of North America. Journal of Experimental Marine Biology and Ecology 342: 99-108.

Camargo, J. A.; Alonso, Á. (2006). Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: a global assessment. *Environment International*, *32*(6), 831-849.

Carroll EL, Rayment WJ, Alexander AM, Baker CS, Patenaude NJ, Steel D, Constantine R, Cole R, Boren LJ, Childerhouse S 2014. Reestablishment of former wintering grounds by New Zealand southern right whales. Marine Mammal Science 30(1): 206-220.

Carter, K. (2005). The effects of dissolved oxygen on steelhead trout, coho salmon, and chinook salmon biology and function by life stage. *California Regional Water Quality Control Board, North Coast Region*.

Chang, H. (1990). First record of a *Heterosigma* (Raphidophyceae) bloom with associated mortality of cage-reared salmon in Big Glory Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research 24*:461-469.

Chiswell, S. M. (1996). Variability in the Southland Current, New Zealand. *New Zealand Journal of Marine and Freshwater Research 30*: 1-17.

V006T05A004-V006T05A004). American Society of Mechanical Engineers.

Crampton, V.; Hølland, P. M.; Bergheim, A.; Gausen, M.; Næss, A. (2003). Oxygen effects on caged salmon. *Fish Farming International*, Jun 2003, pp. 26–27.

Davidson, R. (2002). Ecological report in relation to an existing mussel farm and a proposed marine farm application site located in Menzies Bay, Banks Peninsula. Report prepared for Te Wharau Investments Ltd.

Davis, G. E.; Foster, J.; Warren, C. E.; Doudoroff, P. (1963). The Influence of Oxygen Concentration on the Swimming Performance of Juvenile Pacific Salmon at Various Temperatures. *Transactions of the American Fisheries Society*, *92*(2), 111-124.

Dempster, T.; Wright, D.; Oppedal, F. (2016). Identifying the nature, extent and duration of critical production periods for Atlantic salmon in Macquarie Harbour, Tasmania, during summer. FRDC Report.

Diggles, B. (2016). Updated disease risk assessment report – relocation of salmon farms in Marlborough Sounds, New Zealand. Prepared for the Ministry for Primary Industries. DigsFish Services Report: DF 16-01. 77p.

DoC (2012). Stewart island/Rakiura Conservation Management Strategy and Rakiura National Park Management Plan. Published by Southland Conservancy of Department of Conservation.

Farrell, A. P.; Richards, J. G. (2009). Defining hypoxia: an integrative synthesis of the responses of fish to hypoxia. *Fish Physiology*, *27*, 487-503.

Fenwick, G. D. (2002). Marine benthic infauna of three bays around banks peninsula. NIWA report No. CHC02/51 prepared for Environment Canterbury.

Johansson, D., Ruohonen, K., Kiessling, A., Oppedal, F., Stiansen, J.E., Kelly, M. and Juell, J.E. (2006). Effect of environmental factors on swimming depth preferences of Atlantic salmon (Salmo salar L.) and temporal and spatial variations in oxygen levels in sea cages at a fjord site. *Aquaculture*, *254*(1-4), pp.594-605.

Forrest, B. (2011). Seabed impacts of Marlborough Sounds salmon farms: Annual monitoring 2000 Report No. 644, prepared by Cawthron Institute for NZ King Salmon.

Forrest, B. M.; Brown, S. N.; Taylor, M. D.; Hurd, C. L.; Hay, C. H. (2000). The role of natural dispersal mechanisms in the spread of Undaria pinnatifida (Laminariales, Phaeophyceae). Phycologia 39: 547-553.

Forrest, B.; Cornelisen, C.; Clement, D.; Keeley, N.; Taylor, D. (2015). Monitoring the environmental effects of aquaculture in the Waikato Coastal Marine Area: Report 2 - Regional aquaculture monitoring priorities and guidance. Waikato Regional Council Technical Report 15/39. Cawthron Report No. 2429. 53 p.

Forrest, B.; Hopkins, G.; Webb, S.; Tremblay, L. (2011). Overview of marine biosecurity risks from finfish aquaculture development in the Waikato region. Waikato Regional Council Technical Report 11/22. Cawthron Report No. 1871. 78 p.

Forrest, B.; Keeley, N.; Gillespie, P.; Hopkins, G.; Knight, B.; Govier, D. (2007). Review of the ecological effects of marine finfish aquaculture. Prepared for the Ministry of Fisheries. Cawthron Report No. 1285. 73 p plus appendices.

Fuller, S. A.; Henne, J. P.; Carmichael, G. J.; Tomasso, J. R. (2003). Toxicity of ammonia and nitrite to the Gila trout. North American Journal of Aquaculture, 65(2), 162-164.

Gerwing,T. G.; Allen Gerwing, A. M.; Hamilton, D. J.; Barbeau, M. A. (2015). Apparent redox potential discontinuity (aRPD) depth as a relative measure of sediment oxygen content and habitat quality. International Journal of Sediment Research 30(1): 74-80.

Green, M.; Cornelisen, C. (2016). Marine water quality standards for the Waikato Region – Literature review. Report prepared for Waikato Regional Council by Streamlined Environmental and Cawthron Institute, June 2016.

Hamlin, H. J. (2006). Nitrate toxicity in Siberian sturgeon (*Acipenser baeri*). *Aquaculture*, *253*(1), 688-693.

Hartstein, N. D.; Oldman, J. W. (2015). Nitrogen levels and adverse marine ecological effects from aquaculture. New Zealand Aquatic Environment and Biodiversity Report No. 159. Ministry for Primary Industries.

Heath, R. A. (1975). Stability of some coastal inlets. *New Zealand Journal of Marine and Freshwater Research 9(4)*: 449-57.

Hickey, C. W.; Martin, M. L. (2009). *A review of nitrate toxicity to freshwater aquatic species*. NIWA Client Report: HAM2009-099. Prepared for Environment Canterbury.

Houtman, T. J. (1974). A note on the hydraulic regime in Foveaux Strait. *New Zealand Journal of Science 9:* 472-83.

James et al (2016). James, M.; Green, M.; Oldman, J. (2016). South-west sub-regional wastewater treatment plant application – Assessment of the sensitivity of the receiving environment of the south-west Manukau harbour to predicted contaminants. AES report prepared for Watercare Services ltd.

Keeley et al. (2009). Sustainable aquaculture in New Zealand: Review of the ecological effects of farming shellfish and other non-finfish species. Prepared for Ministry of Fisheries. Cawthron Report No. 1476.

Keeley, N.; Taylor, D. (2011). The New Zealand King Salmon Company Ltd: Assessment of environmental effects – Benthic prepared for New Zealand King Salmon Company Ltd. Cawthron Report 1983.

Keeley N.; Forrest BM.; Crawford, C.; MacLeod C. (2012). Exploiting salmon farm benthic enrichment gradients to evaluate the regional performance of biotic indices and environmental indicators. *Ecological indicators 23*:453-466.

Keeley N.; Forrest BM.; MacLeod C. (2015). Benthic recovery and re-impacted responses from salmon farm enrichment: Implications for farm management *Aquaculture 435*: 412-423.

Keeley N.; Forrest B.M.; Crawford C.; Hopkins GA.; MacLeod C. (2014). Spatial and temporal dynamics in macrobenthos during recovery from salmon farm induced organic enrichment: when is recovery complete? *Marine Pollution Bulletin 80* (1-2) 250-262

Kerroux, A.; Hartstein, N. (2017). Big Glory bay carrying capacity update, Stewart Island, New Zealand: Hydrodynamic modelling and flushing. Report by Aquadynamic Solutions Sdn Bhd for Sanford Ltd.

Key, J. (2001). Growth and condition of the greenshell mussel Perna canaliculus, in Big Glory Bay, Stewart Island: relationships with environmental parameters. MSc thesis Otago University.

Kincheloe, J. W.; Wedemeyer, G. A.; Koch, D. L. (1979). Tolerance of developing salmonid eggs and fry to nitrate exposure. *Bulletin of Environmental Contamination and Toxicology*, *23*(1), 575-578.

Lalas, C. (1983). Comparative feeding ecology of New Zealand marine shags (*Phalacrocoracidae*). Unpublished PhD thesis, University of Otago, Dunedin.

Macleod, C. K., Crawford, C. M. and Moltschaniwskyj, N. A. (2004). Assessment of long term change in sediment condition after organic enrichment: defining recovery. *Marine Pollution Bulletin 49*, 79-88.

Marrafini, M. L.; Ashton, G. V.; Brown, C. W.; Chang, A. L.; Ruiz, G. M. (2017). Settlement plates as monitoring devices for non-indigenous species in marine fouling communities. Management of Biological Invasion, 8.

Moore (1999). Foraging range of the Yellow-eyed penguin *Megadyptes antipodes*. *Marine Ornithology 27*: 49-58.

Morrisey, D.; Keeley, N.; Elvines, D.; Taylor, D. (2016a). Firth of Thames and Hauraki Gulf enrichment stage mapping. Cawthron Report No. 2824.

Morrissey, D.; Plew, D. R.; Seaward, K. (2011). Aquaculture readiness data Phase II. MAF technical paper, (2011/68), p.58.

MPI (2013a). Literature review of ecological effects of aquaculture. Ministry for Primary Industries, Wellington, New Zealand. 19 p.

MPI (2013b). Overview of ecological effects of aquaculture. Ministry for Primary Industries, Wellington, New Zealand. 79 p.

MPI (2016). Aquaculture decision report – Sanford Ltd, Coastal permit ES207253, Big Glory Bay, Stewart Island. Report by MPI.

Oppedal, F., Dempster, T.; Stien, L. H. (2011). Environmental drivers of Atlantic salmon behaviour in sea-cages: a review. *Aquaculture*, 311(1), pp.1-18.

Pedersen, C. L. (1987). Energy budgets for juvenile rainbow trout at various oxygen concentrations. *Aquaculture*, 62(3), pp.289-298.

Pridmore, R. D.; Rutherford, J. C. (1990). Factors influencing phytoplankton abundance in Big Glory Bay, Stewart Island: a modelling study. Water Quality Centre Consultancy Report No 6107. Hamilton. 30pp.

Pridmore, R. D.; Rutherford, J. C. (1992).Modelling phytoplankton abundance in a small enclosed bay used for salmon farming. *Aquaculture and fisheries management 23*:525-542.

Remen, M.; Oppedal, F.; Imsland, A. K.; Olsen, R. E.; Torgersen, T. (2013). Hypoxia tolerance thresholds for post-smolt Atlantic salmon: dependency of temperature and hypoxia acclimation. *Aquaculture*, 416, pp.41-47.

Robertson, B.M.; Stevens, L.;, Robertson, B.; Zeldis, J.; Green, M.O.; Madarasz-Smith, A.; Plew, D.; Storey, R.; Oliver, M. (2016). NZ Estuary Trophic Index Screening Tool 2. Determining Monitoring Indicators and Assessing Estuary Trophic State. Prepared for Envirolink Tools Project: Estuarine Trophic Index, MBIE/NIWA Contract No: C01X1420, 68 pp.

Robertson, H. A.; Baird, K.; Dowding, J. E.; Elliott, G. P.; Hitchmough, R. A.; Miskelly, C. M.; McArthur, N.; O'Donnell, C.F.J.; Sagar, P. M.; Scofield, R.P.; Taylor, G. A. (2016). Conservation status of New Zealand birds, 2016. *New Zealand Threat Classification Series 19.*

Rutherford, J. C.; Pridmore, R. D.; Roper, D. S. (1988). Estimation of sustainable salmon production in Big Glory Bay, Stewart Island. 59 p. + 14 figs. Prepared for MAFFish. Report number T7074/1.

Sanford Ltd (2017). Big Glory bay salmon farms. Change of conditions Application and assessment of Environmental Effects. Submitted to Environmental Southland.

Sim-Smith, C.; Kelly, S.; Carbines, G. (2017). Ecological assessment of a proposed mussel farm site in the western Firth of Thames. Coast and Catchment Ltd report to Watercare Services Ltd.

Stenton-Dozey, J. (2013). Assessment of the benthic environment for two proposed fish farm sites at Stewart Island. Report CHC2013-140 prepared for Sanford Ltd.

Stenton-Dozey, J. (2014 – revised 2016). Assessment of the water column and benthic environments of marine farms in Big Glory Bay, Stewart Island from July 2013 to June 2014. Report CHC2014-120 prepared for Sanford Ltd.

Stenton-Dozey, J. (2015). Assessment of the water column and benthic environments of marine farms in Big Glory Bay, Stewart Island from July 2014 to June 2015. Report CHC2015-105 prepared for Sanford Ltd.

Stenton-Dozey, J.; Brown, S. (2009). Monitoring of environmental effects of longline mussel farming, Big Glory Bay, Stewart Island: Twelfth annual report. NIWA report CHC2009-165 prepared for Sanford Ltd.

Stenton-Dozey, J.; Brown, S. (2010). Monitoring of environmental effects of longline mussel farming, Big Glory Bay, Stewart Island: Thirteenth annual report. NIWA report CHC2010-139 prepared for Sanford Ltd.

Stenton-Dozey, J.; Brown, S.; Morrisey, D.; Cole, R. (2012). Assessment of the water column and benthic environments of marine farms in Big Glory Bay, Stewart Island from August 2011 to June 2012. Report CHC2012-108 prepared for Sanford Ltd.

Stevens, E. D., Sutterlin, A.; Cook, T. (1998). Respiratory metabolism and swimming performance in growth hormone transgenic Atlantic salmon. *Canadian Journal of Fisheries and Aquatic Sciences*, 55(9), pp.2028-2035.

Stien, L. H.; Bracke, M.; Folkedal, O.; Nilsson, J.; Oppedal, F.; Torgersen, T.; Kittilsen, S.; Midtlyng, P. J.; Vindas, M. A.; Øverli, Ø.; Kristiansen, T. S. (2013). Salmon Welfare Index Model (SWIM 1.0): a semantic model for overall welfare assessment of caged Atlantic salmon: review of the selected welfare indicators and model presentation. *Reviews in Aquaculture*, 5(1), pp.33-57.

U.S. Environmental Protection Agency (USEPA). (1986). Ambient Water Quality Criteria for Dissolved Oxygen. Office of Water Regulations and Standards, Criteria and Standards. Washington, DC.

U.S. Environmental Protection Agency (USEPA). (1989). Ambient Water Quality Criteria for Ammonia (Saltwater)-1989. EPA 440/5-88-004. Office of Water Regulations and Standards, Criteria and Standards. Washington, DC.

Vaughan, M.; Walker, J. (2014). Marine water quality annual report: 2014. Auckland Council Technical Report, TR2015/032.

Vincent, W. F.; Howard-Williams, C.; Tildesley, P.; Butler, E. (1991). *New Zealand Journal of Marine and Freshwater Research Vol 25*:21-42.

Williamson, R. B.; Hickey, C. W.; Robertson, B. M. (2015) Preliminary assessment of limits and guidelines available for classifying coastal waters. Report by Diffuse Sources, NIWA and Wiggle –Coastal Management to Auckland Council, 129 pp.