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Nitrogen Levels and Adverse Marine Ecological Effects from Aquaculture

New Zealand Aquatic Environment and Biodiversity Report No. 159

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This review has highlighted the range of approaches that have been adopted worldwide for understanding the potential impacts of aquaculture developments in terms of the nitrogen loadings in the receiving marine environment. Four international sites with comparable physical settings and levels of aquaculture development to four key New Zealand aquaculture areas were reviewed. These sites are:

- Macquarie Harbour (Tasmania) which is comparable to Big Glory Bay;
- Saldanha Bay (South Africa) which is comparable to Tasman Bay and Golden Bay;
- Huon River and D'Entrecasteaux Channel (Tasmania) which is comparable to the Marlborough Sounds; and
- Faroe Islands which are comparable to the Firth of Thames.

For each of these sites a summary of the current understanding of the hydrodynamics and nutrient budgets has also been provided. The review has highlighted a lack of data for some of the international sites but has also shown that a lot of information has already been gathered at the New Zealand sites to provide a good understanding of the bay wide nutrient budgets.

The report also provides an overview of the potential adverse effects of nitrogen within the water column and seabed and on benthic communities. It also details a number of international case studies where nitrogen loading has been regulated to minimise the impacts of proposed new aquaculture facilities or the expansion of existing aquaculture areas.

A variety of modelling methodologies and management frameworks have been used to limit nutrient loads and ensure the long-term health of coastal areas and to ensure sustainable aquaculture production. All of the case studies follow similar methodologies that:

1. Set some environmental trigger/s at which point a review of production, management or some other intervention occurs,
2. Provide an understanding of the dynamics of the system (through modelling and/or monitoring) and
3. Quantify the impacts of the existing and/or proposed aquaculture production in the context of the natural variability of the system.

The modelling methodologies and monitoring regimes selected for a region must be relevant to the hydrodynamics, nutrient loads and to the planned extent of aquaculture in an area.

Simple box model/mass balance approaches (as used in the Scotland) can provide good management guidelines, but only in relatively semi-enclosed systems where good information on aquaculture impacts is available. Such an approach could not be universally applied in New Zealand because many of our aquaculture areas are relatively open, highly connected systems (e.g. the Firth of Thames).

The use of fully coupled, three dimensional hydrodynamic, sediment and biogeochemical models in the Tasmanian case studies is the approach that this report recommends should be applied in New Zealand. Such models provide quantification of the impact(s) of aquaculture expansion in the context of other sources of nitrogen (rivers and nutrient upwelling) and the natural variability. Model results help determine how changes in seabed and water column enhancement of nutrients may impact on marine ecosystems. The models can also be used to estimate the likely recovery period of the seabed and

estimate changes in the phytoplankton volume and community structure due to additional nutrient loading, which provides useful guidance for both farmers and regulators.

In the context of the New Zealand aquaculture industry, the RMA framework promotes the concepts of monitoring and adaptive management as tools for understanding and assessing the potential impacts of aquaculture development. The international case studies presented show the importance of understanding not only the local effects of aquaculture (i.e. cage/cage and farm/farm interactions and impacts) but also the wider cumulative effects of aquaculture development particularly in areas already impacted by relatively high loadings of nutrients.

It is crucial that a wide range of field data spanning a broad range of conditions are collected both in the pre-development and the development phases. The pre-development data allows calibrated models to be established and assists in informing the development of operational guidelines and management triggers in relation to the proposed development. Data collection (and ongoing modelling) during the development phase allows the impacts of the development to be assessed in terms of agreed trigger levels and for appropriate adaptive management procedures to be implemented.

The coverage and duration of field data collection will be very site specific but it is recommended that a minimum of 12 months of data is collected for the purposes of model calibration and for defining baseline water quality conditions. The number of stations to be sampled and monitored should be decided on a case by case basis and to some extent will also depend on the data that has already been collected at a site. Parameters to be monitored should include tides, currents, waves, dissolved oxygen, nutrients (NH_4, NO_3), chlorophyll-a, temperature, salinity, sediment characteristics (sediment grain size, POC/PON content) and if possible, sediment denitrification rates. Field sampling should be designed around findings from models and should provide data in both the near-field and far field. Data collected should be used by regulators (in terms of management triggers and consent conditions) and farmers (in terms of day-to-day management). Where possible a modelling framework that combines the collection of in-situ real time data and remote sensing information with forecast models should be developed. Building on this framework it is recommended that a decision support system should be developed to enable all stakeholders (i.e. regulatory and operational) to obtain and understand field data and model outputs via web based applications.

The trigger level approach adopted overseas (and being developing in New Zealand) should become the standard for assessing how a system is responding to the potential impacts of aquaculture and should be an integral part of adaptive management plans. The process of establishing such triggers requires developing a thorough understanding of the marine system being considered through a combination of modelling and monitoring. With such tools in place potential aquaculture development impacts and the role of natural variability can be thoroughly understood and quantified.

1. Introduction

The New Zealand aquaculture industry began over 40 years ago and is today a significant primary industry generating over $400 million in revenue, with the goal of revenue reaching $1 billion by 2025 (New Zealand Aquaculture, 2012). The current focus of the industry is on three primary species - Green-lipped Mussels™, King Salmon and Pacific Oysters (Table 1). In total, the allocated space for marine-based aquaculture totals more than 29 000 hectares. Existing farms account for nearly half of this total space with 55% of the active space being in the near shore and the remaining 45% in the open-ocean (MPI 2013).

Notified new space and Maori Aquaculture Settlement areas account for 8% of the total allocated space for marine-based aquaculture.

Just over 40% of the total allocated space is assigned within applications frozen under the 2001 aquaculture moratorium. These applications are being reviewed under the new regulatory framework being considered as part of the Aquaculture Reform Act (2011) and associated legislation.

Industry growth is predicted to occur across all of the three key species with potential diversification to finfish species (e.g. kingfish and hapuku), seaweeds, sea cucumbers and kina. Growing competition for resources and the social interest around aquaculture means that an understanding of the cumulative effects of aquaculture on the marine environment is critical.

Table 1: Production and revenue for New Zealand Aquaculture 2011 (New Zealand Aquaculture, 2012).

	Green-lipped Mussels	King Salmon	Pacific Oysters
Harvested product (greenweight tonnage)	101 311	14 037	1 804
Revenue ($NZ millions)	253	128	25

The potential effect of nutrient loading on the marine environment is a key issue affecting the sustainability of the industry. Total nitrogen and phosphorous loads to the coast in New Zealand have been estimated at 167 300 and 63 100 t/yr, respectively (Elliot et al. 2005). A long term increase in nitrogen loading in New Zealand streams and water ways reflects similar trends overseas (Ballantine & Davies-Colley 2009). The potential cumulative effect of existing land use and additional nitrogen from aquaculture (particularly finfish) on the marine environment has been well documented (e.g. Pridmore & Rutherford 1990; Pohle et al. 1994; Mirto et al. 2000). This additional nitrogen loading has the potential to impact on the overall health of the marine environment by altering the carrying capacity of a system, and limiting the number of finfish or bivalves that can be harvested in a sustainable fashion (Montiero et al. 1998; Hall et al. 2000; Zeldis et al. 2010; Hartstein et al. 2011).

With these trends in mind, the Ministry for Primary Industries (MPI) commissioned a study to examine the potential ecological effects of aquaculture derived nitrogen in the context of New Zealand's marine environment and its expanding aquaculture industry. The study involved an examination of chosen New Zealand aquaculture sites and a literature review of comparable international aquaculture operations within temperate embayments where information on nitrogen levels, and management frameworks and standards are available.

The literature review examines four broad questions and these comprise the main sections of this report:

1. What overseas temperate coastal marine systems could be used to inform questions on nitrogen levels and adverse ecological effects from aquaculture in New Zealand (Section 2)?

2. What levels of nitrogen inputs would cause negative ecological/toxicological impacts (Section 3)?
3. Have regional nitrogen management levels for receiving marine coastal waters been implemented anywhere comparable to New Zealand, and how were these levels set (Section 4)?
4. Are there any guidelines or lessons from overseas that can be generally applied to New Zealand for understanding the ecological effects of nitrogen loading (from aquaculture and other sources) on our coastal marine environment and, conversely, what needs to be determined with case-by-case regional modelling studies in order to plausibly determine likely nitrogen effects (Sections 4 and 5)?

Section 6 of the report provides a summary of the findings of this literature review. Through the use of numerical modelling, monitoring and adaptive management methods, these will assist New Zealand aquaculture consenting authorities in determining what level of nitrogen loading from aquaculture activities may lead to adverse effects.

2. Comparison of International and New Zealand Coastal Marine Systems

2.1 Overview

The aim of this section is to compare selected New Zealand aquaculture sites with similar international sites that match the general scale of aquaculture production, the nitrogen budgets and the physical settings (i.e. hydrodynamics, geomorphology, wave and wind climates and oceanic/estuary exchange mechanisms) of the New Zealand sites. New Zealand has a temperate climate and where possible preference was given to international sites within a similar climatic region. It was found that no international sites provided an exact match to the selected New Zealand sites, but the comparisons still provide a useful background to understand how nitrogen loading from aquaculture production is managed internationally (Section 4).

Internationally, 40% of aquaculture production is marine based (Bostock et al. 2010) with a very broad range of finfish and shellfish species being commercially farmed. New Zealand's aquaculture industry is based on three key species - Green-lipped Mussels™, King Salmon and Pacific Oysters. Green-lipped Mussels are endemic to New Zealand and are grown in Tasman Bay, Golden Bay, the Hauraki Gulf, Firth of Thames and Marlborough Sounds. King Salmon are produced in the Marlborough Sounds, Big Glory Bay (Stewart Island) and Canterbury. Pacific Oyster aquaculture occurs along the Coromandel Peninsula, Kaipara Harbour, Northland and the Marlborough Sounds (Keeley et al. 2009).

The nitrogen budget of a particular marine area is determined by quantifying the catchment loads influencing an area, general mixing characteristics of the area, the potential for upwelling of oceanic waters (bringing in nutrient rich waters) and the scale and type of aquaculture in place or being planned. Ideally the partitioning between dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON) and particulate organic nitrogen (PON) fluxes within the embayment being considered should be quantified. The primary sources and sinks of nitrogen to embayments (as shown in Figure 1) include:

- Riverine/catchment discharges, including agriculture and horticulture runoff (the largest source of nitrogen into the coastal zone, Elliot et al. 2005), residential wastewater and storm-water overflow;
- Inputs from aquaculture farms (e.g. excess feed and fish faeces);
- Losses from aquaculture (e.g. removal by shellfish);
- ocean sources/general primary productivity;
- waste water discharge;
- groundwater (e.g. Giblin & Gaines 2009); and
- denitrification of sediments.

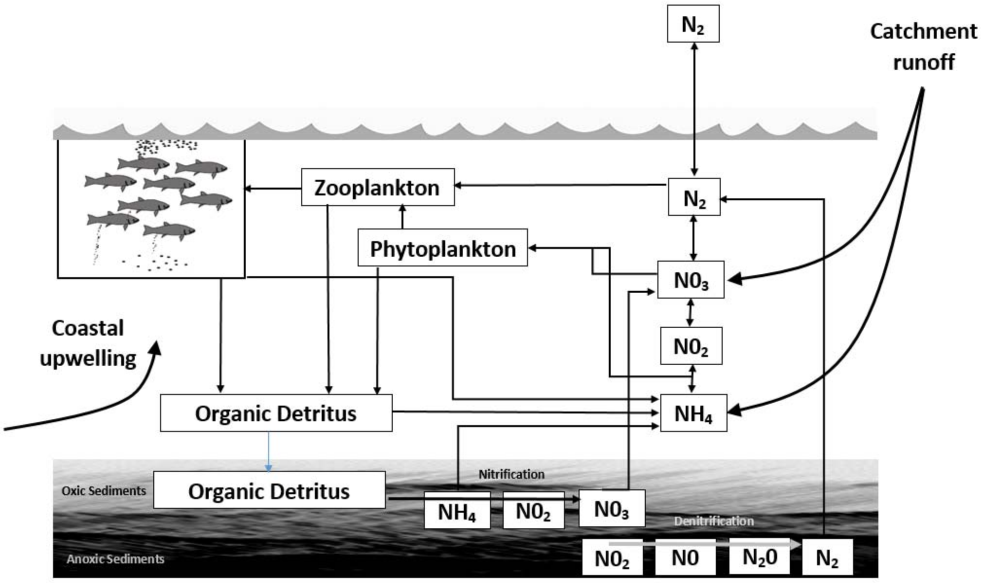

Figure 1: Schematisation of the main Nitrogen processes relating to aquaculture. See Glossary (Section 9) for details of components.

Where data was not found in the literature for the net effect of mussel production for an area on the nitrogen budget it has been assumed to be nitrogen neutral. Technically they do remove some nitrogen from the embayment but the release of ammonium (through faeces and pseudo faeces excretion) means that it is very difficult to quantify their true net loss or gain to the system in terms of feedback to primary production (Lehane & Davenport 2002; Jeffrey et al. 2006).

New Zealand Sites

Aquaculture sites within New Zealand were categorised according to the broad classification system of Hume et al. (2007) using the wave climate at the site and the connectivity of the site to both oceanic and land derived sources of nutrient. The classification system of Hume et al. (2007) uses the following specific features:

- the bathymetry and depth of the estuarine bay;
- the morphological shape of the bay;
- the direction of the water flow in contrast with the land;
- the exposure of the estuarine bay to the sea;
- the extent of the intertidal and sub tidal areas;
- the flushing rate/ resident times within the embayment;
- the type of sedimentation existing within the bay;
- common terms or names of the embayments (examples; fjords, coastal lakes etc.);
- the tidal prism of the embayment;
- the volume of river flow versus the volume of sea water within the embayment;
- mixing of the water column or stratification; and

- the presence of salt wedges and sills.

This approach assumes that the physical character of a marine system reflects a range of physical processes occurring within it (e.g. evaporation, inflows and outflows of oceanic and fresh water into the estuarine basin, stratification, flushing, circulation, mixing and sedimentation).

Based on the above criteria, the type and scale of aquaculture being carried out in New Zealand were considered, and in consultation with MPI the following representative New Zealand areas were selected:

- Tasman Bay and Golden Bay (Figure 2);
- Marlborough Sounds (Figure 2);
- Firth of Thames including the inner Hauraki Gulf (Figure 3); and
- Big Glory Bay (Figure 4).

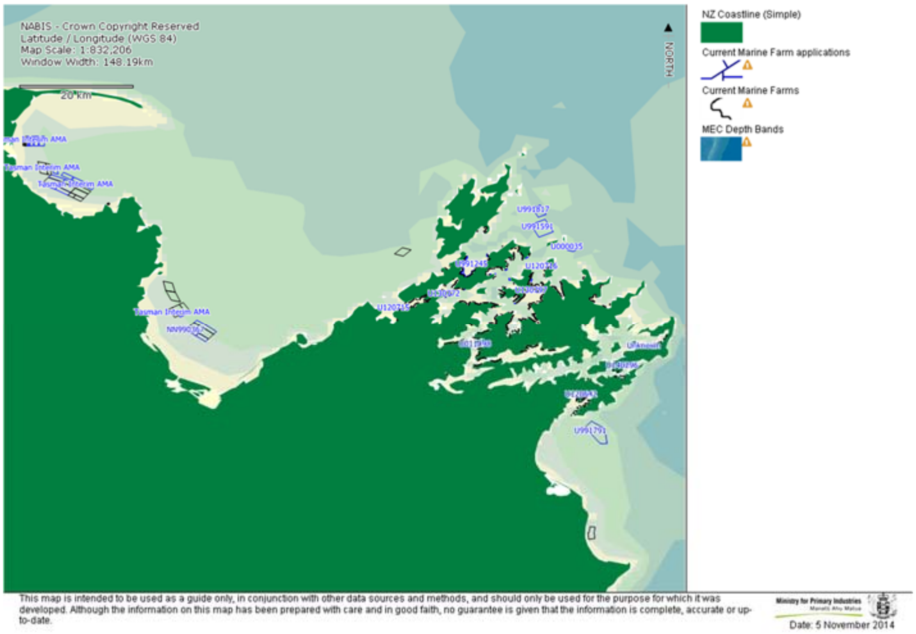

Figure 2: Existing marine farms and applications - Tasman and Golden Bay and Marlborough Sounds (MPI NABIS mapping).

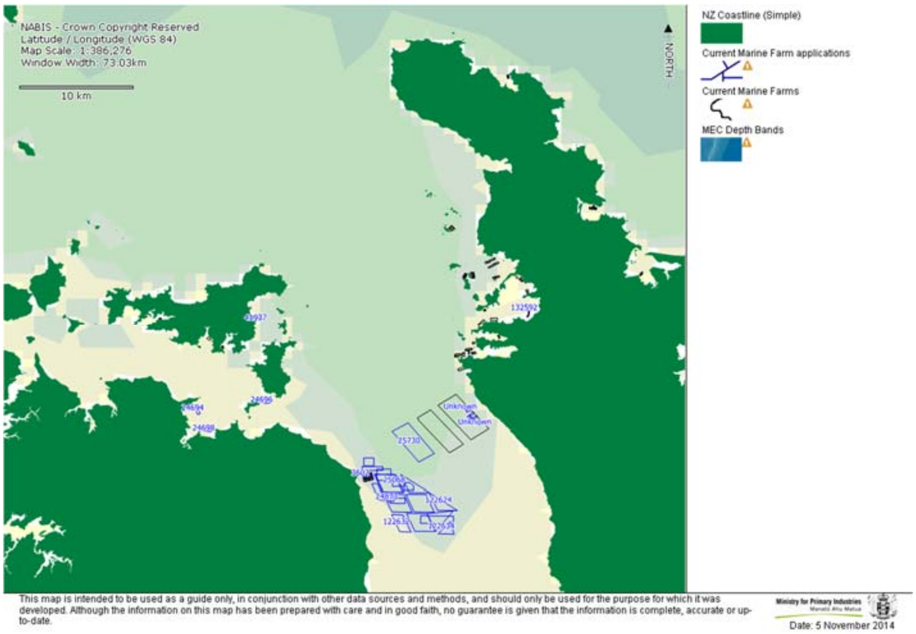

Figure 3: Existing marine farms and applications – Firth of Thames (MPI NABIS mapping).

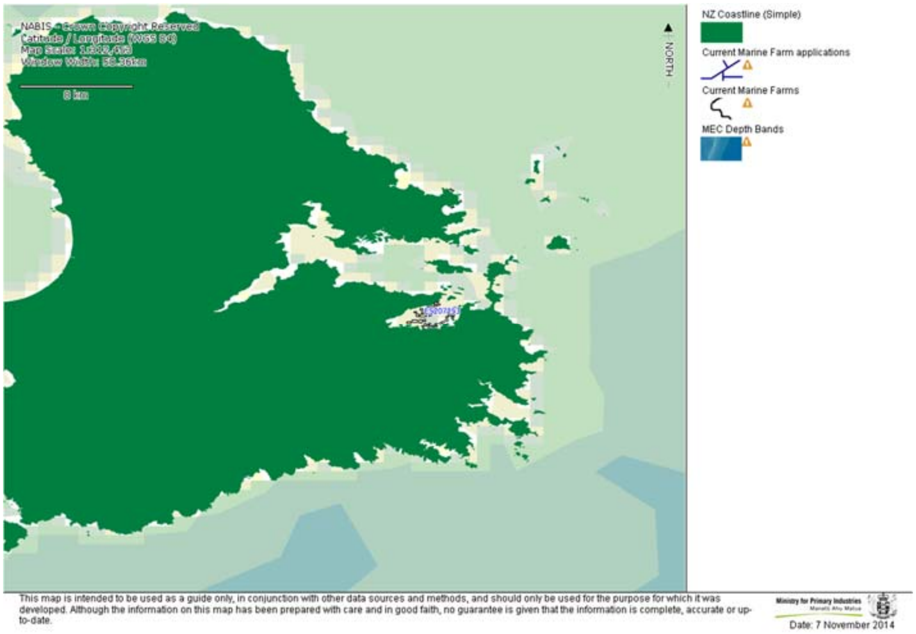

Figure 4: Existing marine farms and applications – Big Glory Bay, Stewart Island (MPI NABIS mapping).

International Sites

International sites were selected based on how similar they were to the New Zealand sites. To ensure that relevant sites were selected the following elements were investigated;

- **The general physical setting.** International sites within a similar climatic region and with a similar general physical setting (e.g. potential for oceanic upwelling, wave climate, oceanic/estuarine exchange processes) were considered.
- **Relative role of aquaculture derived nitrogen compared to other sources.** Coastal areas are generally exposed to multiple nitrogen sources (e.g. river runoff, upwelling from nearby offshore areas). It is important to understand the potential impacts of inputs from aquaculture in context of these other sources of nitrogen.
- **Availability of data**. To quantify the relative roles of various sources of nitrogen, data on each of those sources must be available. For some of the international sites initially considered insufficient data was available to carry out a thorough review of the relative role of the potential impacts of aquaculture.
- **The type of aquaculture.** The type of aquaculture involved is important as nitrogen inputs and outputs differ depending on the type of aquaculture and feed used. For example mussels are filter feeders and could in fact reduce the nutrient influx within an estuarine area (Hartstein 2003). For salmon farms, the sedimentation of faecal waste is one of the key potential impacts. To quantify how farm waste is transported away from a farm site it is important to understand the current and wave regime at the site (Hartstein et al. 2010).

An extensive literature search using the above considerations produced the following international sites that are comparable with each of the representative New Zealand sites and for which scientific information is available (Figure 5);

- Macquarie Harbour (Tasmania) which is comparable to Big Glory Bay;
- Saldanha Bay (South Africa) which is comparable to Tasman Bay and Golden Bay;
- Huon River and D'Entrecasteaux Channel (Tasmania) which is comparable to the Marlborough Sounds; and
- Faroe Islands which are comparable to the Firth of Thames.

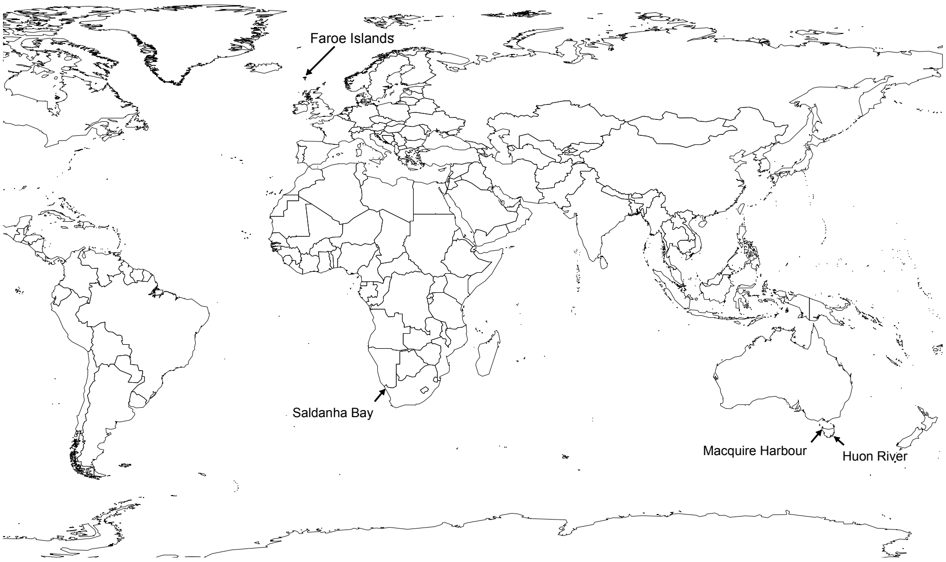

Figure 5: The location of the four international aquaculture sites.

2.2 New Zealand Site Description - Tasman and Golden Bay

Golden Bay

Two aquaculture management areas (AMA 1 Waikato and AMA 2 Puramakauare) have been established within Golden Bay for the production of New Zealand's green-lipped mussel. Mussel spat collection operations also occur within Golden Bay, as well as a commercial scallop fishery which is not currently fished due to uneconomic densities. Golden Bay borders the Abel Tasman National Park, the Farewell Spit Wildlife Sanctuary, and contains the Tonga Island Marine Reserve and receives significant nutrient loadings from farming within its catchment.

Overview of Hydrodynamics of Golden Bay

Golden Bay is a relatively shallow bay with a mean depth of 20 meters (Figure 6). Mean significant wave height is less than 0.4 m and wave heights very rarely exceed 1.5 meters (Bradford-Grieve et al. 1994, Zeldis et al. 2011) with a predominance of short period wind driven waves due to the sheltering effects of Farewell Spit. The tidal range is one of the largest in New Zealand with a mean range of 2.7 m and a spring tide range of 4.2 m.

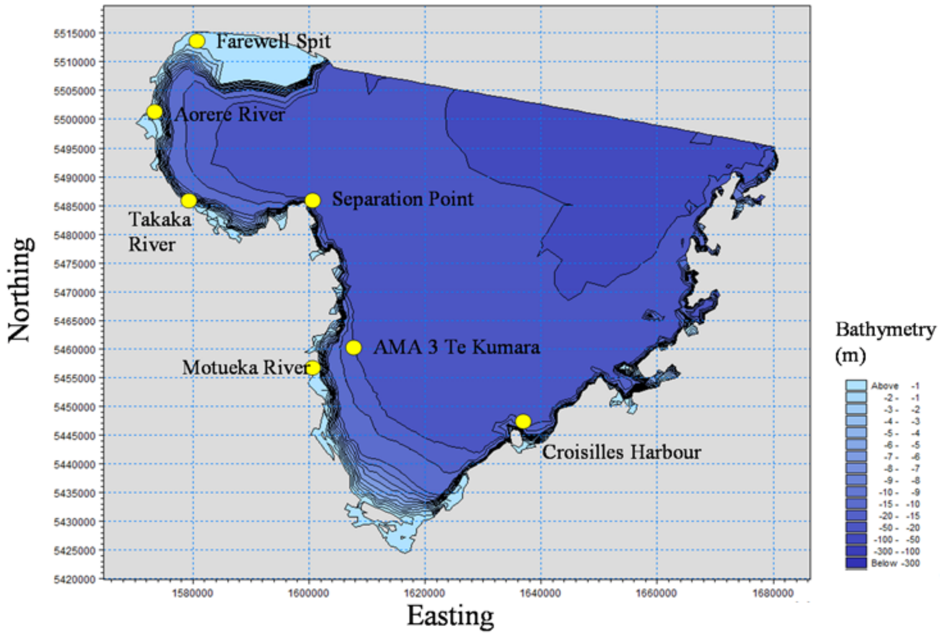

Figure 6: Tasman and Golden Bay bathymetry.

Tidal flow in Golden Bay is reasonably well understood as a result of modelling work and supporting field data (Heath 1976a, Bowman et al. 1980, Plew et al. 2003, 2005, Tuckey et al. 2006). Field observations by Heath (1969, 1973, 1976a) indicated a clockwise mean circulation with water entering the bay in a broad stream north of Separation Point and leaving in a narrower stream along the southern edge of Farewell Spit (Bowman et al. 1980). More recent modelling studies (Plew et al. 2003 and Tuckey et al. 2006) also predicted similar circulation patterns with residual clockwise residual currents of less than 0.05 m/s occurring in the Bay.

Due to the relatively shallow depths within the bay, winds can have a significant effect on bay wide circulation, increasing residual currents and therefore altering the flushing characteristics of the Bay. Hydrodynamic modelling by Plew et al. (2003) estimated a flushing rate of approximately 7 days consistent with the salt budget approach used by Zeldis (2008a) which provided an estimated residence time of 11 days.

Overview of Nitrogen Loads to Golden Bay

The main rivers entering Golden Bay are the Aorere River and Takaka River, with a combined discharge of nitrogen of approximately 1080 tonnes per year, while ocean flux and denitrification were estimated at 2600 and 3200 tonnes respectively (Snelder et al. 2004; Zeldis 2008a, Zeldis et al. 2011). Ground water nutrients entering the bay are thought to be less than 40 tonnes per year (Zeldis 2008a), and waste water is estimated at approximately 40 tonnes of nitrogen per year (Table 2).

Given that Golden Bay aquaculture areas are used for mussel farming, the loading of these aquaculture areas to the bay was assumed to be negligible.

Table 2: Estimated annual nitrogen loads (tonnes) for Golden Bay.

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
1 100[a]	40[b]	40[b]	2 200[a]	-3 400[a]	0

[a]MacDiarmid et al. (2013), [b]Zeldis 2008a.

*Net Ocean flux refers to dissolved inorganic nitrogen (DIN) plus dissolved organic nitrogen (DON).

Tasman Bay

Tasman Bay (Figure 7) is bounded by Abel Tasman National Park to the west and the Marlborough Sounds to the east. Tasman Bay has an area of approximately 1387 km^2 with a catchment area of approximately 3876 km^2 (Hartstein & Senior 2006).

A single aquaculture management area (AMA 3 Te Kumara) is established within Tasman Bay and sits 5 km offshore of the Motueka River. The AMA will be used for the production of New Zealand Green-lipped Mussel along with both scallop and mussel spat collection operations. Tasman Bay is also a commercial and recreational fishing area targeting scallops, oysters and inshore finfish species. Significant areas of the catchment are agricultural and a number of horticulture crops are grown in the region. Fertilizers used for these activities will be a source of nitrogen runoff into Tasman Bay (Gillespie et al. 2011a).

Overview of Hydrodynamics of Tasman Bay

Tasman Bay has a mean depth of 40 meters making it a relatively shallow bay. Because it is exposed to the north-west, wave heights within Tasman Bay are generally greater than in Golden Bay. The one-year average recurrence interval wave height at the Nelson Port has been estimated at 3.9 m (Goodhue et al. 2012). Tidal currents in the outer Bay flow in a north-east direction during the ebbing tide and a south-west direction during a flooding tide (Hartstein & Senior 2006). Towards the southern end of the bay, residual tidal currents are directed clockwise while residual flows north of Croisilles Harbour are directed to the north-east (Figure 7).

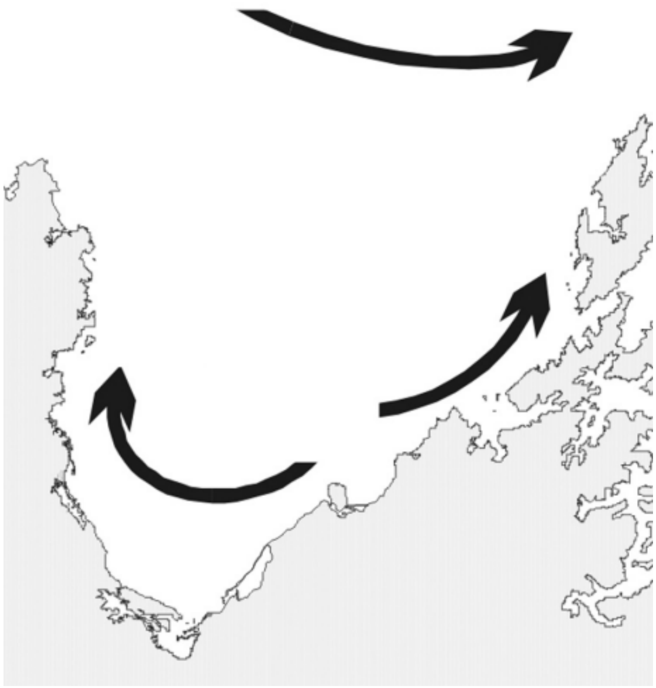

Figure 7: Schematised residual currents in Tasman Bay (from Tuckey & Gibbs, 2004, reproduced with permission).

The Motueka River is the main river entering Tasman Bay and has a mean annual flow of 68 m^3/s (Bowden et al. 2004). Flows of greater than 200 m^3/s are often observed after moderate rain (Tuckey et al. 2006). During flood events the Motueka River plume covers portions of the Tasman Bay (Tuckey et al. 2006) leading to stratification that is broken down by the combined effects of winds and waves (MacKenzie & Adamson 2003). Numerical modelling indicates that the flushing time of the bay ranges from 30–41 days (Hartstein & Senior 2006).

Overview of Nitrogen Loads to Tasman Bay

Nutrient sources within Tasman Bay include oceanic sources transported into the Bay, terrestrial nutrients, and to a lesser extent, atmospheric input through rainfall.

As with other temperate coastal areas nitrogen is generally considered to be the most limiting nutrient for phytoplankton production in Tasman Bay (MacKenzie et al. 2003). Nutrients contained within the Motueka River plume nourish coastal phytoplankton productivity within a large plume-affected region in the vicinity of the AMAs and the western portion of the Bay (MacKenzie & Adamson 2003; Hartstein & Senior 2006, Tuckey et al. 2006). The rates at which nutrients are internally recycled (i.e. Figure 1) within Tasman Bay have also been well studied.

The total input of nitrogen to Tasman Bay was estimated to be 890 tonnes during 2005 (Gillespie et al. 2006). This consisted of 313 tonnes from the Motueka River, 273 tonnes from other tributaries feeding into the Bay and 304 tonnes from the four main point source wastewater discharges (Gillespie et al. 2001). By extrapolating observed benthic denitrification rates at two sites within Tasman Bay Christensen et al. 2003 estimated that a total of 1800 tonnes of nitrogen could be removed from the system per year across the Bay. Estimates by Zeldis et al. (2011) predicted that 2800 tonnes of denitrification per year could occur across the Bay while MacDiarmid et al. (2013) gave a denitrification rate of 2900 tonnes per annum. Zeldis et al. (2011) estimated that oceanic exchange contributed approximately 2500 tonnes to the nitrogen budget for the Bay while MacDiarmid et al. (2013) gave a rate of 2300 tonnes per annum (Table 3). The groundwater contribution to the nitrogen budget of Tasman Bay is relatively small at around 10% of the assumed loading from the river input (Zeldis 2008a).

It has been assumed that because the aquaculture sites in Golden and Tasman Bay are mussel farming areas, the loading from these aquaculture farms is zero.

Table 3: Estimated annual nitrogen loads (tonnes) for Tasman Bay

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
600[a]	60[b]	304[d]	2300[a]	-1800[c], -2900[a]	0

[a]MacDiarmid et al. (2013), [b]Zeldis 2008a, [c]Christensen et al. 2003, [d]Gillespie et al. 2001.

* Net Ocean flux refers to dissolved inorganic nitrogen (DIN) plus dissolved organic nitrogen (DON).

2.3 International Site Description - Saldanha Bay, South Africa

Saldanha Bay, South Africa (Figure 8) was chosen as a comparison site to Golden Bay and Tasman Bay as it has a number of similar characteristics including a dominance of mussel aquaculture, similar nutrient loadings from the catchment/wastewater, and similar flushing times and oceanic exchange processes.

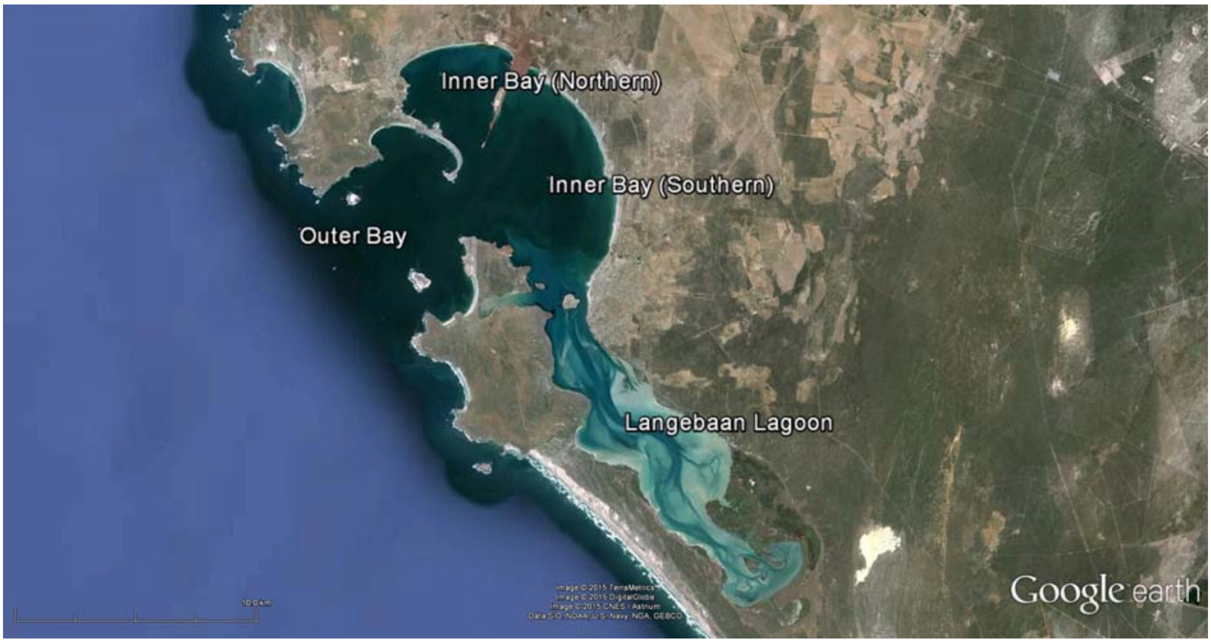

Figure 8: Saldanha Bay, South Africa.

Overview of the of the Hydrodynamics of Saldanha Bay

Saldanha Bay is one of the few semi sheltered marine systems along the South African coastline that is suitable for large scale shellfish aquaculture (Monteiro et al. 1998). Saldanha Bay has four distinct zones: an outer bay with a depth range of 25–50 meters with strong connections to shelf waters, two inner bays (separated by coastal structures) with depths varying between 5 and 20 meters and the Langebaan Lagoon to the south east of the embayment with a mean depth of around 2.5 m.

Monteiro et al. 1998 describe two physical mechanisms that drive the hydrodynamics and water exchange within the bay. The first of these is the movement of surface water from the highly productive coastal shelf environment into the bay (Brown & Henry 1985). Secondly, deeper water is brought into the bay through subsurface flows of cold denser water that cause stratification of a cyclic nature (Monteiro 1996). During the active phase of the stratification events warm and nutrient depleted surface waters are undercut by the subsurface inflow of nitrogen rich water from the Cape Columbine Southern Benguela upwelling system. This results in an influx of nutrients into the Bay and strong bay-wide thermal stratification with up to 8^{o} C difference in temperature between the upper ten meters of the water column and water at depth.

The flushing rate for the whole bay due to tidal forcing alone has been estimated to be 25 days and during the upwelling cycles flushing time can be reduced to about 12–16 days (Monteiro & Largier 1999).

Overview of the Nitrogen Load of Saldanha Bay

In Saldanha Bay there are two sea based Pacific Oyster farms and a 145 ha area designated for mussel farms with an estimated annual harvesting potential of 2000–3000 tonnes wet weight of shells (Monteiro 1996, Olivier et al. 2013).

In addition to the oceanic input of nutrients to the Bay other nutrient sources include seafood processing, wastewater discharges and storm water overflow (Monteiro et al. 1999, Anderson et al. 1999). The re-suspension of sediments during underwater blasting and dredging at an iron ore facility within the Bay will also contribute to the nutrient dynamics but the intermittent nature of this operation means that the contribution to the overall nitrogen budget is negligible (Table 4). Stenton-Dozey et al. (2001) carried out an analysis of nutrient dynamics under mussel farms highlighting the importance of organic enrichment and its role with regard to denitrification in the immediate vicinity of mussel rafts. They indicated that at times sediments were a net source of nutrients but no quantification of Bay wide denitrification could be found in the literature for Saldanha Bay. However the direct relationship between the presence of harmful algal blooms and high bay-wide levels of nutrients following upwelling events during summer (Probyn et al. 2001) indicate the importance of ocean flux in terms of the nitrogen budget of Saldanha Bay. It should be noted that the presence of harmful algal blooms has led to closures of mussel farms within Saldanha Bay in 2009, 2010 and 2011 (Olivier et al. 2013).

Based on an ecological carrying capacity approach, it has been estimated that Saldanha Bay could support up to 10–28 times the current levels of production (Olivier et al. 2013).

Table 4: Estimated annual nitrogen loads (tonnes) for Saldanha Bay from Olivier et al. (2013) and references therein.

Rivers	Groundwater	Wastewater	Ocean flux	Denitrification	Aquaculture
120	20?	650	2100 approx	?	?

2.4 New Zealand Site Description - Firth of Thames, New Zealand

Because of the strong linkages between the dynamics of the Firth of Thames and the Hauraki Gulf this section of the report discusses both of these areas. There is no clear definition of where the Gulf ends and the Firth begins, but the Hauraki Gulf Marine Park covers an area as far north as Mangawhai, south of Whangamata and extents 40–100 km offshore (Figure 9). The LINZ Chart for the Firth of Thames (NZ 533) extends just north of Coromandel Township and across to Waiheke Island where the mean depth is around 30 m.

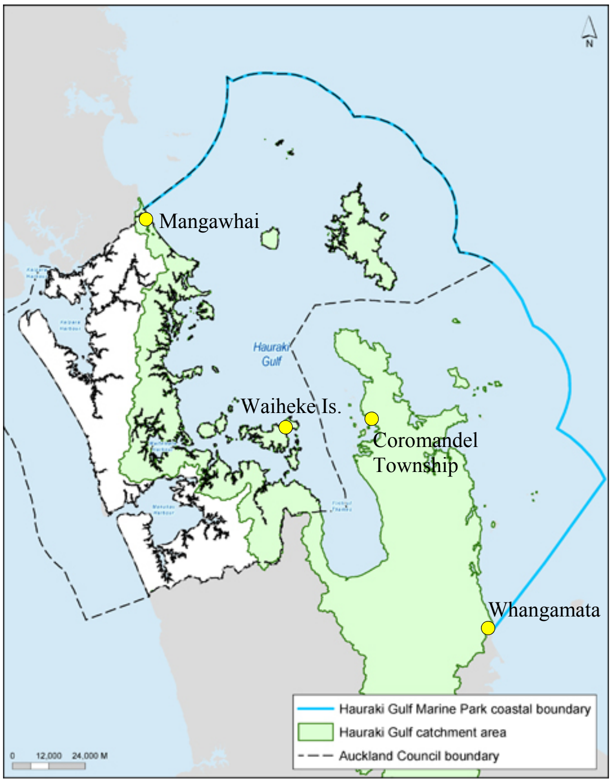

Figure 9: Extent of the Hauraki Gulf Marine Park. Auckland Council Unitary Plan.

Overview of Hydrodynamics of Firth of Thames/Greater Hauraki Gulf

The Firth of Thames, New Zealand (Figure 10) is classed as a meso-tidal estuarine system[1] with a catchment area of 3600 km^2 (Swales et al. 2008) and the two major river systems - the Waihou River and Piako River. MacDiarmid et al. (2013) indicate that the areas of the Hauraki Gulf and Firth of Thames are 2700 km^2 and 1100 km^2 respectively.The Firth of Thames is an important aquaculture production area with a number of existing and planned Greenshell mussel and Pacific oyster farms and the area has been designated for future finfish farming. In 2001, mussel production from Wilson's Bay and Matingarahi Point (Figure 10) was the second largest mussel growing area in the country with production of around 28 000 tonnes (Zeldis et al. 2010).

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[1] A coastal system where the tidal influence is strong, but not necessarily dominant with a tide range of less than 2.4 m.

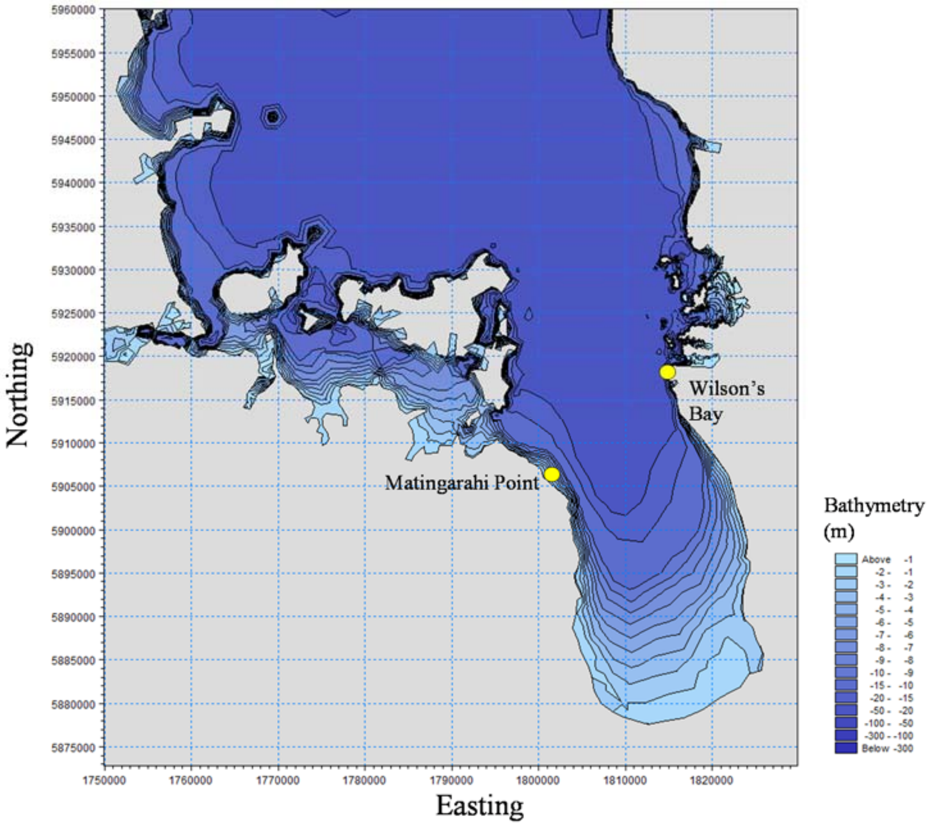

Figure 10: Depth ranges within the Firth of Thames (DHI NZ Regional Model). Depths relative to mean sea level, NZTM coordinates.

Much of the tidal flow within the Firth runs parallel to the coastline on entry and tends to hug the shoreline before heading out to the Hauraki Gulf through the centre of the bay (Black et al. 2000). The duration of the ebb and flood tides and the peak tidal currents are very similar so that residual currents driven by tides alone are weak (Procter & Greig 1989, Black et al. 2000). Freshwater inflows to the Firth are small compared to the tidal exchange between the Hauraki Gulf and Firth of Thames but overall the rivers contribute significantly to the nutrient budget of the Firth (MacDiarmid et al. 2013) and have localised effects in terms of hydrodynamics (Oldman et al. 2007). In addition to the dynamics of the oceanic exchange between the Firth and the Hauraki Gulf, the influence of winds is significant in terms of the overall circulation patterns within the Firth (Oldman et al. 2004). Persistent winds from the north/east result in clockwise residuals within the Firth while persistent winds from the south/west produced anti-clockwise residuals.

Several previous studies have identified the influence of the East Auckland Current on the Hauraki Gulf in terms of coastal upwelling (Bowman & Chiswell 1982; Greig & Procter 1988; Procter & Greig 1989). These upwelling events of continental shelf bottom waters (which are sometimes subtropical) results in an increase in nutrient levels in the Hauraki Gulf and can lead to subsequent algal and toxic algal blooms (Chang et al. 1995; Zeldis et al. 1995; Prasad et al. 1995, Zeldis et al. 2004). These intrusions also bring nutrient rich oceanic waters into the Firth of Thames (Sharples 1997; Zeldis et al. 1998). In spring and summer the Hauraki Gulf can become thermally stratified (Black et al. 2000) but salinity in the open waters of the Hauraki Gulf only ranges between 34.5 and 35.7 (Sharples 1997) as the influence of catchment derived runoff is minimal in the Hauraki Gulf.

The Firth of Thames has a residence time of approximately 12 days (Broekhuizen & Zeldis 2005), but the mean residence time of water flow within the Hauraki Gulf is around 56 days (Broekhuizen et al. 2002).

Overview of Nitrogen Loads to Firth of Thames/Greater Hauraki Gulf

The Firth of Thames/Hauraki Basin catchment is one of New Zealand's most intensively farmed areas and as such receives significant terrestrial nutrient inputs (Broekhuizen & Zeldis 2005; Zeldis 2008b). In addition, both water bodies (the Firth of Thames via the Hauraki Gulf) receive water and nutrients from the continental shelf during upwelling events (Broekhuizen et al. 2002; Zeldis et al. 2010). Other nutrient sources include waste water treatment discharges from residential and industrial sources (WRC 2011; Auckland Council 2014). Residential growth in areas surrounding the Firth of Thames (e.g. Hauraki Plains, Auckland and Waikato regions, and western Coromandel Peninsula) is significant (Giles et al. 2006).

The nitrogen budget for the Firth of Thames and the Greater Hauraki Gulf (see Table 5 and Table 6) was quantified during a previous review by Zeldis (2008b). This review broke down the nitrogen/ nutrient budget into dissolved organic nitrogen (DIN), particulate organic nitrogen (PON), dissolved organic nitrogen (DON), total Nitrogen from rivers[2] and oceanic flux across the boundary between the greater Hauraki Gulf and the South-East Gulf/Firth of Thames. This work also showed that overall the riverine supply of inorganic and organic Nitrogen to the Firth exceeds the contribution from the greater Hauraki Gulf (driven mainly by mixing across the boundary). When ocean upwelling is dominant the rivers contribute about half of the dissolved inorganic N load, but when down welling is dominant around 70% of the load comes from the rivers.

Primary production in the Firth of Thames is nitrogen limited over most of the year (Broekhuizen et al. 2002; Zeldis et al. 2004, Zeldis et al. 2010), and it is most likely the denitrification sink which accounts for this. Zeldis (2008b) estimated the denitrification rate to be 8000 tonnes of N per year. At present there are no additional loadings of nitrogen into the embayment due to aquaculture as finfish production has not been commercialised but data from Zeldis (2008b) provided an estimated loss of nitrogen from mussel harvesting of 126 tonnes per annum.

Table 5: Estimated annual nitrogen loads (tonnes) for the Hauraki Gulf.

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
950[a]	?	578[a]	-200[a]	-700[a]	-126[b]

[a]Zeldis et al. (2006), [b]Zeldis et al. (2008b)
* Net ocean flux refers to dissolved inorganic nitrogen (DIN) plus dissolved organic nitrogen (DON).

Table 6: Estimated annual nitrogen loads (tonnes) for the Firth of Thames.

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
4600	?	262	3500	-8100	-126[b]

[a]Zeldis et al. (2006), [b]Zeldis et al. (2008b).
* Net ocean flux refers to dissolved inorganic nitrogen (DIN) plus dissolved organic nitrogen (DON).

[2] The particulate and dissolved riverine organic material was not separately quantified.

2.5 International Site Description - Faroe Islands

The major similarity between the Faroe Islands and the Firth of Thames is that they are both influenced by offshore upwelling events bringing nutrient rich waters into the aquaculture areas (Cartensen et al. 2003 and Á Norði 2010). The Faroe Islands fjord systems are similar in character to both the Marlborough Sounds and Fiordland with similar residence times and summertime water temperatures. Other sites globally similar to the Firth of Thames/Hauraki Gulf either have no aquaculture production or have much higher water temperatures.

In 2009, the Faroe Islands was the fifth largest producer of farmed salmon in the world producing approximately 50 000 tons of salmon per year (Á Norði 2010).

Overview of the Hydrodynamics of the Faroe Islands

The majority of the Faroese aquaculture areas are inside long (3 – 14 km) and narrow (about 1 km wide) coastal fjords. Aquaculture areas are at present sited in maximum depths of between 30 and 60 m, although there is a growing trend to move farms into deeper, more exposed environments (Hansen 2000; Simonsen & Gislason 2002) similar in terms of processes (i.e. nutrient upwelling, currents and waves) to the greater Hauraki Gulf.

Current speeds vary greatly in the Faroe fjords with maximum tidal currents ranging from 0.10–0.75 m/s (Hansen 2000, Larsen et al. 2008). Strong wind events (i.e. daily mean wind speeds of greater than 15 m/s) can occur throughout the year but are more frequent during winter. The presence of these strong persistent winds have a strong influence on the currents and mixing within Faroese fjords. Often there is no clear pycnocline in the fjords, but rather a dynamic transition zone between the upper layer (salinity: 32.1–34.9 PSU and temperature: 5.8–11.2 °C) and the deeper oceanic water (salinity: 34.9–35.3 PSU and temperature: 6.2–10.8 °C) which is entrained from outside the fjords (Á Norði 2010).

Offshore the water is well mixed by a persistent tidal current, which surrounds the shelf at about the 100–130 m bottom depth contour (Gaard & Signar. 1998; Larsen et al. 2002). In addition, residual currents have a persistent clockwise circulation around the islands with strengths ranging from 0.02–0.16 m/s (Simonsen 1999; Gaard & Hansen 2000). These circulation patterns lead to a relatively uniform shelf ecosystem on the Faroe Shelf.

Flushing time varies significantly from fjord to fjord and ranges from a few days to weeks (Á Norði 2010, Gaard et al. 2011). Exchange between the Faroe Shelf waters and the deeper oceanic waters ranges from a typical value of 75 days to as low as 8 days (Larsen et al. 2008).

Overview of Nitrogen Loads to Faroe Fjords.

Generally there is little or no human induced nutrient input to the Faroe Islands Fjords, due to the very low population density (Mortensen 1990). Á Norði's study of a "typical" Faroe Island Fjord found that on average anthropogenic nutrients accounted for 14% of the total nutrient input and that a significant amount (over 50%) of this input was from aquaculture. The majority of the inflow of nutrients came from offshore transport/upwelling into the fjords (Table 7). Exact numbers in tons per year were not quantified, but several studies have looked at the average net uptake of new nutrients and its relationship with the uptake as a percentage of the primary production. Cartensen et al. (2003) and Á Norði (2010) found that the average uptake of new nutrients in the euphotic zone accounted for 70–75% of the nitrogen demand of the primary production. This is higher than similar observations made at Scottish and Norwegian fjords where anthropogenic inputs of nutrients are also higher at 25–60% of the total primary production (Wassmann 1986, Wassmann 1990; Rees et al. 1995).

Table 7: Typical nutrient loading for Faroe Islands Fjord (Á Norði 2010).

Rivers	Groundwater	Wastewater	Ocean flux	Denitrification	Aquaculture
Negligible	?	?	10000+	?	4000

2.6 New Zealand Site Description - Big Glory Bay, Stewart Island

Overview of Hydrodynamics of Big Glory Bay

Big Glory Bay (Figure 11) is an embayment within Stewart Island, South of the South Island of New Zealand. Its catchment area is around 15 km^2. The bay is connected to Paterson Inlet through a channel about 25–28 meters deep in the centre. To the north-east of Paterson Inlet is Foveaux Strait. Within Big Glory Bay the mean depth is less than 20 meters (Kerroux et al. 2011).

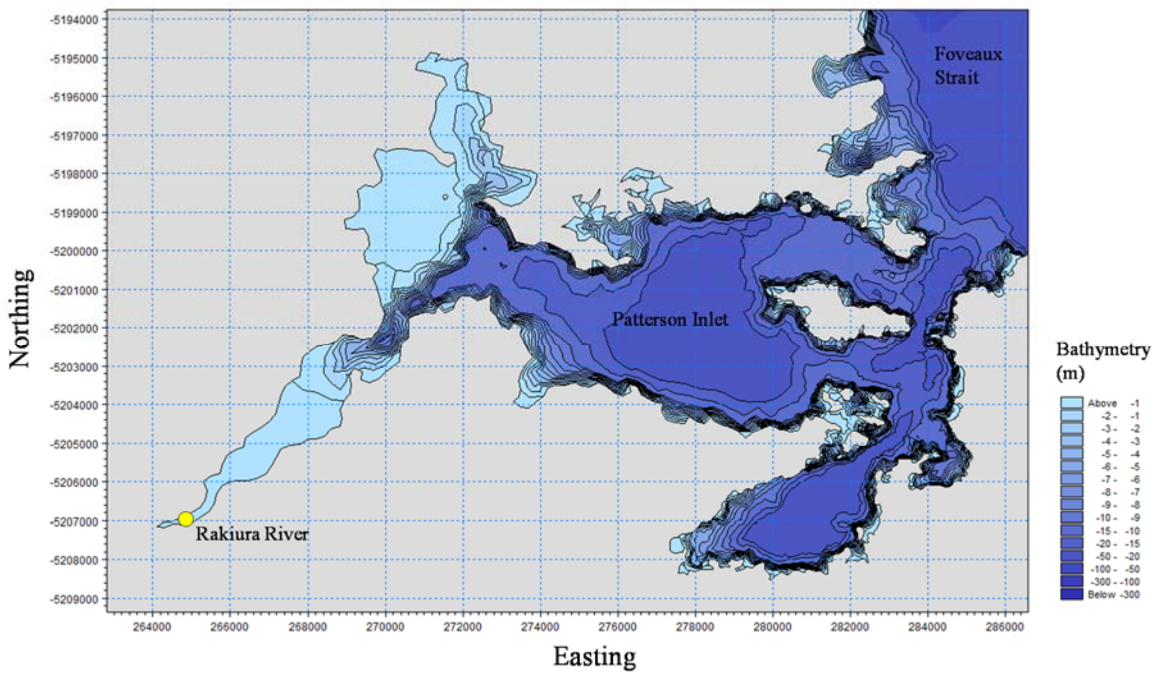

Figure 11: Big Glory Bay Bathymetry, New Zealand.

Rakiura River provides freshwater runoff to Big Glory Bay with an average flow of 1 m^3/s. There is no reported stratification within the bay (Stenton-Dozey & Brown 2010). Wind plays a key role in the general circulation patterns and flushing of Big Glory Bay. During westerly winds the surface waters move quickly out of the bay and the return flow of oceanic waters at depth move into the bay (Morrisey et al. 2000; Kerroux et al. 2011). Key (2001) also describes the process of water entering the bay from deep within Patterson Inlet and also counter flows along the sides of the bay. Observations in summer indicate that during periods of little or no wind, current flows are at a minimum resulting in reductions in dissolved oxygen and increases in temperatures in surface waters (Hartstein pers. comm.).

According to the modelling work undertaken by Pridmore & Rutherford in 1990, flushing time of the bay was estimated at 10–14 days during periods of light wind and decreasing to 5 days when there were strong westerly winds. The influence of the freshwater input to the Bay is negligible (Key 2001).

Overview of Nitrogen Loads to Big Glory Bay

Within Big Glory Bay and the surrounding coastal environment (i.e. parts of Foveaux Strait) nitrogen is a limiting nutrient with regard to primary production (Edwards 1988; Pridmore & Rutherford 1990; Rutherford et al. 1988; O'Callaghan 1998; Key 2001). The compiled dataset on estuarine environments such as Big Glory Bay indicate that nitrogen levels can be as low as a few µg-N/L during summer periods (and sometimes spring), thereby limiting primary productivity.

Higher available nitrogen concentrations have been observed in Big Glory Bay compared to Paterson Inlet (O'Callaghan (1998) and unpublished data from Pridmore 1995) with values of up to 200 µg/L in Big Glory Bay and up to 150 µg/L in Paterson Inlet. The difference between the water bodies was estimated at around 30% and attributed to increased nutrient loading created by finfish farming (Pridmore 1995).

Key (2001) observed similar trends in nitrogen concentrations between July 1999 to 2000 at sites in Big Glory Bay, Paterson Inlet and Foveaux Strait with high spikes in data attributed to the offshore upwelling of nutrients. Observed data offshore of the western coast of the South Island indicate that nitrate enriched waters in excess of 200 µg/L can occur (Bradford et al. 1991).

A similar exercise was undertaken with the data obtained from the mussel monitoring programme in Big Glory Bay (Stenton-Dozey & Brown 2010). The dataset covers only the summer period, but shows that nitrogen levels range from 10 to 100 µg-N/L and that available nitrogen may remain in the water column during the period of highest primary productivity, the high temperature peaks of summer.

While a number of studies have measured nutrient levels within Big Glory Bay, few have directly quantified the nitrogen budget (Table 8). In this instance nitrogen loading was estimated from knowledge of the existing farm tonnage, catchment load and exchange period/flushing rate based on Rutherford et al. (1988) and Kerroux et al. (2011). The ocean flux rates calculated by Rutherford et al. (1988) vary greatly depending on the exchange rate, which varies depending on the forcing conditions in Foveaux Strait. This is particularly important in regard to nutrient upwelling which appears to promote the development of algae blooms (as with other sites presented in this report) and subsequent fish kills that have occurred within the Bay (Rutherford et al 1988; Pridmore 1995; Key 2001; Kerroux et al. 2011).

Table 8: Estimated annual Nitrogen loads (tonnes) for Big Glory Bay.

Rivers	Groundwater	Wastewater	Ocean flux	Denitrification	Aquaculture
7[a]	?	?	150–850[b]	?	240[b]

[a]Rutherford et al. (1988) and [b]Kerroux et al. (2011) after Rutherford et al. (1988).

2.7 International Site Description - Macquarie Harbour, Tasmania, Australia

There are several similarities between Macquarie Harbour and Big Glory Bay. Both embayments have little or no anthropogenic nitrogen input (other than aquaculture), they are both semi enclosed embayments (although the entrance to Macquarie Harbour is smaller and shallower than for Big Glory Bay). In addition, both sites have well established salmon farms where carrying capacity has been quantified through a combination of observations and numerical modelling (e.g. Hartstein et al. 2010; 2011).

Overview of Hydrodynamics of Macquarie Harbour

Macquarie Harbour is located within Tasmania's World Heritage West Coast (Figure 12). Two large rivers, the Gordon and the King, drain into the harbour. Their average mean flows are 365 m^3/s and 55 m^3/s respectively. The catchment area is predominantly within the World Heritage site and therefore considered to be pristine. Macquarie Harbour is a semi filled-in river valley, with some of the geomorphological features of a fjord (Hartstein et al. 2011). It has a shallow restricted entrance which opens into a long deep basin with depths up to 50 m and steep sloping walls.

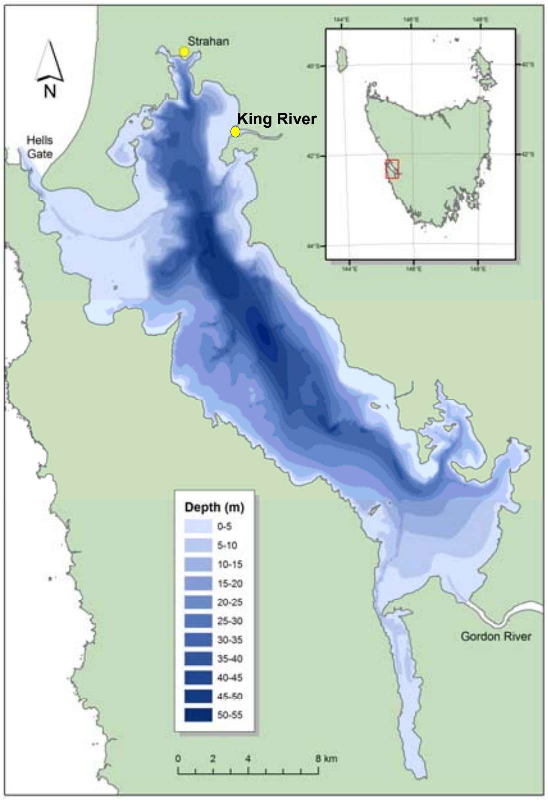

Figure 12: Macquarie Harbour bathymetry and geomorphology.

The combination of significant freshwater inflows and the entrance sill results in a surface layer of brackish water flowing over a volume of deeper and denser waters. Vertical salinity profiles indicate various degrees of mixing (Hartstein et al. 2011) due to either density gradients or wind generated waves (due to a sizeable fetch of more than 30 km).

A simple salt balance calculation Koehnken (2001) estimated the flushing time for the mixed layer of the harbour to be 80 days (assuming a 50% mixed salinity layer of 15 m). Hartstein et al. (2011) applied a similar methodology and derived an annual average flushing time of 47 days. In reality the residence time can vary significantly according to wind conditions, which increase the mixing between fresh and salt water (Koehnken 2001). The residence for the whole embayment (not just the mixed layer) is potentially much higher. For example, Hartstein et al. (2011) suggest that the minimum residence time for the deep bottom waters of the harbour may be as long as 120–150 days.

Overview of Nitrogen Loads to Macquarie Harbour

Macquarie Harbour has an estimated carrying capacity of 28 000 tonnes of finfish (trout and salmon aquaculture, Hartstein et al. 2011). Currently, the aquaculture production level within the harbour is estimated at 16 000 tonnes (Hartstein per.comm. 2013).

Nitrogen loading into the harbour is summarised in Table 9. Significant inputs occur in the form of dissolved nitrogen from the Gordon River and the aquaculture farms themselves. The Strahan Township supplies less than 0.25% of the loadings to the harbour (Hartstein et al. 2012b).

Table 9: Summary of Nitrogen loading for Macquarie Harbour (Hartstein et al. 2012b).

Rivers	Groundwater	Wastewater	Ocean flux	Denitrification	Aquaculture Existing	Aquaculture Maximum Consented
1150	?	2	0	?	1160	2480

2.8 New Zealand Site Description - Marlborough Sounds

Overview of Hydrodynamics of Marlborough Sounds

The Marlborough Sounds consists of numerous sheltered inlets and bays, broad reaches and has close proximity to the deep waters of nearby Cook Strait (Figure 13). In total the Marlborough Sounds covers an area of over 150 000 ha with an average depth of 40 m. Its combination of location and geomorphology results in a diversity of physical and water column conditions. For the purpose of this study we divide the Marlborough Sounds into the Queen Charlotte and Pelorus Sound Regions.

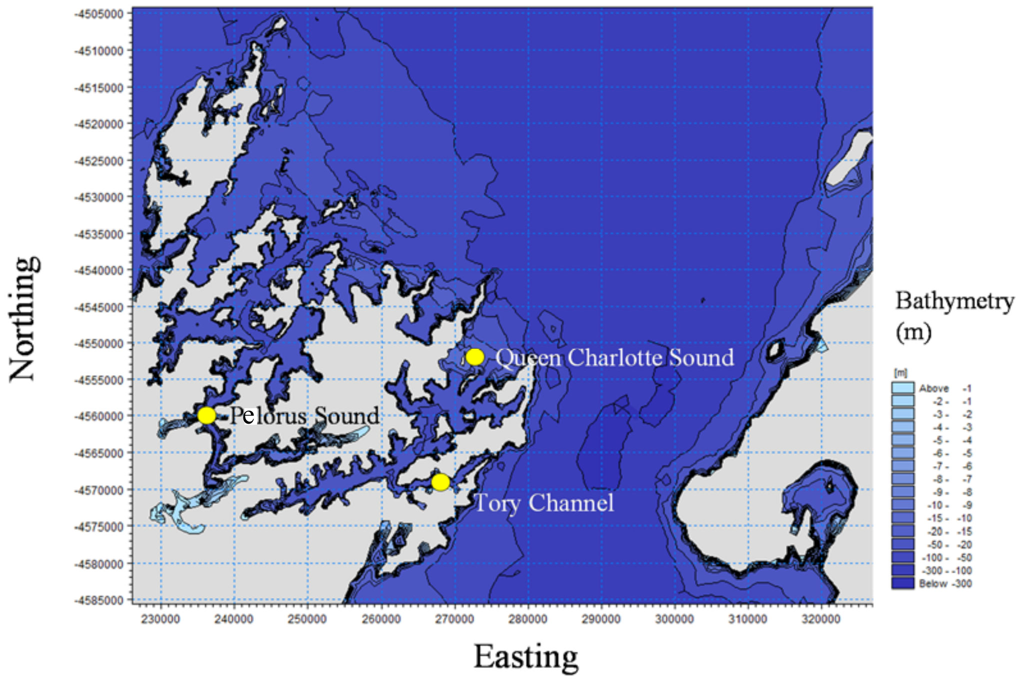

Figure 13: Outer and middle Pelorus Sound bathymetry.

Riverine input into the outer Pelorus Sound is relatively small compared to the large volume of water exchanged with Cook Strait through tidal and wind driven processes (Gibbs et al. 1992; Dupra 2000; Hartstein et al. 2012a). This exchange results in a relatively short residence time of approximately 2–3 weeks (Heath 1976b; Zeldis et al. 2008). Exchange in the inner Pelorus Sound is thought to be somewhat longer and more variable. In the inner sounds, water from the Pelorus River tends to form a surface layer of lower saline water overlying nutrient enriched bottom water (originating from Cook Strait).

Calculating the flushing time in Queen Charlotte Sound is complicated by the fact that it has two entrances with differing tidal phases. Gillespie et al. (2011b) suggest that the residence time of Queen Charlotte Sound is of the order of 30–40 days. The hydrodynamics of Queen Charlotte Sound also differ spatially with the inner portion of the sound having relatively low currents (resulting in a strong depositional environment) while the outer sounds (towards Tory Channel) are highly dynamic with flows of up to 1 m/s Gillespie et al. (2011b).

Overview of Nitrogen Loads to Marlborough Sounds

The Marlborough Sounds supports the three major aquaculture species - Green-lipped Mussels, King Salmon and Pacific Oysters. Of the over 500 functioning marine farms within the Marlborough Sounds the majority support green-lipped mussel production with a small proportion of mussel spat catching and holding sites and a number of King Salmon sites. In total nearly 2% of the total area of the Marlborough Sounds is allocated to the aquaculture industry.

The catchment area of Marlborough Sounds is sparsely populated and so the nitrogen loads from land are low compared to the potential inputs of nitrogen from oceanic exchange. However cases of harmful algal blooms have been documented (MacKenzie & Berkett 1997, Rhodes et al. 2001) associated with nitrogen imports from outside of the Sounds.

Data in Table 10 and Table 11 show the nitrogen budget for both Queen Charlotte and Pelorus Sounds and are collated from a number of reports (Gowen & Bradbury 1987, Gibbs et al. 1992) as summarised by Gillespie et al. (2011b).

Table 10: Summary of Nitrogen loading for Queen Charlotte Sound from Gillespie et al. (2001) and references therein.

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
867	0	9	350	-367[a]	868 (finfish) -12 (mussel farm)

Table 11: Summary of Nitrogen loading for Pelorus Sound from Gillespie et al. (2001) and references therein.

Rivers	Groundwater	Wastewater	Ocean flux*	Denitrification	Aquaculture
580	0	0	200–4200*	-465	336 (finfish) -266 (mussel farms)

* Net ocean flux refers to Dissolved Inorganic Nitrogen (DIN).

2.9 Site Description - Huon River and D'Entrecasteaux Channel, Tasmania, Australia

The Huon River is the fourth largest river in Australia and drains into the D'Entrecasteaux Channel and the Tasman Sea (Figure 14). The Huon River system is similar geo-morphologically and hydrodynamically to the Marlborough Sounds in that it has a series of complex embayments opening to the ocean. The Huon River system is one of the largest aquaculture producing area in Australia and like the Marlborough Sounds is subject to offshore upwelling events that have led to harmful algae blooms (Mackenzie & Berkett 1997; Clementson et al. 2004).

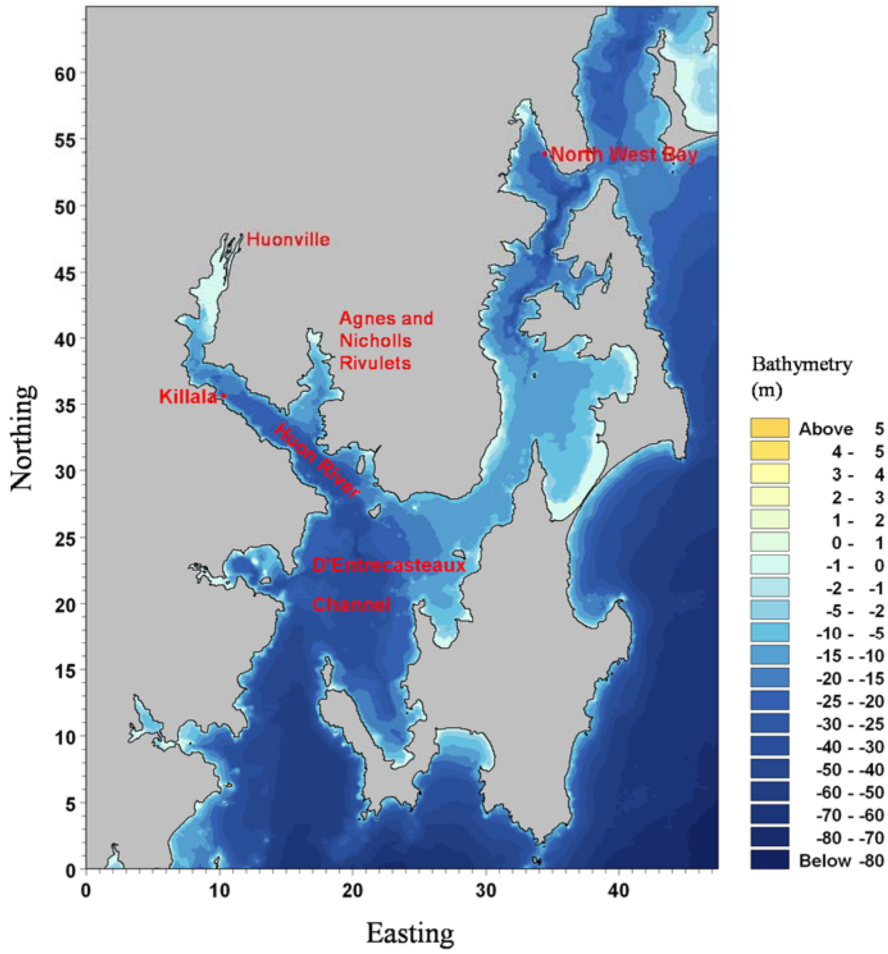

Figure 14: Bathymetry and morphology of the Huon Estuary.

Overview of Hydrodynamics of Huon River and D'Entrecasteaux Channel

The discharge of the Huon River results in a two layer circulation pattern within much of the Huon Estuary (Herzfeld et al. 2005). A shallow brackish surface water layer moves seawards and seawater from the southern end of the D'Entrecasteauz Channel moves upstream forming a salt wedge that migrates upriver depending on wind conditions and the volume of river flow (Figure 15). During low freshwater flows, the salt wedge extends as far inland as Huonville but during high freshwater flows the wedge only migrates as far as the mouth of the Huon River. At the mouth of the Huon River the surface water currents bifurcate with some of the water mass heading east towards Northwest bay while the remainder moves southward of the D'Entrecasteaux Channel (Butler et al. 2001; Herzfeld et al. 2005). The freshwater input from Agnes and Nicholls Rivulets is very small compared to the Huon River input and tidal processes that dominate this area of the system.

The flushing time for the D'Entrecasteaux Channel is estimated to be 20–26 days (Herzfeld et al. 2010) while the Huon River estuary exchanges much more rapidly (several hours to several days depending on freshwater flows). Flushing times for the main D’Entrecasteaux Channel have been estimated to range from 7.5 days during February to 8.8 days during October (Herzfeld et al. 2005), while estimates for North West Bay specifically have ranged from 5 to 7 days (Jordan et al. 2002, Herzfeld et al. 2005). A flushing estimate for the combined D’Entrecasteaux Channel/Huon Estuary waterway, based on the average time for neutrally buoyant particles to exit the system, was computed as approximately 26 days.

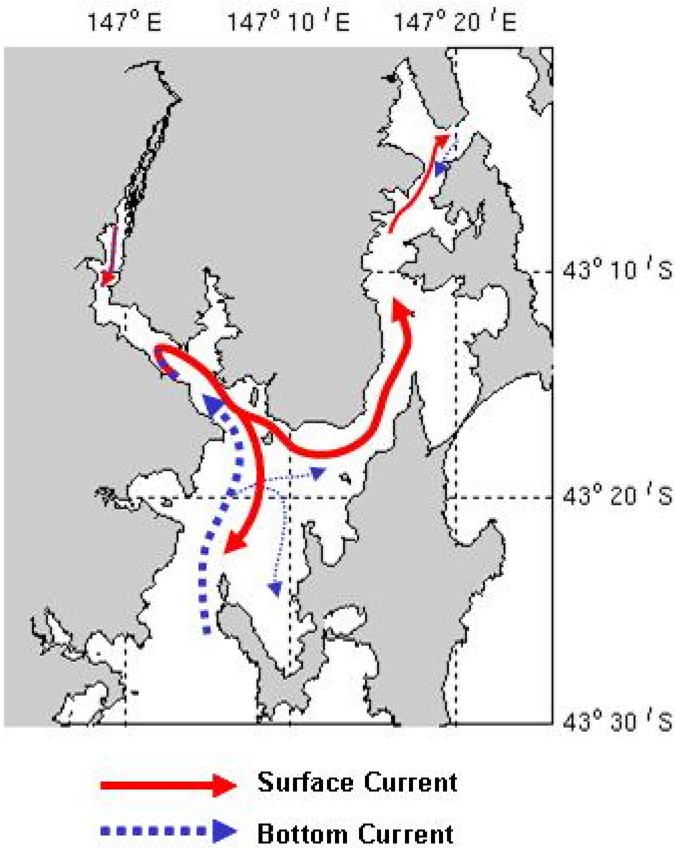

Figure 15: **Movements of surface and bottom currents (from Herzfeld et al. 2005, reproduced with permission).**

Overview of Nitrogen Loads to Huon River and D'Entrecasteaux Channel

Within the Huon Estuary there are a number of marine farms licensed for finfish, mussels and oysters (Figure 16) with the majority of production from salmon (over 20 000 tonnes/year, Marine Farming Branch, Department of Primary Industries, Parks, Water and Environment).

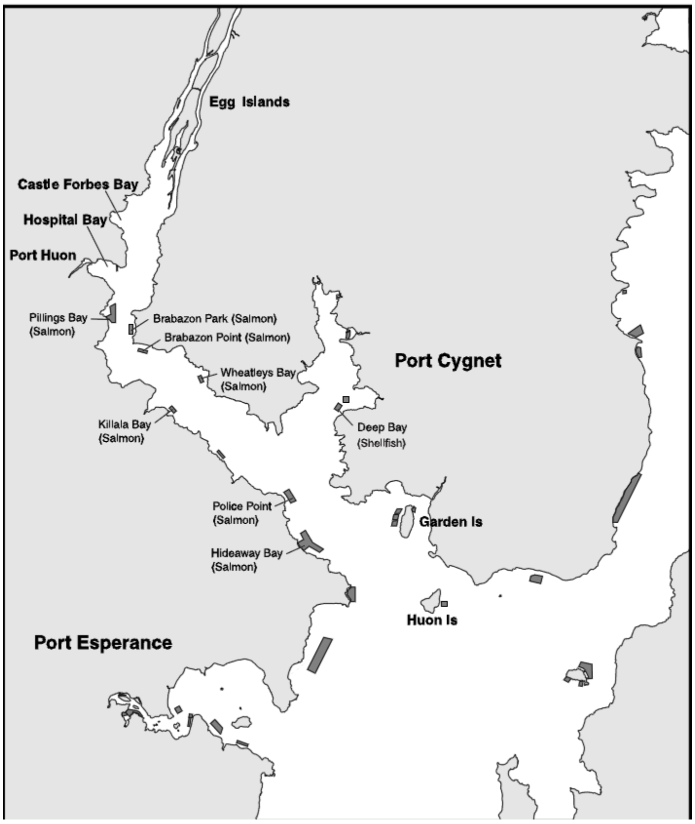

Figure 16: Aquaculture marine farms within the Huon Estuary in 1999 (CSIRO Huon Estuary Study Team. 2000, reproduced with permission).

The present population for the region is about 13,000 and includes the towns of Huonville, Cygnet, Geevestone and Franklin and overall population density is low. Commercial activities carried out within the region that could contribute to the nutrient dynamics of the Huon are horticulture, grazing and dairying. Fruit processing could also provide some influx of nutrients (CSIRO Huon Estuary Study Team. 2000). A nutrient budget for the channel has been estimated using data from ANRA (2001), for the existing farm loading (Table 12).

Harmful algal such as the dinoflagellate *Gymnodinium catenatum* are known to bloom within the Huon Estuary and have caused a number of stock losses (Clementson et al. 2004). Several more recent algae blooms have occurred in the last 18 months causing large numbers of fatalities at several farms (Hartstein pers. obs. 2012). There has been recent debate on whether the nutrient events triggering the recent algae blooms have come down the Huon River or were caused by offshore upwelling. This is similar to issues found in New Zealand where several blooms have been observed in Big Glory Bay, Marlborough Sounds and Firth of Thames. In each case these blooms were linked to nutrient upwelling rather than land or aquaculture related nutrient releases.

Table 12: Estimated annual Nitrogen loads (tonnes) for D'Entrecasteaux Channel and Huon River Estuary.

Rivers	Groundwater	Wastewater	Ocean flux	Denitrification	Aquaculture
600[a]	?	40–50[b]	1000+[d]	?	2100[c]

[a]ANRA 2001, [b]Parsons (2012), [c]DPIPWE pers.comm. 2012, [d]Butler et al. (2000).

2.10 Summary

The review of overseas and New Zealand aquaculture sites has shown that each embayment is unique and there is no exact match between the physical settings, level of aquaculture development and physical processes within the selected New Zealand sites and the chosen international sites – especially when considering only temperate settings.

To properly understand the dynamics of nitrogen within marine systems it is important to quantify the partitioning that occurs between dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON) and particulate organic nitrogen (PON). Quantification of the nitrogen load from riverine, oceanic and groundwater sources, wastewater treatment plant and stormwater loads and aquaculture operations should be carried out. Similarly the role of nitrogen sinks such as oceanic fluxes, aquaculture operations and sediment processes (burial and denitrification) should be quantified. With a good understanding of the processes driving the nitrogen budget of a marine system and its natural variability the potential impacts of an aquaculture development can then be understood.

The role of upwelling of nutrients is important in terms of nitrogen dynamics and the development of harmful algae blooms and has been well documented (e.g. Pridmore & Rutherford 1990; Mackenzie & Berkett 1997; Key 2001, Broekhuizen & Zeldis 2005; Stenton-Dozey & Brown 2010; Zeldis et al. 2010). This review has highlighted the importance of understanding the relative role of nutrient inputs from aquaculture and coastal upwelling (e.g. Huon River and D'Entrecasteaux Channel and Big Glory Bay).

The case studies presented show that the scale of development of aquaculture in New Zealand is generally much larger than those occurring overseas. For example, Macquarie Harbour has around 500 ha of salmon production and Saldanha Bay has around 200 ha of bivalve production. By comparison, Tasman Bay and the Firth of Thames have just over 4000 and 1000 ha respectively of existing mussel farms. In terms of the effects of nitrogen on the marine environment the scale, staging and type of development needs to be considered. The overseas examples presented provide valuable information on the planning process used for new aquaculture developments and the monitoring framework put in place which can be applied to New Zealand’s aquaculture industry.

The sites considered cover a range of conditions in terms of the relative roles of anthropogenic sources of nitrogen, aquaculture related nitrogen and the influence of oceanic derived nitrogen. For example, for both Big Glory Bay and the Faroe Islands there is negligible input of nitrogen from the land and both sites are highly influenced by upwelling of oceanic waters. Within Macquarie Harbour the contribution of aquaculture to the nitrogen budget is very similar to that from riverine sources and there is little exchange between the harbour and offshore waters. Data from the Firth of Thames indicates the importance of considering the seasonal variability of nitrogen sources while information presented for the Marlborough Sounds, Saldanha Bay and the Huon River system shows the importance of understanding the different dynamics within an individual system. Finally the data presented for the Tasman and Golden Bay nitrogen budget shows the importance of quantifying the oceanic and denitrification fluxes.

Data presented in this section along with the review of international regulations (Section 4) and management strategies (Section 5) relating to nitrogen loads will provide guidance in terms of determining the potential ecological effects of aquaculture derived nitrogen in the context of the New Zealand’s marine environment.

3. Nitrogen Related Adverse Effects

3.1 Introduction

Negative environmental impacts that can occur as a result of elevated nitrogen levels can generally be divided into two major categories - toxicity and eutrophication (e.g. US Environmental Protection Agency 2001, Cloern 1996, 2001). In this section of the report we present an overview of the impact of elevated nitrogen levels of nutrients (and related water column properties) in terms of the toxicity to marine fauna. Additionally an overview of water column and seabed effects of elevated nitrogen levels is presented.

Toxicity effects of elevated nitrogen levels are generally only near-field effects that occur in the immediate vicinity of fish cages or mussel farm long lines 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, a toxicity criterion of 96h LC50 refers to a concentration of a substance that is lethal to 50% of a population after 96 hours of exposure (Lewis & Morris 1986; Spotte & Adams 1983; USEPA 1986, 2009b). This is an instance of "acute" toxicity levels where pronounced effects occur over relatively short time frames (in this case death of 50% of a population). "Chronic" toxicity levels are also important because they indicate the impacts of exposure over longer time frames (i.e. weeks to months), not necessarily immediately apparent in a population and are generally set at much lower thresholds than acute toxicity values. Chronic toxicity can lead to a general decline in the health of an organism, higher feed conversion ratios and ultimately increased levels of mortality in a population. Elevated concentrations that only occur for only very short durations may or may not cause harm to the particular organisms. The actual processes involved in the toxicity relate to species assimilation and metabolism rates.

The need to consider both the time and intensity of exposure and different levels of toxicity means that toxicity thresholds are generally set as guidelines only and the generalised application of literature values is difficult.

In terms of the toxic effects of nitrogen on marine fauna this report considers the following parameters

- Enhanced Chlorophyll-a production due to excess nutrients;
- Dissolved oxygen depletion due to breakdown of nutrients;
- Nitrates;
- Nitrites; and
- Ammonia.

Eutrophication is the response of the aquatic ecosystem to the excess addition of nutrients. Gray (1992) defines eutrophication as "when nutrients are added to a body of sea water they lead, provided that there are no toxic compounds present and provided that there is sufficient light, to increases in autotrophic and heterotrophic growth". For example, the release of nutrients in quantities that enable primary producers to overcome the limitations that are typically imposed in healthy unaltered ecosystems (e.g. Boynton et al. 1982 and Cloern 1996) can lead to excessive growth. Indeed nitrogen has been identified by the scientific community as the limiting nutrient in most coastal marine environments (Ryther & Dunstan 1971, Howarth et al. 1988; Broekhuizen & Zeldis 2005). As increased amounts of nutrients are added, grazers may not be able to keep up with the increased autotrophic growth rates leading to organic material settling to the seabed. During sinking the organic material is degraded (consuming oxygen) which can result in severe reductions in oxygen concentrations near the sea bed.

Eutrophication can occur across a variety of marine environments (Nixon 1995, Cloern 2001) and is most severe in areas impacted by high run-off from land or high nutrient exchange with oceanic systems. Phytoplankton blooms can occur when the high nutrient concentrations coincide with optimal conditions for growth (i.e. salinity, temperature and water clarity). These blooms can have a large extent and be so dense as to greatly reduce light penetration in the water column (Justic 1987 and 1988). The blooms can be of species which produce toxins potentially lethal to marine fauna. Following the decay of algal blooms, large quantities of organic matter are deposited on the seabed. The degradation of organic matter consumes oxygen and can occur in combination with stratification. This can lead to anoxic conditions in the sediment and subsequently a simplification in benthic community structure (e.g. Pearson & Rosenberg 1978) or in extreme cases the development of dead-zones which cannot support any form of marine life (e.g. Rabalais et al. 2010).

3.2 Toxicology impacts relating to Nitrogen

Chlorophyll-a

Chlorophyll-a is an indicator of phytoplankton in the water and often used as indicator of eutrophication. A range of Chlorophyll-a values are presented in Table 13 that reflect a range of trophic states, saline regimes, phytoplankton populations and ecological conditions from the US EPA (2003). The direct link between excess chlorophyll-a (associated with algae growth) and lethal conditions for fauna is difficult to assess. Furthermore, mortality is usually caused by shading (reducing light in the water column), depressed oxygen concentrations beneath the algae and the production of toxins that can be highly dangerous for fish and other fauna.

There are no New Zealand-specific ANZECC (2000) guideline trigger values for chlorophyll-a in New Zealand marine and estuarine waters. The ANZECC (2000) guidelines suggest that New Zealand uses the south-east Australia trigger values. However these trigger values are based on low-nutrient waters and may not be suitable for New Zealand estuary systems (ECAN 2009).

Table 13: Summary of Chlorophyll-a concentrations reflecting trophic-based water quality, phytoplankton community and ecological conditions. (US EPA 2003).

	Chlorophyll-a Concentrations (µg/L)					
	Oligotrophic Conditions (Low Primary Production)		**Mesotrophic Conditions** (Intermediate Primary Production)		**Eutrophic Conditions** (High Primary Production)	
Salinity Regime	**Average or General Range**	**Peak Range**	**Average or General Range**	**Peak Range**	**Average or General Range**	**Peak Range**
Spring						
Tidal-Fresh (< 0.5 PSU)	0.8–3.4[a] 0.3–3.0[b] < 3.5[c] < 4[e] 1.1[f]	2.6–7.6[a]	3.0–7.4[a] 2–15[b] 3.5–9[c] 4–10[e] 4.3[g] 15[g]	8.9–29[a] 13.5[g]	6.7–31[a] 10–500[b] 9–25[c] >10[e] 6.7[g]	16.9–107[a] 42.9[g] <33[g]
Oligohaline (0.5 – 5.0 PSU)	2.3[f]		9.6[g] 15[i]	24.3[g]	5.0[g]	29.8[g] <33[g]
Mesohaline (5.0 – 18.0 PSU)	3.7[f]		5.6[g] 5[g]	24.6[g]	11.1[g]	44.9[g] <25-30[g]
Polyhaline (18–30 PSU)	<1[c] <2[d] 3.9[f]		1–3[c] 2–7[d] 2.9[g] 5[g]	6.7[g]	3-5[c] >7[d] 9.1[g]	18.0[g] <25–30[g]
Summer						
Tidal-Fresh (< 0.5 PSU)	0.8–3.4[a] 0.3–3[b] <4[e] 1.1[f]	2.6–7.6[a]	3.0–7.4[a] 2–15[b] 3.5–9[c] 4–10[e] 8.6[g]	8.9–29[a] 15.9[g] 15[g]	6.7–31[a] 10–500[b] >10[e] 25.3[g]	16.9–107[a] 62.1[g] 33[g]
Oligohaline (0.5 – 5.0 PSU)	2.0[f]		6.0[g]	25.2[g] 15[g]	17.1[g]	60.5[g] 33[g]
Mesohaline (5.0 – 18.0 PSU)	4.4[f]		7.1[g] 5[g]	14[g]	12.2[g]	52.5[g] <25-30[g]
Polyhaline (18–30 PSU)	<1[c] <2[d]		1–3[c] 2–7[d] 4.4[g] 5[g]	8.7[g]	3-5[c] >7[d] 6.1[g]	25.8[g] <25-30[g]

[a]Ryding & Rast 1989; [b]Wetzel 2001; [c]Smith 1998; [d]Molvaer et al. 1997; [e]Novotny & Olem 1994; [f]Olson 2002; [g]US EPA (2003).

Oxygen

Dissolved oxygen (DO) levels are generally a good indicator of a healthy environment (US EPA 1976) and as such dissolved oxygen levels close to saturation are desirable. Vaquer-Sunyer & Duarte (2008) concluded that 5 mg/L of dissolved oxygen is sufficient for the protection of 90% of all marine species. For aquaculture species, literature values for LC50 (below which lethal effects occur) range between 1.2 mg/L (24h LC50) for Coho Salmon (Davison et al. 1959) to 3.1 mg/L (7 days LC50, maximum of sample batch) for juvenile rainbow trout (Merkens & Downing 1957). Marine invertebrates appear more resistant to reduced oxygen levels, and the equivalent LC50 dissolved oxygen concentrations are usually as low as 0.5–1.0 mg/L. As such, setting dissolved oxygen levels to protect fish or bivalves within farms is likely to ensure the survival of other organisms (Vaquer-Sunyer and Duarte 2008).

However, DO levels are not only important in terms of fish mortality. Reductions in dissolved oxygen below optimum levels can have a negative effect on fish growth, commercial value and additional stress. The US EPA (1986) carried out an assessment of the tolerance to low oxygen levels on a variety of non-commercial, cultivated fish species. The conclusion of this study, (summarised in Figure 17) is that DO levels decreasing below 7 mg/L cause increasingly larger reductions in optimum fish growth with between 5 to 10% growth reduction at 6 mg/L, around 15% growth reduction at 5 mg/L, around 25% growth reduction at 4 mg/L and up to 35% growth reduction at 3 mg/L.

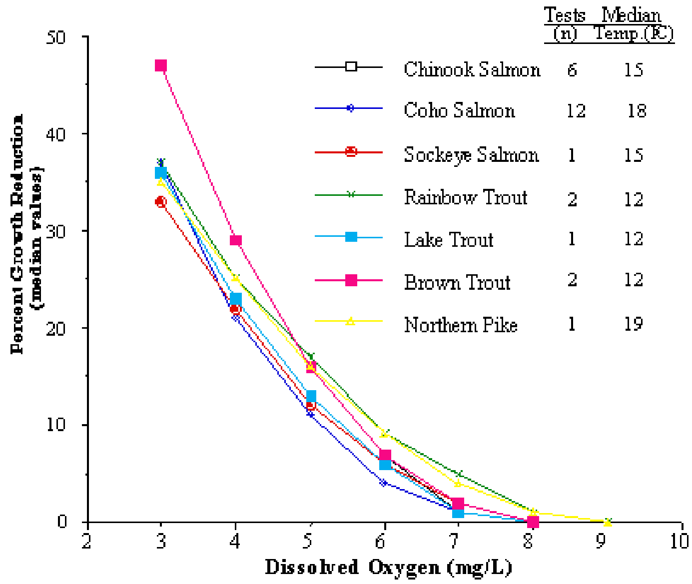

Figure 17: Summarised effects of dissolved oxygen concentrations on fish growth reduction (from US EPA 1986 data, redrawn in Department of British Columbia Environment Protection Division 1997);

It appears that controlling the oxygen levels based on the well-being of fish and preservation of their growth rates will also provide a degree of conservatism in terms of protecting other organisms. Best fish growth rates are obtained with dissolved oxygen values around 7 mg/L, with rapid reduction below this value. For the protection of fish and other aquatic organisms, it is recommended that long term dissolved oxygen levels remain above 5 mg/L (Vaquer-Sunyer & Duarte 2008).

Inorganic Nitrogen

Inorganic nitrogen can be subdivided into a number of individual components. Three of these will be addressed here, nitrates (NO_3-), nitrite (NO_2-) and unionized ammonia (NH_3). These nutrients form a very important part of the water column nitrogen cycle and have an important relationship with phytoplankton growth.

Nitrates

Nitrates are often introduced into the environment by agricultural fertilizers and natural runoff, but they are also the result of the two-step nitrification process. Ammonia is first transformed into nitrites (by bacteria and archaea), while in the second oxidisation stage nitrites are transformed into nitrates (by nitrobacter). The nitrates can then be taken up by algae for their growth or undergo anaerobic denitrification producing molecular nitrogen (N_2) that volatilises to the atmosphere.

In general toxicity of nitrate is not considered to be an issue in connection with marine aquaculture sites as concentrations in marine waters in general are very low (less than 1 mg/L). Nitrate N (NO_3-N) are considered as non-toxic to aquatic life (and particularly fish) at concentrations below 90 mg NO_3-N/L (US EPA 1976).

Several datasets are available for aquatic species, some of which closely relate to those cultivated in New Zealand (Westin 1974; Knepp & Arkin 1973; Dowden & Bennett 1965, etc. summarised in Camargo et al. 2005 and Hickey & Martin 2009). Toxicity limits for salmon and trout to nitrate are relatively high, with lethal concentrations (96h LC50) over 1300 mg NO_3-N/L for rainbow trout and King Salmon fingerlings. The salmonids appear to be particularly resistant compared to other fish - the Siberian sturgeon for example has been reported to have a 96h LC50 of 397 mg NO_3-N/L (Hamlin 2006).

The lethal concentration (96h LC50) for nitrite gives a general idea of the typical maximum acute concentrations, however maximum no effect concentrations are more relevant for the farming activity. These no effect conditions have been tested for earlier stages (i.e. eggs and fry) of salmon and trout, with much lower resulting concentrations: 30 days LOEC (Lowest Observed Effect Concentration) and NOEC (No Observed Effect Concentration) both ranging between 1.1 to 4.5 mg NO_3-N/L.

Invertebrates generally have a lower resistance to nitrate levels (e.g. Camargo et al. 2005), with 96 hours LC50 sometimes as low as 100 mg NO_3-N/L depending on species. Other values are reported in the literature but different timescales make them difficult to compare.

In summary, nitrates appear generally only critical for fish and aquatic organisms in the sense that interruptions in the nitrification process can lead to the build-up of nitrite at toxic levels.

Hickey & Martin (2009) reviewed nitrate toxicity for Environment Canterbury who have subsequently adopted a conservative (protective) freshwater guideline concentration of 1 mg NO_3-N/L for chronic exposure for all species considered (ranging from invertebrates to amphibians and fish). An equivalent acute concentration of 20 mg NO_3-N/L has been adopted for point source discharges. Hickey (2013) reviewed grading and surveillance guidelines for freshwater species and recommended that a default chronic range that would provide protection for 95% of species of 2.4 – 3.5 NO_3-N/L should be adopted for slightly or moderately disturbed systems[3].

The Waikato Regional Council has set a limit on the total nitrogen released from fed aquaculture through their Coastal Plan (Waikato Regional Council 2011). The implementation of this Plan would limit the release of total nitrogen to a maximum of 300 tonnes per year within the Firth of Thames and a maximum of 800 tonnes per year within the Coromandel marine farming zone.

Nitrites

Nitrites are a transitory product of the denitrification process transforming the ammonia/ammonium into nitrates. However, if the second stage of oxidation handled by Nitrobacter fails or slows down, an excess amount of nitrites may be left in the water column that could potentially reach toxic thresholds for fish and other aquatic organisms. Nitrite is also found in industrial waste, sewage and agriculture discharge and is a metabolic waste product of fish (Fuller et al. 2003).

[3] Ecosystems in which aquatic biological diversity may be adversely affected to a relatively small but measurable degree by human activity.

Although it is less toxic than ammonia (Fuller et al. 2003), elevated levels of nitrite still present a threat to fish health. Prolonged exposure to low levels can lead to stress and is often associated with stress-related disease such as bacterial ulcers and fin-rot.

At levels of long-term exposure greater than 0.5 mg/l skin and gill epithelia can be damaged and opportunistic bacteria and parasites may take advantage of stressed fish (Smith & Williams 1974). The main danger from high nitrite levels relates to actively transport across the gills and into the fish's bloodstream where it oxidises normal haemoglobin to methemoglobin. This leads to a change of fish blood to a brown colour, hence nitrate toxicity is often referred to as "brown blood disease".

Lewis & Morris (1986) have compiled data on nitrite toxicity and shown a wide range of toxic concentrations according to the environmental characteristics and species involved. For instance, 96 hours median lethal concentrations for the rainbow trout could be as low as 0.15 mg/L NO_2-N/L with 0.24 mg/L of chloride (Cl-), whereas that same 96-hr LC50 would reach 12.2 mg/L NO_2-N/L in the presence of 40.9 mg/L chlorides. Crawford & Allen (1977), which Lewis et al. (1986) included in their analysis, also stressed the inhibition factor of chlorides present in seawater. For Chinook salmon fingerlings, the 48hr LC50 was 19 mg/L of nitrites in freshwater while with chlorides 1070 mg/L nitrites only caused 10% mortality. Mortalities are 50–100 times lower in saltwater (Kroupova et al. 2005). The US EPA has chosen a nitrite nitrogen level of 0.06 mg NO_2-N/L for salmonids based on the tested minimum where no mortalities for rainbow trout was observed (Russo et al. 1974). However, it is possible that this criterion (most likely derived in freshwater) is overly conservative, especially if considered in the marine environment. The Canadian and British Columbia governments have accepted values of 1 mg NO_2-N/L.

Unionized Ammonia

The Ammonia chemical exists in two forms in solution: the abundant ionized Ammonium ion (NH_4+) and the toxic un-ionized or free Ammonia (NH_3). Ammonia is very toxic for fish and other aquatic organisms while Ammonium is almost innocuous. The partitioning of Ammonium and Ammonia is driven by the pH, the temperature and the salinity as shown in Figure 18.

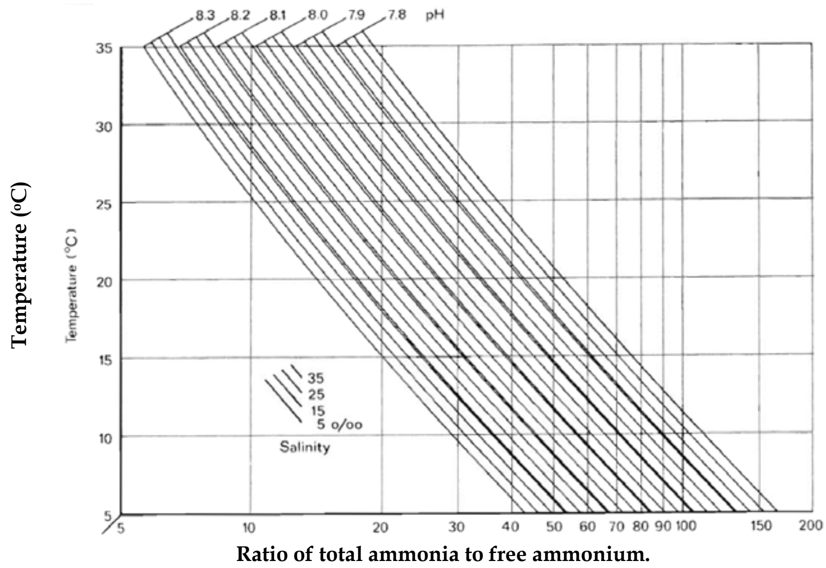

Figure 18: Ratio of total ammonia (sum of Ammonia, NH_3 and Ammonium, NH_4^+) to free ammonia nitrogen (NH_3-N) in saline waters at different values of pH, temperature, and salinity. From Spotte & Adams (1983), reproduced with permission.

Ammonia is toxic to marine life at relatively low levels (e.g. Table 14) and poses a threat to fish health. Raised ammonia levels affect fish health in several different ways. At low levels (less than 0.1 mg/litre NH_3) it is a strong irritant, especially to the gills. At higher levels salmon and trout can become greatly stressed creating conditions for opportunistic bacteria and parasites to proliferate. Elevated levels are a common precursor to bacterial gill disease.

Fish response to sub lethal levels are similar to those of any other form of irritation, i.e. flashing and rubbing against solid objects. Without water testing it would be very easy to incorrectly conclude that the fish had a parasite problem, when actually these behaviours may be related to long term exposure to ammonia.

Trout are usually considered to be particularly sensitive to ammonia. As shown in Fuller et al. (2003), Gila trout (*Oncorhynchus gilae*) survived 0.36 mg/L of unionized ammonia (UIA) nitrogen, but half of the population died after a 96 hours exposure to 0.47 mg/L of UIA nitrogen.

The criteria for protection of aquatic life established by the US EPA agency (US Environmental Protection Agency 2009a) 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 (Table 15 and Table 16) show a selection acute toxicity criteria for different marine species.

For seawater with salinity of 30 PSU, temperature of 15 ºC and pH of 8.4, maximum acute concentration allowed by EPA criteria is 4.2 mg total ammonia nitrogen (TAN)/L while chronic concentration is only 0.62 mg-TAN-N/L. In more restrictive pH and temperature conditions (e.g. T=25 ºC, pH = 8.6, and salinity = 30 PSU), the acute concentrations would reduce to 1.4 mg-TAN-N/L and the chronic concentrations for fish would be 0.22 mg-TAN-N/L. However, in varying conditions of pH, temperature and salinity, these thresholds may be overprotective.

A summary of trigger parameters and their values for acute and chronic toxicity (in estuarine and marine waters) for Chlorophyll-a, Dissolved Oxygen, Nitrate, Nitrite and Ammonia are presented in Tables 17a and 17b.

Table 14: Acute Toxicity of Ammonia for saltwater fish and invertebrates, US EPA (1989). Ambient Water Quality Criteria for Ammonia (Saltwater)-1989. EPA 440/5-88-004. Office of Water Regulations and Standards, Criteria and Standards.

Species	Mean Acute Value (mg/L NH_3)	Range of pH	Range of Temperature (°C)	Range of Salinity (PSU)
Winter flounder (Pseudopleuronectes americanus)	0.492	7.9–8.1	7.5	31
Red drum (Sciaenops ocellatus)	0.545	8.0–8.2	25–26	28–30
Sargassum shrimp(Latreutes fucorum)	0.773	8.07	23.4	28
Prawn (Macrobrachium rosenbergii)	0.777	6.8–8.3	28	12
Copepod (Eucalanus pileatus)	0.793	8.2	20.5	34
Planehead filefish(Monocanthus hispidus)	0.826	8.07	23.4	28
Copepod(Eucalanus elongatus)	0.867	8.0	20.3	34
White perch(Morone americana)	2.130	8.0	16	14
Striped bass(Morone saxatiois)	2.481	7.2–8.2	15–23	5–34
Mysid(Mysidopsis bahia)	1.021	7.0–9.2	19–26	10–31
Spot (Leiostomus xanthurus)	1.040	7.92	20.4	9.3
Atlantic silverside (Menidia menidia)	1.050	7.0–9.0	11–25	10–30
Inland silverside (Menidia beryllina)	1.317	7.1–9.1	18–26	11–33
Striped mullet (Mugil cephalus)	1.544	7.99	21	10
Grass shrimp (Palaemonetes pugio)	1.651	7.9–8.1	19–20	10–28
American lobster (Homarus americanus)	2.210	8.1	21.9	33.4
Sheepshead minnow (Cyprinodon variegatus)	2.737	7.6–8.1	10–33	10–33
Three-spined stickleback (Gasterosteus aculeatus)	2.932	7.6	15–23	11–34
Brackish water clam(Bangia cuneata)	3.080	7.95	20.2	9.2
Quahog clam (Mercenaria mercenaria)	5.360	7.7–8.2	20	27
Eastern oyster (Crassostrea virginica)	19.102	8.0	20	27

Table 15: Maximum Concentration of Total Ammonia Nitrogen (mg/L of Nitrogen) for Protection of Saltwater Aquatic Life (US EPA 1989).

Salinity equals 10 g/kg; Temperature (T) in degrees Celsius

pH	T = 0	T = 5	T = 10	T = 15	T = 20	T = 25
7	270	191	131	92	62	44
7.2	175	121	83	58	40	27
7.4	110	77	52	35	25	17
7.6	69	48	33	23	16	11
7.8	44	31	21	15	10	7.1
8	27	19	13	9.4	6.4	4.6
8.2	18	12	8.5	5.8	4.2	2.9
8.4	11	7.9	5.4	3.7	2.7	1.9
8.6	7.3	5	3.5	2.5	1.8	1.3
8.8	4.6	3.3	2.3	1.7	1.2	0.92
9	2.9	2.1	1.5	1.1	0.85	0.67

Salinity equals 20 g/kg; Temperature (T) in degrees Celsius

pH	T = 0	T = 5	T = 10	T = 15	T = 20	T = 25
7	291	200	137	96	64	44
7.2	183	125	87	60	42	29
7.4	116	79	54	37	27	18
7.6	73	50	35	23	17	11
7.8	46	31	23	15	11	7.5
8	29	20	14	9.8	6.7	4.8
8.2	19	13	8.9	6.2	4.4	3.1
8.4	12	8.1	5.6	4	2.9	2
8.6	7.5	5.2	3.7	2.7	1.9	1.4
8.8	4.8	3.3	2.5	1.7	1.3	0.94
9	3.1	2.3	1.6	1.2	0.87	0.69

Salinity equals 30 g/kg; Temperature (T) in degrees Celsius

pH	T = 0	T = 5	T = 10	T = 15	T = 20	T = 25
7	312	208	148	102	71	48
7.2	196	135	94	64	44	31
7.4	125	85	58	40	27	19
7.6	79	54	37	25	21	12
7.8	50	33	23	16	11	7.9
8	31	21	15	10	7.3	5
8.2	20	14	9.6	6.7	4.6	3.3
8.4	12.7	8.7	6	4.2	2.9	2.1
8.6	8.1	5.6	4	2.7	2	1.4
8.8	5.2	3.5	2.5	1.8	1.3	1
9	3.3	2.3	1.7	1.2	0.94	0.71

Table 16: Temperature and pH dependent acute criterion for Ammonia in freshwater based on Total Ammonia Nitrogen (mg-N/L) – mussels absent (US EPA 2009b).

pH	0–18	20	22	24	26	28	30
6.5	58.0	43.7	37.0	31.4	26.6	22.5	19.1
6.6	55.7	41.9	35.5	30.1	25.5	21.6	18.3
6.7	53.0	39.9	33.8	28.6	24.3	20.6	17.4
6.8	49.9	37.6	31.9	27.0	22.9	19.4	16.4
6.9	46.5	35.1	29.7	25.2	21.3	18.1	15.3
7.0	42.9	32.3	27.4	23.2	19.7	16.7	14.1
7.1	39.1	29.4	24.9	21.1	17.9	15.2	12.8
7.2	35.1	26.4	22.4	19.0	16.1	13.6	11.5
7.3	31.2	23.5	19.9	16.8	14.3	12.1	10.2
7.4	27.3	20.6	17.4	14.8	12.5	10.6	8.98
7.5	23.6	17.8	15.1	12.8	10.8	9.18	7.77
7.6	20.2	15.3	12.9	10.9	9.27	7.86	6.66
7.7	17.2	12.9	11.0	9.28	7.86	6.66	5.64
7.8	14.4	10.9	9.21	7.80	6.61	5.60	4.74
7.9	12.0	9.07	7.69	6.51	5.52	4.67	3.96
8.0	9.99	7.53	6.38	5.40	4.58	3.88	3.29
8.1	8.26	6.22	5.27	4.47	3.78	3.21	2.72
8.2	6.81	5.13	4.34	3.68	3.12	2.64	2.24
8.3	5.60	4.22	3.58	3.03	2.57	2.18	1.84
8.4	4.61	3.48	2.95	2.50	2.11	1.79	1.52
8.5	3.81	2.87	2.43	2.06	1.74	1.48	1.25
8.6	3.15	2.37	2.01	1.70	1.44	1.22	1.04
8.7	2.62	1.97	1.67	1.42	1.20	1.02	0.862
8.8	2.19	1.65	1.40	1.19	1.00	0.851	0.721
8.9	1.85	1.39	1.18	1.00	0.847	0.718	0.608
9.0	1.57	1.19	1.00	0.851	0.721	0.611	0.517

Table 17a: Summary of trigger values for acute and chronic toxicity for Chlorophyll-a in estuarine and marine waters.

Concentration based criteria
>60 µg/L, hypereutrophic for non-sensitive systems (Fereira et al. 2007, Bricker et al. 2003)
>70 µg/L Greek Eutrophication Assessment (Ahuja 2013)
>2.21 µg/L Greek Eutrophication Scale (Pagou 2009)
>5 µg/L, sensitive systems, example Florida Bay and Egypt National Eutrophication Assessment (UNEP/MAP 2007)
>20 µg/L eutrophic state within Huon Estuary (CSIRO 2000)
>8.9 µg/L, eutrophication and HABs of *Phaeocyctisglobosa sp* along the coasts of Belgium and Netherlands.(European Environment Agency 2013)(OSPAR 2008)
>40 µg/L, North Carolina, eutrophication standard (Ahuja 2013)
>12.2 µg/L along Mecklenburg Coast, Gulf of Riga and Gulf of Gdansk (European Environment Agency 2013)
> 55 µg/L within Southampton Eutrophic Waters (Holley & Hydes 2002)
Cell based criteria
>2000 cells/mL at Werribee River Estuary (Sherwood et al. 2005)
3 Alert Levels for algal biomass for New South Wales (Smith 2003):
Alert Level 1: 500 – 2000 cells/mL
Alert Level 2: 2000–15000 cells/mL
Alert Level 3:>15000 cells/mL

Table 17b Summary of trigger parameters and their values for acute and chronic toxicity in estuarine and marine waters.

	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 ºC, pH = 8.6, and salinity = 30 PSU)	Worst case > 0.22 mg NH_3-N/L (EPA) (T=25 ºC, pH = 8.6, and salinity = 30 PSU)

3.3 Water Column impacts from Eutrophication

In this section of the report a range of examples are presented where nitrogen enrichment has been observed across a range of conditions and environments. Where possible the nitrogen loading required to trigger an adverse environment response is identified.

Effect on the Water Column near Sea-Cages

Large-scale sea-cage systems contribute inorganic nutrients to the water column either directly through secretion of ammonia by fish, or indirectly through organic matter deposition and remineralisation (see Section 3.4). The extent to which nutrients in the water column are detectable near sea-cages is variable and dependent on the speed at which they are assimilated, the mixing characteristics of the receiving environment and background levels. McKinnon et al. (2012) and Burford et al. (2003) both concluded that changes in water quality resulting from aquaculture are difficult to resolve because other processes driving changes in water quality (such as tidal or seasonal forcing) result in large natural variability and the inputs of anthropogenic nutrient are 'swamped' by this natural variation. Ammonium levels have been observed to be markedly increased inside the cages and in the immediate vicinity of farms (e.g. Hartstein et al. 2010, 2011) but increased levels that can be attributed to farm sources even hundreds of metres away from farms are much more difficult to detect.

Impacts of Eutrophication

Increases in organic nitrogen from offshore upwelling have been responsible for growth of phytoplankton and subsequent blooms that can have an adverse impact on the marine environment and aquatic organisms (e.g. Smith 2003; Holley & Hydes 2002; HELCOM 2009) as discussed in Section 2. The effect of elevated levels of inorganic nutrients on sensitive organisms (i.e. corals, sponges and soft corals and other seabed habitats) depends on the species, the level and the duration of exposure to nutrients. Effects can include epiphytic algal growth (Hatcher & Larkum 1983; Harrison & Ward 2001), reduced or increased productivity and/or interruption of reproductive processes (Erftemeijer et al. 2012, Harrison & Ward 2001, Koop et al. 2001).

Many believe that these phase shifts are overwhelmingly dependent on the degree of herbivory present on any reef system (e.g. Littler & Littler 1984, Rasher et al. 2012) and further, that human harvest of marine herbivores plays a pivotal role in observed reef decline (Jackson et al. 2001, Bellwood et al. 2004, Mumby & Steneck 2008, Hughes et al. 2010). While the majority of the above research relates to soft-sediment communities the same principles can be applied to rocky reef ecosystems. For example, Dunmore & Keeley (2013) found significant changes at some sites in a sensitive reef community near two salmon farms in the Tory Channel, Marlborough Sounds. However, it was concluded that the observed changes were more likely due to natural recruitment and mortality rather than an influx of nutrients from the farms.

The tables below (Table 18 and Table 19) provide a review of published thresholds that may bring about community phase shifts in or have negative impacts on reef ecosystems. This provides some context in terms of providing an understanding of the levels of nutrients that may lead to adverse environmental effects. This is again particularly important as many of New Zealand’s farming areas are adjacent to very sensitive seabed habitats (e.g. Marlborough Sounds, the marine reserves of Golden Bay, and Stewart Island).

Table 18: Published thresholds for nutrients which may lead to a phase shift from marine coral to macroalgal-dominated reefs, and summary of potential mediating factors (i.e. herbivory); Nutrients thresholds are presented as DIN (Dissolved Inorganic Nitrogen) and SRP (Soluble Reactive Phosphorus).

Study Focus	Location	DIN (µM)	SRP (µM)	Findings	Conclusion	Reference
Nutrient thresholds for bottom up control of macroalgal blooms on corals reefs.	Hawaii, Kanehoe Bay	1.13	0.36	Author provides a review of DIN and SRP values that have been reported to lead to macroalgal blooms of a number of species on coral reefs. Concludes that thresholds for sustained macroalgal blooms are about 1.0 µM DIN, and 0.1 µM SRP. In warm and clear tropical waters, both phytoplankton and macroalgae can respond simultaneously to slight increases in water-column nutrient concentrations.	Supports the theory that DIN concentrations greater than 1.0 µM lead to a phase shift from coral to macroalgal or turf dominated habitats, as also shown in supporting studies (see reference list). This study argues that top-down controls are perhaps less important than suggested by other authors (i.e. Hughes 1994).	Lapointe (1997) and references contained within
	Hawaii west Maui	4.70	0.29			
	Red Sea, Gulf of Eilat	1.2				
	Martinique	1.2	0.25			
	Bermuda, Harrington Sound	1.66	0.14			
	Southeast Florida	1.61	0.19			
	Jamaica, Discovery Bay	12.2	0.17			
	Belize Barrier Reef, Man-O-war Cay	4.84	1.56			
Eutrophication threshold models (ETM) and associated water quality and nutrient threshold concentrations (NTC) developed in the late 80s and early 90s. The ETM, based on a number of studies, led to much controversy and debate, including development of theories as to the relative importance of top-down versus bottom up controls.	Great Barrier Reef (GBR), Queensland, Australia	More than 1.0	0.2 to 0.1	The derivation of water quality thresholds was based on correlations between changes in coral reef health indicators (e.g. coral cover, fleshy algal cover, coralline algal cover and coral recruitment/recovery rates) and water quality over time (temporal gradients) and/or space (spatial gradients). This group of studies set a threshold concentration for chl-a of <0.5 µg/L. Oceanica (2013) comment: *But see later work by De'ath & Fabricius (2008) which set a more conservative level for chl-a (0.63 µg/L and 0.32 µg/L for summer and winter, respectively).*	The ETM showed similarities in the annual mean concentration of DIN, SRP and chl-a that corresponded to the demise of coral communities, namely those in Barbados (Bell et al. 2007). Resulted in the widely held paradigm that DIN concentrations >1.0 µM could lead to a major phase shift. Note that these thresholds were further supported by Lapointe 1997.	Bell at al. (2007).

Study Focus	Location	DIN (µM)	SRP (µM)	Findings	Conclusion	Reference
The Enrichment of Nutrients on Coral Reefs Experiment (ENCORE). The ENCORE project consisted of fertilisation of microatolls.	One Tree Island, GBR, Queensland, Australia	36.2 (Phase II[2])	2.3 (Phase II[2])	No differences between treatments and controls were detected. The study found that addition of nutrients did not cause coral reefs to undergo a phase shift.	Concluded that results were contrary to the widely held view that thresholds developed as per the ETM. Oceanica (2013) comment: *These findings were later invalidated (see Bell et al. 2007). However, the findings are included here to highlight the level of uncertainty regarding the actual mechanisms behind phase shifts.*	Steven & Larkum (1993)
Nutrient enrichment experiment, 24 month duration. Gradients of herbivory pressure and of nutrient availability were included as treatments.	Belize, Carrie Bow Cay	Background DIN (µM): UD; 1–0.51 SRP (µM): UD; 0.07	Enrichment DIN (µM): 1.9–7.1 SRP (µM): 0.18–0.88	The results show that reduced nutrients alone do not preclude fleshy algal growth when herbivory is low, and high herbivory alone does not prevent fleshy algal growth when nutrients are elevated.	Despite many advocates, herbivory patterns alone do not consistently explain the distributions and abundances of benthic algae on coral reefs.	Littler & Littler (2006)
Nutrient enrichment study examining the addition of nutrients to lagoonal waters. Tested the effect of top down versus bottom up controls.	One Tree Island (GBR), Queensland, Australia.	Reevaluates the Phase I and II treatments tested as part of the ENCORE study. Previously, the ENCORE study had concluded that there were no significant effects despite additions of nutrients at concentrations far in excess of the ETM and nutrient threshold concentrations.		Assessment of ENCORE results, reinterpretation and refutation of conclusions. Discussion of algal growth rates and demonstration that even very small additions of nutrients can dramatically increase growth rates, (e.g. 1 ppb equals 33% increase in growth rate) but at higher concentrations, growth plateaus.	Reinterpretation of the ENCORE data by Bell et al. (2007) found strong evidence to support the validity of the previously accepted ETM and associated nutrient threshold concentrations.	Bell et al. (2007)
Field experiments assessed the effects of herbivore exclusion, nutrient enrichment, and the interaction of these factors on algal community development, sediment accumulation, and coral growth at two shallow reef flat sites.	Fiji, coral coast Viti Levu	Manipulative experiment where slow-release fertilisers pellets (19:6:12, N:P:K) were added to chambers. Response of macroalgae tested under exclusion and inclusion of herbivores.		Herbivore exclusion increased total and upright macroalgal cover by 9–46 times, upright macroalgal biomass by 23–84 times, and cyanobacteria cover by 0–27 times, but decreased cover of encrusting coralline algae by 46–100% and short turf algae by 14–39%.	This study concluded that herbivory is the dominant process controlling phase shifts between algae/coral. Oceanica (2013) comment: *Despite 30 years of research, the relative effects of herbivory and nutrient supply on macroalgal proliferation remain debated.*	Rasher et al. (2012)

Notes:
1. UD = Undetectable (below detection limit)
2. Based on phase two of the experiment

Table 19: Nutrient and Chlorophyll-a concentrations within eutrophic waters, examples of observed impacts (thresholds) from sub-tropical and temperate waters.

Location	Nutrient/Chl-a Concentrations					
	Ammonium	Nitrate	Total Nitrogen	Total Phosphorus	Phosphate	Chlorophyll-a
Egyptian coastal waters (UNEP/MAP 2007)	>2.00 µM	>4.00 µM			>1.99 µg/L	4–63 µm
Australian and New Zealand Environment and Conservation Council 1992 (CSIRO 2000)		10–100 µg/L	5.7 µM	>1.45 µM	>15 µg/L	63–125 µm
Southampton Water, UK (Holley & Hydes 2002)		>400 mmol/m^3		>20 mmol/m^3		55 mg/m^3
Temperate Rivers (Dodds et al. 1998)			>1500 µg/L		>75 µg/L	>30 µg/L, suspended >70–200 mg/m^{3}benthic
Les cotes de Sfax and Golfe de Gabes (Tunisia) (UNEP/MAP 2007)					0.2–0.9 µM	5–9 mg/m^3
Izmir Bay, Turkey (UNEP/MAP 2007)		0.13–27 µM			0.0.–10 µM	0.10–26 µg/L
Greek Estuaries (Pagou et al. 2002)		>16.66 µg/L			>21.08 µg/L	>70 mg/m^3 Greek Eutrophication Assessment >2.21 µg/L Greek Eutrophication Scale
Gulf of Bothnia (HELCOM 2009)			109 069 tonnes/year	4612 tonnes/year		
Gulf of Finland (HELCOM 2009)			129 671 tonnes/year	5006 tonnes/year		>4 µg/L
Gulf of Riga (HELCOM 2009)			58 417 tonnes/year	2659 tonnes/year		>4 µg/L
Danish Straits (HELCOM 009)			102 395 tonnes/year	2835 tonnes/year		
Chesapeake Bay, USA (Hagy et al. 2004; Chesapeake Bay Program 1994; Harding & Perry 1997)			81 Gg/year 2398.12 kg/ha	468.19 kg/ha (watershed unit area average total)		>8 mg/m^3

3.4 Seabed impacts from Eutrophication

Sea-cage aquaculture has the potential to impact the sediment when organic wastes settle beneath, or in close proximity to the sea-cages (Mazzola et al. 2000, Carroll et al. 2003). The deposition of organic material may lead to local organic enrichment or, under worst-case conditions, degradation of organic matter can release nutrients and cause regional-scale eutrophication (Pearson & Rosenberg 1978). The breakdown of organic material can lead to hypoxic conditions in sediments which can lead to local extinction of benthic populations (Gaston & Edds 1994), reduced growth rates of benthic fauna (Forbes & Lopez 1990, Forbes et al. 1994) and changes in benthic communities (Pearson & Rosenberg 1978, De Zwaan et al. 1992, Josefson & Jensen 1992, Stachowitsch 1992). Different organisms have different tolerances to a lack of oxygen, and thus the modifications in the ecosystem are not always predictable, but they generally result in a reduction in biodiversity of benthic communities. These changes typically occur because the less abundant, more sensitive species (such as sea fans) disappear first. These rarer species may play an important functional role in maintaining ecosystems (Dimitriadis & Koutsoubas 2011). More resilient species such as polychaete are known to be resistant to hypoxic or near-hypoxic conditions (Pearson & Rosenberg 1978, Gray 1992, Dauer et al. 1992).

Benthic infauna are widely regarded as sensitive indicators of environmental degradation and restoration in marine sediments (Clarke & Green 1988, Austen et al. 1989, Warwick et al. 1990, Weston 1990, Warwick and Clarke 1991, Agard et al. 1993, Ferraro et al. 1994). Numerous studies demonstrate a correlation between the level of organic enrichment and changes in infauna communities (e.g. Hargrave 2010) as summarised in Table 20. It is widely accepted that deposition rates of greater than 700 g C m^{-2} y^{-1} represent a critical value in terms of benthic infauna (e.g. Cromey et al. 1998; Gillibrand et al. 2002). Sediments exposed to this rate of deposition are considered to be degraded, resulting in reduced habitat biodiversity. Although useful in terms of defining a threshold of effects this value does not take into account duration of deposition, bioturbation, sediment reworking or recovery times (sometimes called remediation) once the source of organic enrichment has been removed.

The next section of the report discusses the importance of seabed recovery processes.

Table 20: Infauna community response to sedimentation and corresponding declines in sediment quality (modified from Hargrave 2010).

Organic Carbon deposition[1] (g C m^{-2} y^{-1})	Percentage reduction in infauna taxa richness[2] (SW index)	Oxygen stress[3]	Chemical remediation time (recoverable within 5 yr)[4]	Cross-reference with comparative studies
0	No detectable change	Pre-hypoxic	N/A	
1–36	<1.9% (>4)	Pre-hypoxic	N/A	<36 g C m^{-2} y^{-1} was found to have little effect on the infauna Cromey et al. 1998.
36.5	1.9% (>4)	Pre-hypoxic	N/A	36–365 g C m^{-2} y^{-1} enriched the infauna community Kelly & Nixon 1984, Frithsen et al. 1987, Oviatt et al. 1987, Maughan & Oviatt 1993.
255.5	38.5% (>4)	Pre-hypoxic	2.5 yr (Yes)	250–750 g C m^{-2} y^{-1} large change in infaunal trophic index (ITI) from 10–67 Cromey et al. 1998.
548	~50% (4–3)	Aperiodic	4.0 yr (Yes)	>548 C m^{-2} y^{-1} produced degraded sediment conditions Kelly & Nixon 1984; Frithsen et al. 1987; Oviatt et al. 1987; Maughan & Oviatt 1993.
730	55.5% (4–3)	Aperiodic	5.0 yr (No)	>700 g C m^{-2} y^{-1} led to a significant reduction in infauna diversity Cromey et al. 1998; Gillibrand et al. 2002. >750 g C m^{-2} y^{-1} Large change in infaunal trophic index (ITI) from 1–35 Cromey et al. 1998. ~767 g C m^{-2} y^{-1} led to enriched infauna Eleftheriou et al. 1982.
1460	75% (3–2)	Moderate	8.6 yr (No)	>1498 g C m^{-2} y^{-1} Degraded conditions for infauna (Eleftheriou et al. 1982).

1. Rate of organic carbon sedimentation (g C m^{-2} y^{-1}) from Cromey et al. (2002) and Chamberlain & Stucchi (2007).
2. Number of macrofauna taxa present (parentheses: % reduction in number of taxa from a mean of 51.5 for reference sites with sulphur concentrations <100 uM). Values from Brooks (2001) and Brooks & Mahnken (2003). SW Index = Shannon Weiner Index associated with different concentrations of sulphur and redox potential values (Hargrave et al. 2008)
3. Oxygen stress = qualitative assessment of infauna stress associated with levels of hypoxia (following Diaz & Rosenberg 1995)
4. Chemical remediation time = time (years) required for sediments to achieve sediment chemical characteristics not significantly different from reference sites. Estimates based on conservative sediment organic carbon mineralisation rates of 45 mmol per m^2 per year and 365 days of organic carbon deposition. Assumes all organic carbon deposited without re-suspension and steady state mineralisation i.e. mineralisation rates fluctuate seasonally, and typically increases in response to increased volumes of organic carbon.

Recovery

Numerous studies document the recovery of infauna once the source of organic enrichment has been removed (Brooks & Mahnken 2003, and references therein). Several of these document the recovery (also termed remediation) of infauna following the fallowing of sea-cages. Brooks & Mahnken (2003) describe two types of remediation: chemical and biological as summarised in Table 21.

The first stage in the process of recovery is the mineralisation of accumulated waste on the seabed. Only when this has occurred can the chemical environment of the sediment return to its pre-impact state and benthic macrofauna begin to recolonise (Morrisey et al. 2000); this process is termed chemical remediation. The second stage, biological remediation, is achieved when the infauna communities resemble[4] baseline (or pre impact communities) or other appropriate reference communities. Biological remediation may never completely occur, as different guilds of infauna which inhabit similar ecological niches may replace each another.

Table 21: Description of when chemical and biological remediation is achieved (Brooks & Mahnken 2003).

Remediation type	Description
Chemical	Reduction of accumulated organic matter with a concomitant decrease in free sediment sulfide concentrations and an increase in sediment redox potential under and adjacent to salmon farms to levels at which more than half of the reference area taxa can recruit and survive.
Biological	The restructuring of the infaunal community to include those taxa whose individual abundance equalled or exceeded 1% of the total invertebrate abundance at local reference stations. Recruitment of rare species representing less than 1% of the reference abundance is not considered necessary for complete biological remediation.

[4] It is unusual for very rare species to reach pre impact/reference levels.

4. Examples of Worldwide Regulations of Nitrogen Loading in Relation to Aquaculture

4.1 Introduction

Many different approaches are adopted internationally to define the carrying capacity of a particular system with respect to marine and inland aquaculture (e.g. Ross et al. 2013). Internationally, studies are increasingly taking a holistic approach to aquaculture planning to include social, ecological, physical and production aspects of carrying capacity (e.g. McKindsey et al. 2006). Studies with a specific focus on the management and regulation of nutrients (and in particular nitrogen) are not abundant in the literature. This is reflected in one of the findings of the recent International Council for the Exploration of the Sea report (ICES 2008) which highlighted that one of the fundamental gaps in knowledge relating to defining aquaculture carrying capacity was the identification of critical thresholds at which ecosystem disruption occurred.

The case studies presented in this section of the report provide details of the methods used in a number of locations to assess the impacts of nitrogen loading from aquaculture developments and to set limits on production, based on nutrient dynamics. The examples are presented in order of increasing complexity in terms of the available data, research carried out and methodologies used to develop regulations and/or management guidelines.

4.2 Cone Bay - Western Australian

Currently in Western Australia there is little aquaculture production and very little research relating to the management and regulation of nitrogen loading from aquaculture facilities. In 2012 the Western Australia Fisheries Department (WAFD) was given the mandate to develop two areas for aquaculture production. The first of these is in Cone Bay on the north-eastern coast of Western Australia. The WAFD commissioned an Environment Impact Statement study to estimate the carrying capacity of the wider Cone Bay area using numerical modelling and in-situ measurements with a view to sustainably increase the production of Barramundi from the previous level of just over 1000 tonnes per annum.

There were no known precedents in terms of the regulation of finfish farming with regard to nitrogen loading. However as a starting point a guideline developed by the Western Australia EPA relating to dredging activities was used. This regulation (the EAG 7) defines three levels of impact: Zone of High Impact (ZoHI); Zone of Moderate Impact (ZoMI) and the Zone of Influence (ZoI) as set out in Table 22. The application of these categories also considers a recovery time, specifically, how long any impacted habitat may take to recover once the stressors have been completely removed. Habitats requiring more than five years to recover are designated zones of 'high' impact, and habitats requiring less than five years are designated zones of 'moderate' impact.

Table 22: Narrative criteria for the Zone of High Impact (ZoHI), Zone of Moderate Impact (ZoMI) and the Zone of Influence (ZoI) as defined in EAG 7 by the Western Australian EPA. Source: DoE (2006).

Zone of Impact	Criterion
Outside the ZoI	Beyond the boundary of the ZoI, impacts should not be discernible from background. Inferred - No detectable impacts to biota
ZoI	Area where environmental quality may be affected, but where these are not expected to result in detectable impacts to benthic biota
ZoMI	Sub-lethal impacts, or recoverable within five years. Lies immediately adjacent to the ZoHI
ZoHI	Irreversible impacts on benthic organisms, or unable to recover within five years. In or immediately adjacent to dredge and dredge disposal sites

As stated above, with no specific nitrogen regulations in Western Australia a nitrogen specific criteria of 1.0 µmol DIN/L was developed following a review of the international scientific literature. In addition a duration effect was considered in terms of the impact to the benthic community as shown in Table 23.

Table 23: DIN concentrations selected for model interrogation.

Severity of effect	DIN
No effect	<1.0 µM or >1.0 µM for period less than six months duration
Potential benthic phase shift	>1.0 µM for periods longer than six months duration

The modelled fish farm scenario used consisted of a number of stocking rates (up to 20 000 tonnes of biomass) and a number of configurations in the bay. A feed conversion ratio of 1.6 was assumed with nitrogen content in the feed assumed to be 7%. Seventy five percent of this nitrogen was assumed to be released into the water column or deposited on the seabed. It was found that all farming scenarios considered met the "no effect" criteria due to the strong flushing of the bay with its 9–10 m tides (DHI 2013).

The study provided an approval process for future water space allowing either an existing grower to develop into the new water space or new companies to move in with the water space already pre-approved. The main outcome of the study was that Cone Bay and the surrounding area could sustainably produce 20 000 tonness per annum of Barramundi (DHI 2013).

4.3 Scottish Example Using a Simple Box Model to Regulate Nitrogen from Aquaculture Areas

In Scotland a simple box model/mass balance approach is used to assess the fate of soluble waste from fish farms. The models use the flushing rate of Scotland sea lochs, the total consented biomass of all the finfish farms in the loch (including non-salmonid species) and the nitrogen source rates. The model predicts the increase in dissolved nitrogen that would occur above background levels due to finfish farms assuming that nitrogen is only removed by tidal flushing (i.e. ignoring the nutrient cycling processes shown in Figure 1). The model does however consider the effects of the re-suspension of sediments to quantify the depositional footprint of individual farms.

The model uses the core physical parameters for a loch (basin volume, flushing time and the number of consented farms), the species of fish being produced and the consented biomass for each site. Feed wastage is assumed to be 5% and the diet is assumed to be 90% digestible and with a nitrogen content in the feed of 7.2% (Gillibrand et. al. 2002). SEPA (Scottish Environmental Protection Agency) regulates the finfish industry by consenting a maximum biomass that may be held on a site at any time.

For the purposes of this modelling exercise, it is assumed that after salmon smolts are first put into sea cages, with a negligible biomass, the maximum consented biomass is reached during the subsequent twelve months. During the second year of the production cycle, the maximum consented biomass is sustained whilst fish are being harvested, until eventually the biomass falls away as the site is cleared for fallowing.

The model is run for every sea loch in the database and an equilibrium concentration enhancement value predicted (Figure 19) and a ranking score is assigned based on predicted excess nitrogen concentrations. Values less than 0.3 are rated a one, 0.3–1 a two, 1–3 a three, 3–10 a four while anything above 10 µmol l^{-1} is a five (Gillibrand et al. 2002).

The potential degree of nutrient enrichment for each loch is therefore assessed and the systems most at risk of overexploitation can be identified, mapped, and the nitrogen loading reduced as necessary (in the case of existing farms this could require a reduction of biomass) or some sites ruled out totally for fish farming (Gillibrand et al. 2002).

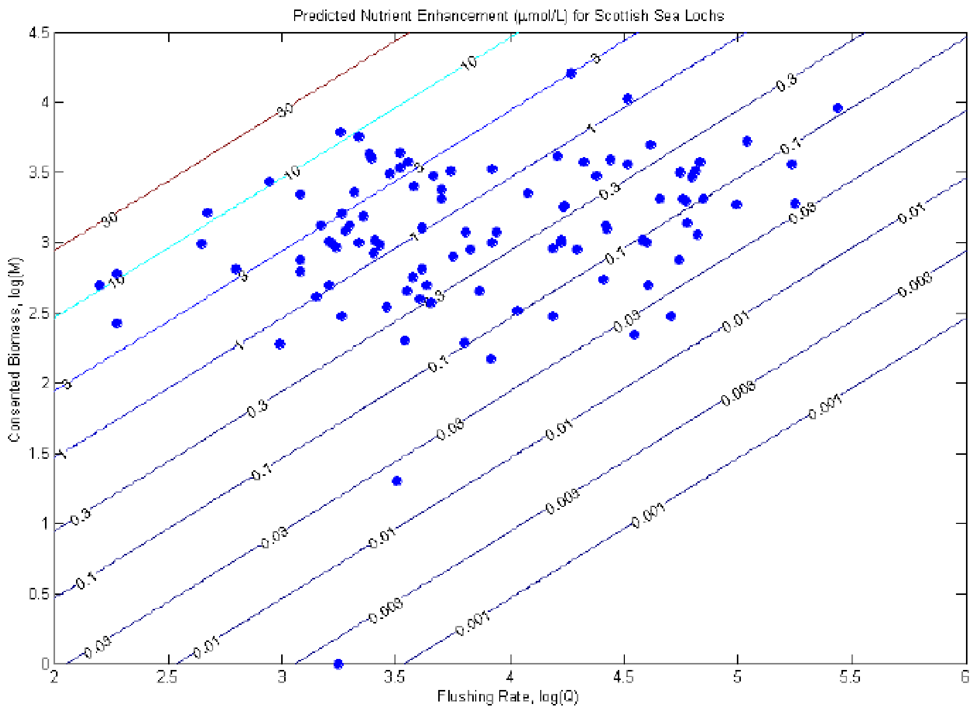

Figure 19: Predicted nutrient enhancement for all 111 Scottish Lochs based on 2002 consent conditions (Gillibrand et al. 2002, reproduced with permission).

4.4 Danish Nitrogen Loading, an Example: Limfjorden Estuary- Nutrient Dynamic Interactions with Mussel Aquaculture

Prior to 1970 Limfjord, one of the largest estuaries in Denmark, was an important mussel aquaculture and fishery area. By the 1970s the finfish fishery had declined dramatically and there was a concurrent increase in blue mussel production (Dinesen et al. 2011). Since the 1940s increased agriculture development in the area has led to total nitrogen (TN) and total phosphorus (TP) loading increasing six-fold through to the 1980s. Since the 1980s, the Danish Government has taken governance of the water quality of their coastal bodies more seriously. Public authorities governing the estuary include the Danish Ministry of Environment, Danish Ministry of Fisheries and Danish Ministry of Food, Agriculture and Fisheries and the European Union (Mathews 2010; Mongruel et al. 2011). Among the governing regulations at the European Union level are the European Union Water Framework Directive, NATURA 2000 Habitat Directive, Nitrate Directive and Birds Directive. On the national scale, action plans are being proposed to comply with these policies under the Fisheries Directorate (Dinesen et al. 2011). Such initiatives have resulted in a current annual loading of 12 tons N/km^2 and 0.92 tons P/km^2 surface area for the Limfjord catchment (Christensen et al. 2006). Since the implementation of these governing regulations, the TN and P loadings have decreased by 40% and 71% respectively. Unfortunately, anoxia, hypoxia and harmful algal blooms still occur between July and September within the inner south-eastern parts of the estuary (Dinesen et al. 2011).

To provide a better understanding of the system a suite of models including a bio-economic model, a fully coupled ecological model, an agent based mussel fishing model and a line mussel culture model were implemented within Extend Sim software framework (Dinesen et al. 2011).

In parallel with the modelling exercise, regulation of mussel fisheries and aquaculture have been implemented ranging from self-regulation by the mussel fishermen with an estimated weekly quota of 45 tons/week, governmental restrictions of 85 tons/week and seasons with closed mussel fishing (Dinesen et al. 2011).

The impacts of implementation of the nitrogen reduction based on the Water Framework Directive were modelled and indicated that full compliance to the nitrogen values prescribed under the EU Water Framework Directive (4.1 T/km^2/year) would cause a negligible decrease in phytoplankton biomass (less than 5%), some biomass decrease in shallow waters (around 25%), a 20% reduction in deep water mussel biomass and a 50% reduction of mussel fishery profit (Markager et al. 2006; Dinesen et al. 2011). In addition, it was estimated that if the reductions of both TN and TP with full compliance were done simultaneously it could lead to the closure of the aquaculture industry in this embayment (Dinesen et al. 2011).

As a result of the modelling results, the Danish Water Action Plans are going through a consultative process examining two possible Action Plans with the proposed targets shown in Table 24 (Dinesen et al. 2011). For comparison, the table also shows the historical observed values and the predicted values based on the initial regulation put in place.

Table 24: Total Nutrient Loadings within Limfjord. Mean values (tons/km^2/year) and proportions relative to the mean values used for the bio economic ecological-social-economic (ESE) model calibration period from 1985–2003 (Dinesen et al. 2011).

	Nitrogen		Phosphorus	
	T/km^2/year	%	T/km^2/year	%
1998–2003 mean observation values	10.3	100	0.44	100
1984–1986 mean observation values	12.3	120	0.91	207
2010 estimated values	6.8	66	0.28	64
Estimated Water Framework Directive full compliance (Markager et al. 2006)	4.1	40	0.23	52
Water Action Plan Proposal Jan 2010	5.6	54	0.24	60
Water Action Plan Proposal May 2010	7.1	69	0.24	60

4.5 Huon Estuary and D'Entrecasteaux Channel

The Huon Estuary and D'Entrecasteaux Channel produces around two thirds of the Tasmanian aquaculture industry output. To support the level of expansion that has occurred in the last 10 years several years of field data were collected so that three dimensional hydrodynamic and biogeochemical models of the region could be calibrated and validated. Using these modelling tools the nutrient budget in salmon-growing areas was used by the Department of Primary Industries (with support from the Tasmanian Aquaculture Review Panel) to assess sustainable yields for the area (CRC 2009).

The impacts of farm discharges on a regional basis were quantified by comparing model simulations over a one year period with and without farm nitrogen loads. In general, farm discharges had greatest impact on the nutrient concentrations and phytoplankton biomass in summer and autumn (alleviating the seasonal near-surface nutrient limitation) and thus promoting phytoplankton growth during this period (CRC 2009). Riverine and marine fluxes of nutrient into surface waters were by comparison relatively small during this period. Model results indicated that farm discharges had only a small impact on simulated dissolved oxygen concentrations due to the relatively high rates of flushing within the system.

To understand the impacts of the proposed farm expansions in the area simulations with 2002, 2006 and 2009 farm loads were modelled. The amount of nitrogen released from the farms was calculated by combining an estimated feed conversion ratio of 1.6 with the total estimate of salmon produced per year (including mortality). The amount of nitrogen per kilogram of feed can be calculated using information on feed composition provided by the feed companies (at 6–7% of the total weight of the feed). Given improvements in feed technology and feed conversion ratios the fish are presumed to consume the vast majority of the supplied food with very little sedimentation on the bottom (Macleod et al. 2002; 2004; 2006; Crawford et al. 2001; Crawford 2003). It was also assumed that approximately 86% of the nitrogen input into the environment is excreted by the fish in a soluble form with the majority of this being ammonium (Gowen & Bradbury 1987). All these factors were combined to give the following loadings;

- 2002, 843 T nitrogen;
- 2006, 1215 T nitrogen; and
- 2009, 2590 T nitrogen.

These nutrient loads were then incorporated into the three-dimensional coupled hydrodynamic, sediment and biogeochemical model to evaluate the environmental impact. The model (validated against field observation) simulated the seasonal cycling of organic and inorganic carbon, nitrogen, phosphorus and oxygen through multiple phytoplankton, zooplankton, nutrient and detrital pools. Model results allowed identification of the system-wide spatial and temporal footprint of the farms in the context of environmental guidelines. Results indicated that while there were some changes to phytoplankton community structure, when combined with adaptive management and ongoing monitoring the regulatory agency were able to grant resource consents with well-defined limits for aquaculture development in Huon River, and D'Entrecasteaux Channel.

In terms of the management of nitrogen the long-term monitoring strategy included nitrogen trigger levels (in this case only for ammonium) of 0.32 µM/L at the surface and 0.42 µM/L at the seabed (CRC 2009).

4.6 Macquarie Harbour Salmon Farming Expansion

The work carried out in Macquarie Harbour, Tasmania provides an example of the development of nitrogen based regulations associated with the expansion of salmon farm production. Macquarie Harbour has had aquaculture activities taking place for more than 20 years over a number of leases totalling 524 ha. Aquaculture production in 2009 had reached approximately 7200 tonnes (DPIPWE 2009). In 2011 a proposal was made to expand existing salmon farming operations in Macquarie Harbour by 360 Ha of leasable area with a view to maximising production in the harbour.

The following section of the report provides an overview of the regulatory process that was carried out and enabled the Macquarie Harbour expansion to proceed.

The Macquarie Harbour expansion was a cooperation between the three existing growers of Macquarie Harbour (Tassal Operations Pty Ltd (Tassal), Huon Aquaculture Group Pty Ltd, and Petuna Aquaculture Pty Ltd). Collectively the consortium set out a proposed development plan and accompanying Environment Impact Statement. The key findings of the Environment Impact Statement allowed stakeholders and regulators to understand the benefits and potential impacts of the proposal. All stages of the study were conducted with open dialogue between growers, authorities (DPIPWE and EPA), consultants and various experts from universities and NGOs.

It was decided at the start of the process that a full ecological model would be calibrated and used to quantify nutrient exchange and phytoplankton dynamics as this approach was considered "best practice" by the industry and regulators alike. The modelling was supported by several field campaigns.

Two sets of environmental water quality guidelines were referenced throughout the Environment Impact Statement – the ANZECC guidelines (ANZECC 2000[5]) and guidelines from the US EPA. These guidelines were used as benchmarks for the monitoring data and model results and helped to define sustainable levels of production.

Under the ANZECC framework there is scope to create area specific guidelines (e.g. if background levels are already above ANZECC guidelines). Such area specific guidelines were established using data from a targeted predevelopment monitoring programme which also provided valuable "pre-development" model calibration data. As part of the framework development process, environmental triggers for nitrogen were developed for the purpose of future monitoring and regulation of the harbour in terms of water quality as outlined in in Section 2.9.

The US EPA criteria for health of aquatic organisms were also used to provide context as to the significance of both the measured and modelled values in regard to the acute and chronic toxicity to fish and other fauna. Additional information was also drawn from the water quality review undertaken by Hickey & Martin (2009).

A range of scenarios were simulated so that the carrying capacity of the harbour could be assessed in terms of production levels. Combinations of standing biomass of fish and stocking density were modelled. Standing biomass of fish rates of 30, 45 and 60 T/ha and stocking densities of 20 and 25 kg/m^3 were modelled across all proposed farm leases. To derive nitrogen loads, a conservative approach was used. For every tonne of salmon grown, the feed conversion ratio was assumed to be 1.6 (with feed having a nitrogen content of 6.6%). It was assumed that 75% of the nitrogen from the feed was released into the water column and sediment, and 25% taken up by the fish (Hartstein et al. 2012b). Results indicated that the scenario with 45 T/ha biomass and 20 kg/m^3 stocking rate had potentially unacceptable effects in terms of reduced dissolved oxygen levels. The scenario with 30 T/ha biomass and 20 kg/m^3 stocking rate had generally acceptable environmental impacts for most parameters. Under worst case conditions there was still some dissolved oxygen depletion effects between neighbouring leases.

Based on the results from a broad range of potential farm scenarios (i.e. combinations of standing biomass of fish and stocking density), a Final Model Scenario was specified based on the maximum long term production for each individual cage given a total biomass of 29 441 tonnes. This level of production would not be reached for at least five years from the commencement of use of the expanded area. Results of monitoring during this initial five years would provide input to the adaptive management plan and further validation of model results. Also, individual lease holders would have a combination of juvenile, adult or harvest stock in any given year so the assumption of maximum production in any year provides a degree of conservatism. Within the Final Model Scenario two seasonal extremes were simulated as follows:

Summer Conditions Scenario: The summer period aimed to simulate the maximum standing biomass and average climatic and hydrological conditions as observed in the harbour during summer months. The summer period represented moderate and consistent wind conditions (derived from the data available), and slightly lower water inflow when compared to winter. Surface temperatures were typically higher than deep water temperatures and river inflows colder than harbour surface waters.

Winter Conditions Scenario: The winter period was designed to simulate significantly higher water column mixing conditions in the harbour. Records indicated the occurrence of slightly higher mean wind speeds during winter when compared with summer months although summer displayed significant peak maxima speeds. Winter stratification patterns down the water column were usually inverted

[5] It should be noted that the ANZECC 2000, guidelines are currently under review.

compared to that of summer patterns, with higher water temperatures typically found in bottom waters than in surface waters.

Following consultation with the expert panel, the Federal Environmental Protection Agency (EPA), and Tasmanian DPIPWE, interim trigger levels of nitrogen were set. If future monitoring finds nitrogen levels above these triggers there is a planned management response (see Section 5). Another aspect of the adaptive management plan was to cap the biomass within the harbour for approximately 18 months while further water column and seabed respiration data were collected. During this period production was limited to 54% of the maximum biomass calculated to be sustainable in the EIS – in this case around 16 000 tonnes.

In addition to the development of triggers levels the implementation of a rigorous monitoring programme and understanding of model results ensures that both harbour wide effects and between and within fish farm effects can be quantified and understood (Hartstein et al. 2011).

5. Strategies to Manage Nitrogen Loadings in Embayments with Aquaculture Production

5.1 Introduction

While there are many examples of the long term management of nitrogen loadings in coastal embayments (e.g. Vermaat et al. 2012), the majority do not focus on the impacts of aquaculture activities. This section of the report reviews the management and monitoring of aquaculture derived nutrient loadings in two areas that have direct relevance to the situation in New Zealand. Both examples are from Tasmania where there has been a well planned expansion of existing production. However, the adaptive management of aquaculture expansion is not unique to Tasmania and there are also examples of this process working in New Zealand. Projects funded by the MPI Aquaculture Planning Fund carried out by the Waikato Regional Council[6] build on the work of Zeldis et al. (2005) in establishing an initial framework for adaptive management of aquaculture in the Firth of Thames (based on an understanding of phytoplankton depletion and the need for farm scale and Firth wide triggers for management action). Internationally the concept of staged development (or Modelling-On growing-Monitoring) has been widely used as an approach to adaptive management (Hansen et al. 2001). Existing aquaculture areas in the Marlborough Sounds have been developed using such an approach, applying the concept of ecological impact zones (MPI 2013).

5.2 Huon Estuary and D'Entrecasteaux Channel Water Quality Management

The major natural sources of nutrients into the Huon Estuary and D'Entrecasteaux Channel include the intrusion of oceanic waters during winter and various inputs from land, streams and rivers (Butler et al. 2000). Changes such as increased sedimentation, nutrient inputs, sewage, habitat loss and regulation of freshwater flows are recognised as potential threats to the system (Jordan et al. 2002, RPDC 2007). Finfish farming is now the largest known anthropogenic source of nitrogen to the ecosystem (CRC 2009). An environmental monitoring system was set up in 2003 to ensure that environment health was maintained and that the ecosystems could sustain a wide range of uses (CRC 2009).

As described in Section 4, a three dimensional coupled hydrodynamic, sediment and biogeochemical model was used to evaluate the environmental impact of the existing and proposed salmonid fish farms. The monitoring program included monthly sampling for water quality parameters (phytoplankton biomass, nutrients and DO) at sensitive sites identified from analysis of model results. These sites covered a transect along the centre of the channel in the Huon Estuary and in D'Entrecasteaux Channel (CRC 2009) as well as model boundaries (rivers and other sources). The study recommended a full review of monitoring data at least every five years (regardless of the state of the environment).

Based on the results of the modelling study, the monitoring data and the recommendations of Fletcher et al. (2004) a three step risk level approach was adopted to determine the severity of changes in ecosystem functioning as indicated by parameters exceeding their trigger values. The trigger values and location of monitoring sites ensure that the three dimensions of risk (intensity, duration and spatial extent) are considered.

For NH_4 the trigger values were set to 0.32 μM at the surface and 0.42 μM at the seabed within the Huon River with slightly lower levels in the D'Entrecasteaux Channel of 0.12 μM at the surface and 0.27 μM at the seabed.

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[6]http://www.fish.govt.nz/NR/rdonlyres/B19A19C5-6B76-4BE7-881E-B5CA5177122F/0/APF1204WebSummary.pdf

The trigger criteria in this case were:

- A Level One risk is triggered if :
    - the summer mean is up by 25%, or
    - three successive annual means are greater than the trigger, or
    - the mean for any one site is up by 50%.

- A Level Two risk is triggered if :
    - the summer mean is up by 50%, or
    - at any site, 8 of 10 successive annual means are greater than the trigger, or
    - the mean for any one site is up by 200%.

- A Level Three risk is triggered if:
    - the summer mean is up by 100%, or
    - the summer mean is greater than the ANZECC guideline (1 µM).

The three levels of risk require an increasing investigative action as follows:

- Level One: add more spatial and/or temporal sampling to evaluate extent of change,
- Level Two: Commission expert review of the monitoring data, monitoring strategy and carry out biogeochemical model run focussing on the impacted sites, to attempt to understand cause for Level Two risk trigger. Report to stakeholders and review management recommendations.
- Level Three: Commission a study to understand the cause. Study to include field observations and new modelling as necessary. Formal expert report to stakeholders with recommendations for planned management intervention and remedial action.

5.3 Macquarie Harbour Inorganic Nitrogen Management

As introduced in Section 4.6, a detailed ecological model looking at nutrient exchange and uptake from phytoplankton was utilised to determine the carrying capacity (maximum sustainable biomass) within Macquarie Harbour. Based on this study nutrient triggers were determined and a monitoring and management programme developed.

Based on a precautionary approach the interim triggers shown in Table 25 were used in management of nitrogen and oxygen in Macquarie Harbour (Macquarie Harbour AMA 2012. The percentiles for triggers were derived using the EPA process for trigger setting for estuaries (DPIW Report No WA 08/52). Model results from a simulation of one year of peak production were used to define 80th percentile values for ammonia and nitrate and a 20th percentile value for dissolved oxygen. All of the proposed trigger levels are substantially below levels thought to have an adverse impact on water quality or seabed organisms, i.e. US EPA (1999); Batley & Simpson (2008); Hickey & Martin (2009).

Table 25: Macquarie Harbour interim trigger levels (Macquarie Harbour AMA 2012).

Water Quality Parameter	Location	Percentile	Interim Limit
Ammonia	Surface water	80th Percentile	0.033 mg/L
Ammonia	Bottom water	80th Percentile	0.024 mg/L
Nitrate	Surface water	80th Percentile	0.053 mg/L
Oxygen	Surface water	20th Percentile	6.82 mg/L

Water quality is monitored at 12 locations in and around the harbour including ocean and river boundaries from which rolling three monthly median values will be determined. If the above triggers are exceeded and the exceedance can be attributable to farming activity, management options including biomass redistribution, destocking or reduced smolt stocking will be considered as summarised in Table 26. As part of the adaptive management plan the triggers will be reviewed every 2 years (based on new monitoring data and model refinement). This feedback between trigger level monitoring and model calibration provides more accurate and "tighter" triggers over time.

Table 26: The Management Actions based on percentile trigger levels (Macquarie Harbour AMA 2012).

Trigger criteria	Management Actions
Regional 3 monthly median are <u>less than</u> trigger percentile.	No action.
Stage 1 Trigger Regional 3 monthly median <u>reaches</u> the trigger percentile	Investigation/attribution analysis (assessed in conjunction with site-specific trends). Identify most effective mitigation strategy.
Stage 2 Trigger Regional 3 monthly median <u>exceeds</u> the trigger percentile	Reassess the model, biomass redistribution and/or destocking mandatory. Reduce smolt input.

6. Conclusions and Recommendations

This review has highlighted the range of approaches that have been adopted worldwide in understanding the potential impacts of aquaculture developments in terms of the impacts of nitrogen on the marine receiving environment.

In Section 2 of the report a summary of the understanding of the hydrodynamics and nutrient budgets of a number of key New Zealand sites and similar overseas sites has been provided. The review highlighted the lack of data at some of the international sites but also showed that in terms of the New Zealand sites a lot of information has already been gathered to provide a good understanding of the nutrient budgets of the sites considered.

In Section 3 of the report a review of the potential adverse effects of nitrogen on benthic communities, and within the water column and seabed are presented. Data presented in this section of the report provides context for defining the potential impacts of aquaculture.

In Section 4 of the report a number of international case studies are presented in terms of how nitrogen loading has been regulated to minimise the impacts of proposed new aquaculture facilities or the expansion of existing aquaculture areas. A variety of modelling methodologies and management frameworks have been used with the aim of limiting nutrient loads to ensure the long-term health of a coastal area and ensure sustainable aquaculture production. All the case studies showed the following similarities:

1. the setting of some environmental trigger(s) at which point a review of production, management or some other intervention occurs,
2. an understanding of the dynamics of the system (through modelling and/or monitoring) and
3. quantification of the impacts of the existing and/or proposed aquaculture production in the context of the natural variability of the system.

Section 5 of the report details the environmental triggers, monitoring programme and adaptive management options used in the Tasmanian aquaculture industry. These case studies are used as both the Tasmanian sites considered have gone through relatively rapid expansions of aquaculture development, had good baseline information prior to expansion and the expansion consenting process has rigorously tested the modelling methodologies and science behind developing environmental triggers for the protection of relatively diverse ecosystems.

The modelling methodologies and monitoring regimes selected for a region must be relevant to the hydrodynamics, nutrient loads and to the planned extent of aquaculture. The box model/mass balance approach used in the Scotland Lochs (Section 4.3) provides good management guidelines because of the relatively semi-enclosed nature of the lochs and the good database information (e.g. maximum consented conditions and flushing rates). Such an approach could not be universally applied in New Zealand because many of our aquaculture areas are relatively open, highly connected systems (e.g. the Firth of Thames).

In the Faroe Islands an understanding of the oceanic exchange processes, relatively rapid flushing times of aquaculture areas combined with well tested management practices (including planned fallowing) provides the industry with a sizeable sustainable aquaculture industry. Details of the management practices and any nutrient guidelines could not be found for the Faroe Islands.

The use of fully coupled, three dimensional hydrodynamic, sediment and biogeochemical models in the Tasmanian case studies benchmarks the approach that we recommend is used in New Zealand. Such models provide quantification of the impact of aquaculture expansion in the context of other sources of nitrogen (rivers and nutrient upwelling) and the natural variability. Model results help to determine how changes in the seabed and water column enhancement of nutrients may impact on marine ecosystems. The models can also be used to estimate the likely recovery period of the seabed and estimate changes in the phytoplankton volume and community structure due to additional nutrient loading, which provides useful guidance for both growers and regulators.

In the context of the New Zealand aquaculture industry the RMA framework promotes the concepts of monitoring and adaptive management as tools in understanding and assessing the potential impacts from aquaculture development. The case studies presented in this report show that the scale of development of aquaculture in New Zealand is generally much larger than those occurring overseas. In terms of the effects of nitrogen on the marine environment the scale, staging and type of development needs to be considered. The overseas examples presented provide valuable information on the planning process used for new aquaculture developments and the monitoring framework put in place which can be applied to New Zealand's aquaculture industry. The case studies provided highlight the need for understanding not only the local effects of aquaculture (i.e. cage/cage and farm/farm interactions and impacts) but also the wider cumulative effects of aquaculture development particularly in areas already impacted by relatively high loadings of nutrients.

So that stakeholders have confidence in the model results it is crucial that a wide range of field data spanning a broad range of conditions are collected both in a baseline ("pre development") phase and during the development of management guidelines and trigger formulation and refinement. The coverage and duration of field data collection will be very site specific but it is recommended that a minimum of 12 months of data is collected for the purposes of model calibration and for defining baseline water quality conditions. The number of stations to be sampled and monitored should be decided on a case by case basis and to some extent will also depend on the data that has already been collected at a site. Parameters to be monitored should include tides, currents, waves, dissolved oxygen, nutrients (NH_4, NO_3), chlorophyll-a, temperature, salinity, sediment characteristics (sediment grain size, POC/PON content) and if possible, sediment denitrification rates. Field data programmes should be designed around findings from models and should provide data in both the near-field and far field. Data collected should be used by regulators (in terms of management triggers and other consent conditions) and operators (in terms of day-to-day management). Where possible a modelling framework that combines the collection of the in-situ real time data, remote sensing information with forecast models should be developed. Building on this framework, decision support systems should be developed to enable all stakeholders (i.e. regulatory and operational) to obtain and understand model outputs and field data via web based applications.

The trigger level approach adopted overseas (and being developed in New Zealand) is recommended as the standard for assessing how a system is responding to the establishment or expansion of aquaculture areas and should become an integral part of adaptive management plans. The process of establishing such triggers necessitates developing a thorough understanding of the marine system being considered through a combination of modelling and monitoring. With such tools in place the relative roles of natural variability and the impacts of potential aquaculture development scenarios can be assessed.

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9. Glossary

Ammonia (NH_3)

Ammonium (NH_4^+)

Total Ammonia Sum of NH_3 and NH_4^+

Dinitrogen (N_2)

Nitrate (NO_3^-)

Nitrite (NO_2^-)

Nitrogen oxides (N_2O and NO)

Denitrification In anoxic conditions bacteria use nitrate converting it to nitrous oxide (N_2O) or dinitrogen (N_2) gas.

Nitrification In anoxic conditions bacteria use ammonia and oxidise it to nitrite and nitrate.

Cyanobacteria photosynthetic nitrogen fixing bacteria which can convert N_2 into ammonia (NH_3), nitrites (NO_2^-) or nitrates (NO_3^-)

Dissolved Inorganic Nitrogen (DIN) sum of the concentrations of nitrate and ammonia.

Particulate Organic Nitrogen (PON) The insoluble (particulate) organic matter in the water column.

Dissolved Organic Nitrogen (DON) The dissolved organic matter in the water column.

Total Dissolved Nitrogen (TDN) sum of the DON plus DIN.

Total Organic Nitrogen (TON) Organically bound nitrogen being the sum of DON and PON.

Total Nitrogen (TN) sum of all organic and inorganic nitrogen components in the water column: phytoplankton and zooplankton, suspended microphytobenthos, dissolved inorganic nitrogen (nitrate and ammonia), dissolved organic nitrogen, labile detritus.

Phytoplankton Photosynthesising component of plankton which inhabit the upper layer of the ocean (e.g. dinoflagellates and diatoms).

Zooplankton Animal constituent of plankton (e.g. small crustaceans and fish larvae).