Ministry for Primary Industries
Manatū Ahu Matua

LITERATURE REVIEW OF ECOLOGICAL EFFECTS OF AQUACULTURE

Cumulative Effects

Photo courtesy of Phil Kirk

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Cumulative Effects

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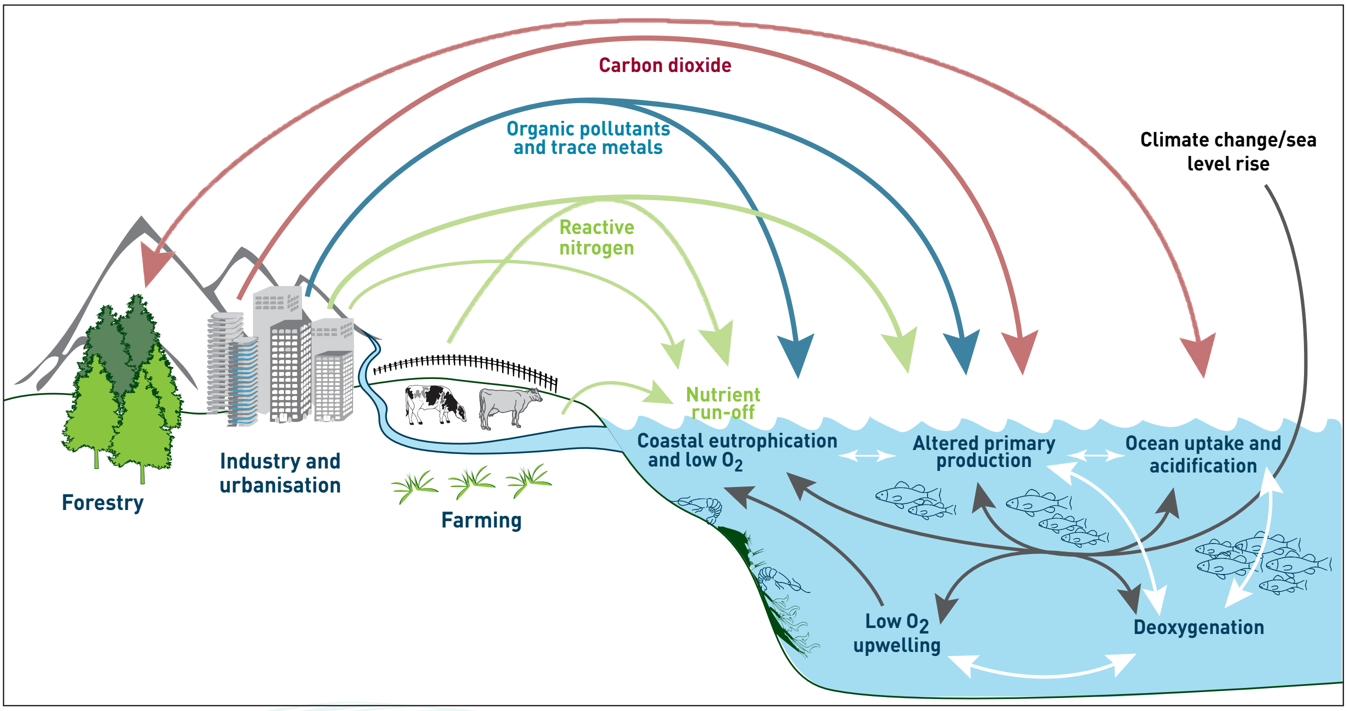
12.1 Introduction

Note: *The following chapter draws heavily on previous reviews of the environmental effects of finfish (Forrest et al. 2007) and non-finfish aquaculture (Keeley et al. 2009). Complementary information on the wider ecosystem effects of aquaculture in relation to the water column is provided in Chapter 2: Pelagic Effects.*

The previous chapters have focused on issue-specific ecological effects of aquaculture developments on the marine environment. Our understanding of these effects is largely based on farm-scale assessments and monitoring; the potential for wider-ecosystem effects (e.g. far-field benthic enrichment, effects on fish populations, migrating mammals, etc) is acknowledged but is far less understood. As aquaculture develops and the number of farms in coastal waters increases, wider-ecosystem issues become more important to consider due to the cumulative environmental effects that could arise from multiple farms combined with additional anthropogenic stressors affecting the marine environment.

Environmental sustainability of both land-based and maritime industries requires an understanding of cumulative effects on the environment and the ability to measure environmental change in response to multiple stressors. Coastal waters are the ultimate receiving environment for a range of contaminants derived from upstream catchments and sea-based industries (e.g. feed-added aquaculture). Additional activities such as fishing, tourism, shipping, and coastal development present multiple stressors that cumulatively interact with natural processes and affect marine environmental quality (see Figure 12.1 for an example of multiple stressors interacting with natural processes). Many of these activities (and in turn their effects) operate on different spatial and temporal scales. The coastal marine environment is physically dynamic and conditions are inherently variable in response to topography, weather and climate-related processes; hence climate change will also contribute to long-term environmental change and could influence the extent to which various human activities impact on the marine environment.

Figure 1: Conceptual diagram of cumulative anthropogenic effects in marine ecosystems.



Note: Diagram includes inputs of materials into the system (colored arrows), indirect effects of climate change and altered ocean circulation (black arrows), and interconnectivity of ocean biogeochemical processes (white arrows). Figure from Doney 2010.

12.1.1 Defining cumulative effects

Aquaculture developments in New Zealand currently occur within 12 nautical miles of the coast (the Coastal Marine Area; CMA). Environmental effects legislation for consenting within the CMA falls under the Resource Management Act 1991 (RMA), and is also guided by the 2010 NZ Coastal Policy Statement (NZCPS). Both require that cumulative effects be addressed to ensure environmental sustainability in coastal catchments and nearshore waters. A cumulative effect is referred to in Section 3 of the RMA as an *effect which arises over time or in combination with other effects*. There is a considerable amount of case law around the broader definition of a cumulative effect, which considers both positive and adverse effects, temporary and permanent effects, as well as past, present and future effects (see Milne & Grierson 2008). Within the context of aquaculture development in the marine environment, cumulative effects are defined here as:

> *Ecological effects in the marine environment that result from the incremental, accumulating and interacting effects of an aquaculture development when added to other stressors from anthropogenic activities affecting the marine environment (past, present and future activities) and foreseeable changes in ocean conditions (i.e. in response to climate change).*

The NZCPS highlights the importance of addressing cumulative effects and contains elements relating to integrated and coordinated management between the land and sea. The following policy (7(2)) from the NZCPS is of particular relevance to managing cumulative effects in the marine environment:

> *Identify in regional policy statements, and plans, coastal processes, resources or values that are under threat or at significant risk from adverse cumulative effects. Include provisions in plans to manage these effects. Where practicable, in plans, set thresholds (including zones, standards or targets), or specify acceptable limits to change, to assist in determining when activities causing adverse cumulative effects are to be avoided.*

As described in the previous chapters, aquaculture can lead to a range of effects on the marine environment and, at some level, contribute to cumulative environmental change. However, a sustainable aquaculture industry ultimately depends on a healthy marine environment and most importantly, high water quality. Land-sea connections and management of the cumulative effects of land-based activities on the downstream marine environment must also be considered. Indeed, NZCPS policy 8 (c) seeks to *ensure that development in the coastal environment does not make water quality unfit for aquaculture activities in areas approved for that purpose*. Addressing cumulative effects in the marine environment also requires an ecosystem-based approach to resource management. As stated in a recent Organisation for Economic Co-operation and Development (OECD) report on agriculture's impact on aquaculture *Intensification and technological improvement within agriculture need to be done in the framework of total ecosystem management, which considers impacts on aquaculture and other ecosystem services* (Diaz et al. 2012).

12.1.2 Individual farm versus regional scale assessment of cumulative effects

The management of cumulative effects in the marine environment can be addressed using a two-tiered approach that not only considers the contribution of effects from individual developments, but also an overall regional assessment of wider environmental change in response to the many stressors impacting on the marine environment (e.g. Dubé 2003). This chapter focuses on how effects associated with aquaculture developments may add to, and interact with existing effects in the marine environment, thus contributing to cumulative effects on the environment. The broader, regional approach of assessing cumulative effects associated with many developments and actions impacting the marine environment, and subsequent assessment of the contribution of aquaculture to environmental change (actual or forecasted), is beyond the scope of the aquaculture industry alone and would best be addressed by government agencies (e.g. Regional Councils, Department of Conservation, Ministry for the Environment). Although beyond the scope of this chapter, it is critical that this task is undertaken in order to develop ecosystem-based management programmes in an adaptive manner.

Critical to regional assessments of cumulative effects in the marine environment is accessibility and co-ordination of datasets, including those derived from consent monitoring at individual farms, and long-term SoE monitoring programmes. Standardised monitoring requirements for aquaculture is an important step to ensuring usefulness of consent monitoring datasets within broader-scale assessments. The requirements for assessing and managing cumulative effects fall beyond the scope of a single consent applicant or industry and is best dealt with through a regional council (e.g. Dubé 2003; Hargrave 2005, Zeldis 2008a,b) or central government departments (Morrisey et al 2009; Zeldis et al. 2011a,b). Notably an ongoing Ministry for Primary Industries Biodiversity project (ZBD2010-42) is seeking to address the following two objectives:

1. Prepare an online inventory of repeated biological and abiotic marine observations/datasets in New Zealand.
2. Review, evaluate fitness for purpose, and identify gaps in the utility and interoperability of these datasets for inclusion in a Marine Environmental Monitoring Programme (MEMP) from both science and policy perspectives.

Therefore any attempts to standardise monitoring datasets for aquaculture should try to learn from the experience of or recommendations from this project.

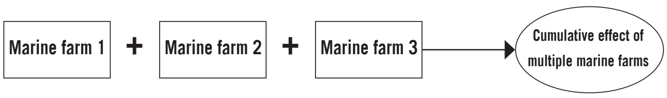
12.2 Summary of main cumulative effects to consider

By definition, multiple stressors associated with aquaculture (e.g. nutrient enrichment and organic deposition on the seabed) as well as other anthropogenic activities contribute to a number of potential cumulative environmental effects (Figure 12.2). More specific to aquaculture, these include additive effects of multiple farms (e.g. nutrient emissions to the water column; see Chapter 2) and also the potential synergistic effects of different stressors (e.g. potential for combined effects of organic enrichment and chemical additives on soft sediment communities). The schematics in Figure 12.2 illustrate mainly the spatial dimension to cumulative environmental effects; however, it is equally important to consider time as described in some of the examples.

Figure 12.2: Examples of the ways in which cumulative environmental effects could arise as a result of aquaculture development.

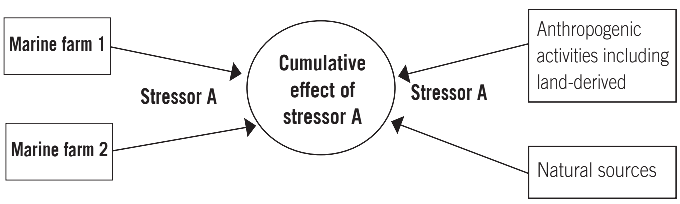
A. Additive effect of increasing numbers of marine farms

There are many examples in this category, for instance the additive effect of multiple local scale benthic footprints; incremental depletion of phytoplankton as a result of shellfish culture; or spread of pests/diseases among farms that leads to multiple reservoir populations.



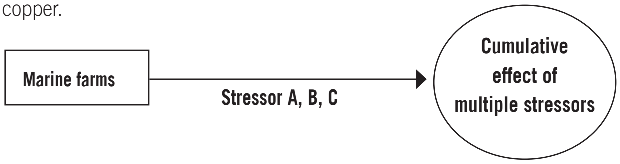
B. Additive effect of a single stressor from multiple sources in addition to marine farms

An example would be where dissolved nitrogen from marine farms adds to that from multiple diffuse and point sources leading to harmful algal blooms. These other sources could be both natural and anthropogenic, including oceanic and atmospheric inputs, consented point sources, and indirect or riverine inputs of land-derived nitrogen. In addition to ongoing inputs, there may also be those from episodic events such as rainfall or ocean upwelling.



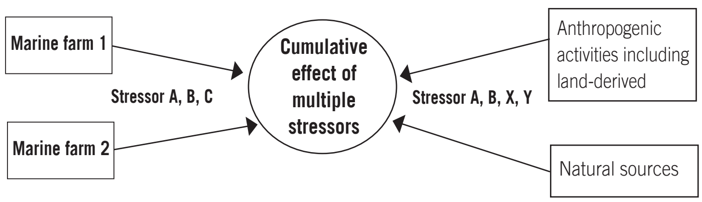
C. Additive and synergistic effects of multiple stressors from a single source

An example would be where fish farm faeces and feed leads to organic enrichment of the seabed, and is added to by potentially ecotoxic effects from copper that is used in antifouling. In that case, high levels of organic matter and sulfides in the sediment are likely to reduce the bioavailability and toxicity of the copper.



D. Additive and synergistic effects of multiple stressors from multiple sources

This situation reflects a blend of the issues above. For instance, in Tasman Bay, the benthic effects of recently-developed mussel farms will reflect multiple farm-derived stressors (e.g. fine sediments and oganic matter), effects of riverine fine sediments, high background concentrations of trace metals derived from natural sources in the catchment, and historic and ongoing disturbance from fishing.



12.2.1 Cumulative effects on the benthos and wider ecosystem

As aquaculture farms increase in number and variety within a region, there is the possibility of changes in the abundances and composition of organisms in the wider ecosystem (e.g. changes in fish or benthic invertebrate populations) due to the alteration of habitat, changes in fishing pressure, and changes in food availability (e.g. see Chapters 3 and 5). For instance, the drop off of mussels, shells and biofouling organisms onto the seabed beneath mussel farms, which leads to the creation of reef-like habitat, can alter the composition and abundance of benthic organisms beneath farms (see Chapter 3). This may be considered a relatively low-level impact on the environment at the local scale; however, high densities of mussel farms, such as the ribbon-like developments in the Marlborough Sounds, could lead to additive (cumulative) effects on the wider ecosystem due to alteration of a larger proportion of the benthos. There is also the potential for changes to habitats and/or migration routes of higher-order organisms such as mammals or seabirds (see Chapter 5 and 6). In the case of farm structures, aquaculture involving numerous farms situated along the coast could also have cumulative effects on nearshore currents and waves, which in turn could affect important processes (e.g. larval transport, nutrient exchange) along the shoreline (see Chapter 11).

12.2.2 Cumulative biosecurity risks

In New Zealand, a cumulative effects issue of particular importance relates to biosecurity risk, which in the case of Harmful Algal Blooms (HAB) is also linked to cumulative effects of nutrient enrichment. As aquaculture development intensifies (no matter what the type), there is likely to be an increase in man-made structures and boat traffic, thereby increasing the risk of invasion and establishment of pests. Cumulative degradation of the marine environment from multiple stressors compromises habitat quality and could enhance biosecurity risks by increasing productivity and proliferation of pest species such as invasive macroalage (e.g. *Undaria*) and invertebrates (e.g. the bivalve *Theora lubrica* and tunicate *Styela clava*) that thrive on the benthos under conditions of high organic enrichment (see Chapter 7). Whilst biosecurity issues are typically considered a primary risk to consider for aquaculture developments, good planning and farm practices within regional frameworks can greatly reduce the potential for biosecurity risks. Chapter 7 provides comprehensive information on methods for minimising biosecurity risk that are applicable to wider, regional scales.

12.2.3 Cumulative changes in nutrients and eutrophication

The above examples of cumulative effects may be important to consider on a case-by-case or regional basis; however, limited resources and uncertainty in understanding all of the potentially complex interactions between aquaculture and the environment necessitates the need to focus on those aspects of aquaculture most likely to contribute to cumulative environmental change. With this in mind, increasing emphasis has been placed on assessing the contribution of aquaculture to cumulative changes in nutrient conditions and primary production, and in turn the carry-on effects on the wider ecosystem (see Hargrave 2005; Volkman et al. 2009 and chapters therein).

All forms of aquaculture addressed in this report contribute to these wider ecosystem effects, whether through nutrient emissions to the water column and seabed through feed-added aquaculture, or the net extraction of plankton (filter-feeding bivalves) and nutrients (nutrient uptake by macroalgae) from the water column. The following section focuses on the potential contribution of nutrient additions from feed-added aquaculture to cumulative effects associated with eutrophication (excessive nutrient enrichment and accelerated primary production). Also discussed below is the role of other types of aquaculture (bivalves, seaweeds) in mitigating these eutrophication-related effects, as well as the potential for different types of cumulative effects that could arise from intensive development of these extractive forms of aquaculture (e.g. oligotrophication – a reduction in nutrient enrichment and levels of primary production). In most cases, the potential contribution of different types of aquaculture to the above cumulative effects will need to be considered together, since both forms of aquaculture are likely to co-occur within the same water bodies and therefore contribute to wider-ecosystem conditions.

12.2.4 Cumulative effects related to nutrient changes and their significance

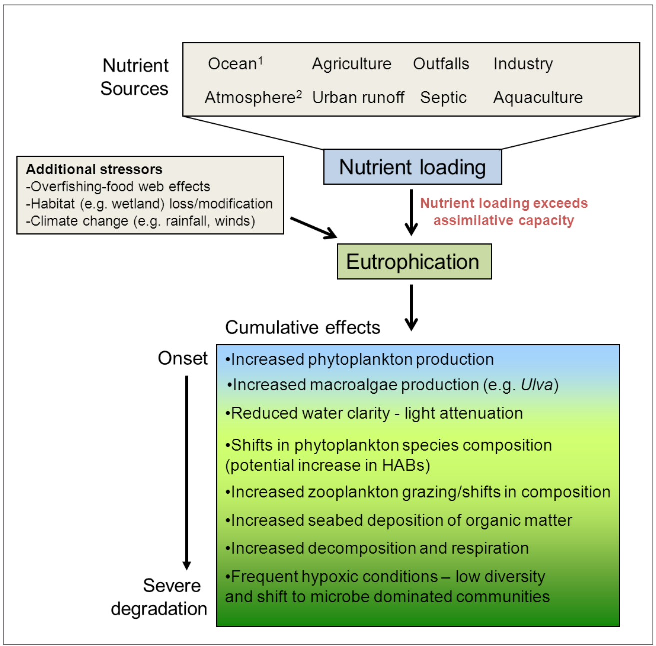
Table 12.1: Cumulative effects associated with nutrient emissions from feed-added aquaculture.

Description of effect(s)	Nutrient emissions (both dissolved and particulate forms) into the water column and onto the seabed contribute to cumulative nutrient loading occurring in the wider environment that may exceed an ecosystem's capacity to assimilate the nutrients. Effects could range from subtle increases in phytoplankton production to more advanced effects such as far-field organic accumulation on the seabed coupled with increased respiration and subsequent low oxygen levels. Nature and extent of effects could vary in time and space depending on a number of factors, such as season, site characteristics, and surrounding developments. Over time, factors such as changes in upstream land use, habitat loss/modification along rivers and coastal margins, fishing and climate change may also contribute to eutrophication of coastal waters.
Spatial scale	*Bay-wide* and *regional* scales.
Duration	Unknown, but potentially *long term*. Contribution of feed-added aquaculture toward cumulative water-column effects are likely to be reversible in a relatively short time frame (days to weeks) due to tidal flushing, whereas benthic effects, such as those associated with far field organic accumulation, may take longer to recover depending on the level of modification to the seabed (see Chapter 3).
Management options	• Setting of conservative limits for nutrient loading from all potential sources based on knowledge (including modelled predictions) of likely carrying capacity of the receiving environment. • Informed spatial planning and site selection to minimise effects. In multiple farm situations, modelling can assist in understanding the spatial distribution of effects under various development scenarios. • Staged development in the presence of long-term regional monitoring of background conditions and environmental change (SoE monitoring).
Knowledge gaps	• Baseline conditions and current level of cumulative effects from past and existing developments and activities (including land based) are not well documented or monitored in the coastal environment. • Capacity for coastal environments to assimilate nutrient loading remains unknown in most regions. • Nutrient inputs to the marine environment from land-derived diffuse (non-point) sources, and natural oceanic sources (and sinks such as denitrification and burial) are not well quantified. • Bathymetric and hydrodynamic data is needed for all regions supporting aquaculture, as this provides the basis for understanding waste dispersion and assimilation.

* Italicised text in this table is defined in chapter 1 – Introduction.

As identified overseas, cumulative effects of particular concern with expansion of feed-added aquaculture relate to nutrient loading and the potential risk of eutrophication (SEPA 2000; Hargrave 2005; Diaz et al. 2012). Eutrophication is the process where excessive nutrient inputs to a water body result in accelerated primary production (phytoplankton and macroalgae growth) and flow-on effects to the wider environment such as reduced water clarity, physical smothering of biota, or extreme reductions in DO because of microbial decay (Figure 12.3; Degobbis 1989; Cloern 2001; Paerl 2006). On a global scale, runoff from land-based agriculture has long been identified as the primary driver of intense eutrophication of coastal environments and an increasing number of hypoxic (low oxygen) zones (Diaz et al. 2012). With increasing growth of aquaculture, feed-added forms of aquaculture have been singled out as an emerging contributor to nutrient enrichment and cumulative effects associated with coastal eutrophication (Diaz et al. 2012).

Figure 12.3: Schematic of the eutrophication process.



Note: Whereby nutrient loading from multiple sources combined with additional stressors and natural characteristics of receiving waters leads to a range of cumulative effects once an ecosystem's capacity to assimilate nutrients is exceeded (i.e. measureable changes occur beyond the envelope of natural variability). [1]Ocean sources to coastal waters include dissolved nutrients through breakdown of organic matter, nitrification, and onwelling/upwelling of nutrient rich deeper waters. [2]Atmospheric deposition of nutrients from fossil fuel combustion, agricultural fertilizers and livestock operations can also significantly contribute to nutrients in coastal waters (see Diaz et al. 2012).

Nutrients of varying particulate and dissolved organic and inorganic forms are added to the environment as a result of feed-added aquaculture. Particulate organic nitrogen (PON) and phosphorus (POP) are primarily deposited onto the seabed as fish faeces but also as waste feed pellets and particles. The effects of this organic enrichment to benthic communities in close proximity to finfish farms are generally well understood (see Chapter 3). As described in Chapter 3, far-field deposition of particulate matter from feed-added aquaculture along with other sources of particulate organic nutrients (outfalls, agricultural runoff) may further contribute to cumulative enrichment of the benthos. As this organic material is broken down, dissolved forms of organic and inorganic nutrients (as well as toxic hydrogen sulphide under anoxic conditions) may in turn be released into the water column through advection and during resuspension of sediments.

Through feeding, the farmed fish excrete dissolved inorganic nutrients such as ammonium (NH4). Smaller particles of feed in the water column (through the addition of feed and/or via resuspension) can be consumed by other organisms such as zooplankton and shellfish, which in turn contributes to the dissolved nutrient pool. The dissolved inorganic nutrients from feed-added aquaculture combined with other sources of nutrient inputs fuel growth of phytoplankton, macroalgae and some bacteria. In New Zealand's temperate waters, nitrogen is likely to be the nutrient potentially limiting phytoplankton growth under certain conditions (e.g. when nitrogen concentrations are generally low and light is plentiful; MacKenzie 2004; Howarth & Marino 2006); however, nutrients such as silica can also play an important role in limiting growth of phytoplankton such as diatoms. Complicating matters is the fact that emissions of nutrients from finfish farms are only one source of nutrients in the marine environment, and, like other sources, their inputs vary over time. Olsen et al. (2008) observed that nitrogen excretion from a typical finfish farm in Norway may be approximately double the annual mean over the summer months (and therefore half the annual mean during winter months). This is consistent with salmon farms in the Marlborough Sounds, where feed levels increase by about 50 percent during summer months, which is also the period of greatest light availability for primary production.

As introduced in Chapter 2, the risk of exceeding the assimilative capacity and accelerating eutrophication will be dictated by the physical characteristics of a region, such as retention time, water depth and ambient nutrient concentrations, combined with the intensity and types of existing and planned aquaculture and upstream land-based developments. Coastal development and sedimentation that leads to loss and modification of habitats (wetlands, seagrass meadows) and organisms (shellfish) that serve to process and filter nutrients, further contribute to accelerating the eutrophication process and subsequent cumulative effects (see Figure 1; and McGlathery et al. 2007). Although nutrient loading from multiple sources (both natural and anthropogenic) is traditionally identified as the primary driver of eutrophication, other 'top-down' stressors, such as fishing, can also make a significant contribution to the eutrophication process through indirect effects of altered food webs (e.g. Heck & Valentine 2007).

Additional stressors, such as altered food webs from fishing, loss of coastal margin wetlands, loss of key species (e.g. natural populations of filter feeding bivalves) as well as factors influenced by climate change potentially contribute to the overall process. The opposite of eutrophication, known as oligotrophication, could theoretically occur in cases where high densities of shellfish farms and/or cultures of macroalgae result in a net reduction of nutrients below natural levels.

The intensity of different types of aquaculture and other anthropogenic activities impacting on coastal waters, combined with a region's physical and biological characteristics, will dictate the nature and extent of cumulative effects that potentially could arise from the addition of nutrients from feed-added aquaculture. Due to uncertainty around the cumulative effects of multiple nutrient inputs in New Zealand's coastal environments, it is difficult to adaptively manage any one activity in response to changes occurring in the wider environment. Hence, a precautionary approach utilising a number of tools (such as modelling and monitoring) is warranted in developing feed-added aquaculture.

12.2.5 Summary of cumulative effects from extractive forms of aquaculture

Table 12.2: Cumulative effects associated with extractive forms of aquaculture, such as the farming of filter-feeding mussels and oysters.

Description of effect(s)	Multiple shellfish (mussels, oysters) farms within a region collectively contribute to the extraction of plankton from the water column, and its conversion to particulate matter deposited on the benthos. The additive effects of multiple farms could potentially lead to cumulative ecological effects on the wider ecosystem, such as oligotrophication, or perhaps changes in plankton abundance/composition and, in turn, carry-on effects on the food web. Farming of macroalgae could add to the oligotrophic process by removing dissolved nutrients from the water column.
Spatial scale	*Bay-wide* and *regional* scales.
Duration	Potentially *long term* – as long as the farms are present in densities exceeding the capacity for the region to sustain production of shellfish and/or macroalgae without ecologically significant changes in plankton and/or nutrient concentrations. Cumulative water-column effects are likely to be reversible in a relatively short time frame (days to weeks), whereas wider ecosystem effects such as shifts in benthic community structure and function would probably take longer to recover depending on the level of modification (about 1–10 years).
Management options	• Setting of conservative limits for development based on knowledge (including modelled predictions) of likely carrying capacity of growing waters, which would be influenced by characteristics such as flushing times, natural levels of primary production, natural populations of filter feeders, and anthropogenic loading of nutrients. • Informed spatial planning and site selection to minimise effects. In multiple farm situations, modelling can assist in understanding the spatial distribution of effects under various development scenarios. • Staged development in the presence of long-term regional monitoring of background conditions and environmental change (SoE monitoring).
Knowledge gaps	• Baseline conditions and current level of cumulative effects from past and existing developments are not well documented or monitored in the coastal environment. • Historic trends in the distribution and abundance of natural shellfish populations and their role in the ecology of coastal environments is not well documented/understood. To some extent, farmed shellfish may restore ecosystem functions that have been compromised through the depletion of natural shellfish populations.

* Italicised text in this table is defined in chapter 1 – Introduction.

Through filter feeding, farmed shellfish have the potential to remove large amounts of plankton from the water column and convert it to shellfish biomass that is harvested (removed from the environment), or into dissolved and particulate waste products that are released into the water column and/or onto the seabed. Filter feeding by bivalves also has the potential to alter size distribution and species composition of plankton (see Chapter 2). There is compelling evidence that bivalve aquaculture can affect nutrient cycling and the quantity and quality of food (plankton) across a range of spatial scales from local to system-wide (Prins et al. 1998; Cerco & Noel 2007; Coen et al. 2007). In turn, these processes could affect the quantity and quality of food available to other consumers (Prins et al. 1998; Dupuy et al. 2000; Pietros & Rice 2003; Leguerrier et al. 2004), with consequences for local populations of higher trophic level organisms such as fish.

In regions where numerous farms with high-density cultures occur, there is the potential risk of exceeding the region's capacity to sustain high shellfish production and the wider ecosystem itself. An example is Pelorus Sound, where decreases of about 25 percent in green-lipped mussel yields between 1999 and 2002 raised questions around the concept of carrying capacity (Zeldis et al. 2008). The observed reduction in yields coincided with increased demand for water space for shellfish culture, which in turn led to research efforts directed toward the question of what level of culture is sustainable (i.e. the carrying capacity) in the growing areas. Although low production yields occurred during some years, the reductions were attributed to climatic forcing conditions and inter-annual variability in phytoplankton biomass over multi-year time scales (Zeldis et al. 2008).

The above example suggests that some shellfish aquaculture regions may be farmed close to sustainable production limits during years of naturally low primary production. Ecological carrying capacity limits are likely to be lower than production limits (Jiang & Gibbs 2005), so it follows that ecological carrying capacity may periodically be exceeded by the current level of culture in some areas. However, it is also possible that shellfish growth (and production) is reduced during low productivity years irrespective of farming intensity, and this is one of the difficulties with attributing lower production to an unsustainable farming intensity. The potential for exceeding carrying capacity is invariably situation-specific and temporally variable due to the influence of environmental factors operating from tidal time scales to longer term climatic events such as El Niño Southern Oscillation cycles (Dame & Prins 1998; Prins et al. 1998; Zeldis et al. 2008).

The potential for shellfish aquaculture to contribute to cumulative effects in the marine environment will be dependent on the size of the culture (including density of farms) and environmental characteristics of the area being farmed (e.g. hydrodynamics, phytoplankton biomass, anthropogenic nutrient inputs etc). Using "sustainability performance indicators", Gibbs (2007) suggests that the retention (flushing) time for a water body should not exceed 5 percent of the clearance time (filtering efficiency) of farmed mussels in order to minimise cumulative effects on the wider ecosystem. Such assessments can be further informed through the use of models. Application of food web models assists in estimating and forecasting the range of possible cumulative effects to higher trophic levels that would otherwise be too difficult to quantify based on field measurements. For example, the ECOPATH steady-state mass balance model (Christensen et al. 2000) was applied to assess the potential of Tasman Bay for mussel aquaculture development (Jiang & Gibbs 2005). The model was used to determine an ecologically sustainable level of mussel biomass beyond which higher trophic levels of the ecosystem might be affected through competition for food resources. Their results indicated that significant ecosystem energy flow changes occurred at mussel biomass levels less than 20 percent of a mussel dominated ecosystem, thus implying that ecological carrying capacity limits may be much lower than production carrying capacity limits.

Models have been used to assist in understanding the cumulative effects of mussel farms in Pelorus Sound (Ross et al. 1999; Inglis et al. 2000, Zeldis et al. 2008), Bay of Plenty (Longdill et al. 2006) and the Coromandel (Broekhuizen et al. 2002; Stenton-Dozey et al. 2008). There has also been considerable research into food depletion and modelling of ecological carrying capacity for oyster culture (Ball et al. 1997; Bacher et al. 1998; Ferreira et al. 1998) as well as for other bivalves and polyculture systems (Carver & Mallet 1990; Prins et al. 1998; Smaal et al. 1998; Gibbs et al. 2002; Nunes et al. 2003). Typically, this work has focused on phytoplankton depletion and maximum production capacity within growing regions.

Simple modelling techniques limit the findings to a broad, bay-wide scale assessment of ecological carrying capacity and do not incorporate feedback mechanisms such as changes to the flushing regimes induced by structures (Grant & Bacher 2001; Plew et al. 2005) or far-field nutrient enhancement and increased phytoplankton growth (Gibbs et al. 1992). Furthermore, literature in this field primarily addresses the role of natural or cultivated bivalve populations, whereas the filter-feeding activities of fouling organisms and other biota associated with shellfish cultures can also be functionally important (Mazouni et al. 2001; Mazouni 2004; Decottignies et al. 2007).

Spatial modelling tools offer a way of estimating the extent to which the cumulative effects of mussel farming may be approaching ecological carrying capacity on "bay-wide" and "regional" scales. However, knowledge gaps are still evident in these models; particularly in the biological aspects (e.g. feeding behaviour and growth of the shellfish) which are still areas of active research. Long-term monitoring of the wider ecosystem is required to validate and improve models and to assess wider cumulative environmental change.

12.3 Management and mitigation of cumulative effects

12.3.1 Managing cumulative effects

The mitigation and management of cumulative effects is difficult for any one industry to address. Aquaculture is only one of many human activities potentially contributing to cumulative effects in the marine environment. In the case of cumulative effects related to eutrophication, there is currently a very limited scientific understanding of the transport, fate and ecological consequences of nutrient loading from different sources and, in turn, how they cumulatively affect marine ecosystems (Olsen & Olsen 2008). Addressing cumulative effects to achieve sustainability ultimately requires regional approaches to managing developments and activities in a holistic, ecosystem-based manner (Dubé 2003; Crain et al. 2008).

Spatial planning can assist in facilitating ecosystem-based management (EBM). As described in the earlier chapters, appropriate site selection can greatly reduce environmental effects of aquaculture developments. On a regional level, and in order to address potential cumulative effects, aquaculture effects should be considered within the context of ecosystem-based plans that include existing and planned developments (land-based agriculture, aquaculture, Marine Protected Areas (MPAs) and activities (shipping, fishing)) that collectively affect the marine environment. An EBM approach that incorporates coastal catchments and the influence of river plumes and runoff must be used since land-based stressors ultimately contribute to the overall conditions of coastal waters where aquaculture is occurring. Tools and approaches for understanding and quantifying cumulative effects in the marine environment at regional to global scales have only recently emerged (e.g. Halpern et al. 2008; 2009). In New Zealand, research in North Island estuaries is also underway for the purpose of assessing cumulative effects from multiple stressors, and in particular, sedimentation (NIWA led research in Porirua and Kaipara Harbours). In addition, assessments of the cumulative effects of developing finfish aquaculture in areas currently used for farming shellfish have been carried out in order to inform regional-based coastal plans (Zeldis et al. 2011a,b).

Although there are mechanisms that enable an EBM approach within New Zealand legislation, actual implementation of EBM practices that consider land-sea connections and address cumulative effects in the coastal receiving environment is not common. Currently, there is limited long-term data on coastal environmental conditions; this, combined with a high level of natural variability in the marine environment, precludes highly adaptive approaches for managing cumulative effects. In the absence of over-arching EBM programmes and a robust scientific base for adaptive management in response to cumulative effects, a precautionary approach is warranted in future developments of feed-added aquaculture. Using a precautionary approach, development should be conducted in a staged manner based on conservative limits of expansion. Staging provides a means of reducing environmental risk, but also helps to ensure that the infrastructure, expertise and institutional arrangements are available to support the pace of developments and address cumulative effects in the most informed manner.

Important tools and components of a precautionary approach include:

1. The use of models and existing data to gauge limits to development[1] within the context of a region's assimilation capacity (i.e. ecological carrying capacity).
2. Establishment of wider-ecosystem, long-term monitoring programmes that include establishment of baseline conditions of a region and adoption of limits of acceptable change.
3. Mitigation of effects through continual improvement of on-farm practices, potentially including improved feed technologies and the use of Integrated Multitrophic Aquaculture (IMTA).
4. Targeted monitoring and research for validating and improving accuracy of predictive models and understanding the role of feed-added aquaculture in driving cumulative effects.

Overseas examples of precautionary approaches include the M-O-M system (Modelling–Ongrowing fish farms–Monitoring),

[1]In some cases, areas may not be suitable for any development of aquaculture.

which has been undertaken in Norway to provide information for adaptive management of salmon farming (Ervick et al. 1997; Hansen et al. 2001). Tasmania has recently gone through an in-depth assessment of the carrying capacity of finfish aquaculture in the Huon Estuary and D'Entrecasteaux Channel (Volkmann et al. 2009). A particular focus of their research has been the use of extensive modelling to understand and isolate the potential effects of nutrient loading from expansion of finfish aquaculture on the wider marine environment (Wild-Allen et al. 2009).

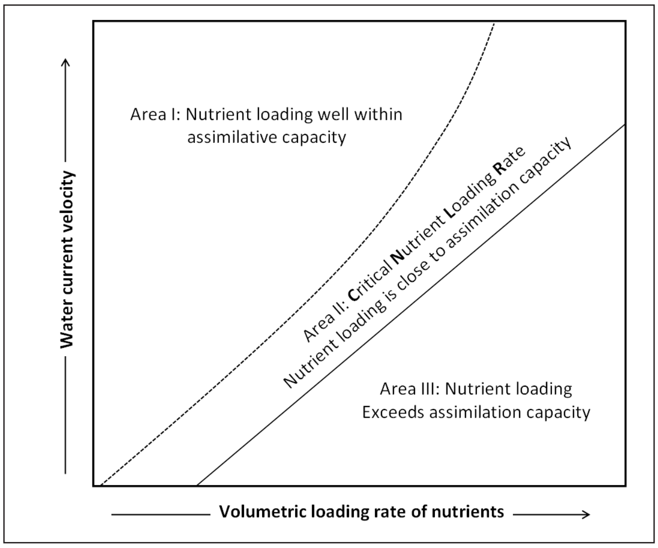
12.3.2 Carrying capacity and setting limits

A precautionary approach necessitates establishment of conservative thresholds or limits to minimise risks and extent of cumulative effects. To minimise risk of eutrophication, setting a limit (or cap) on nutrient loads to a coastal receiving environment would be similar to the approach taken with restoring the Rotorua Lakes. In this situation, levels of nutrient loading under various land-use scenarios are linked with changes in the lake's trophic conditions over time (www.lernz.co.nz). Such programmes greatly assist in land-use management and setting future targets. A similar example includes determining ecological thresholds for downstream estuaries in response to cumulative stressors (mainly associated with sedimentation), which are then used to set limits to contaminant loading from upstream activities (NIWA led research in Porirua and Kaipara Harbours).

In the case of feed-added aquaculture, the same approach can be taken with regard to cumulative nutrient enrichment. The main task is to determine levels of nutrient loading (from all potential sources) that can be reached without exceeding a system's capacity to assimilate nutrients. In broader terms, this falls within the concept of ecological carrying capacity, which has been defined for shellfish aquaculture by Gibbs (2007) as the level of culture that can be supported without leading to significant changes to ecological processes, species, population or communities in the growing environment. This definition also applies to finfish aquaculture, and indeed shellfish and finfish aquaculture must ultimately be considered together where they are co-occurring to fully understand a region's ecological carrying capacity (see for example Zeldis et al. 2011b).

Determining ecological carrying capacity for growing waters under its broad definition is difficult because there is no strong foundation for defining limits within a marine ecosystem based on complex ecological processes. To simplify matters and focus on the primary issue of nutrients, estimates of the potential for nutrient enrichment of the water column from multiple sources can be referenced to a critical nutrient loading rate (CNLR), which is defined by Olsen et al. (2008) as a critical nutrient loading rate which cannot be exceeded without loss of ecosystem integrity. Northern Hemisphere experiments for assessing ecosystem response to changes in nutrient loading have shown that CNLRs exist in marine systems where aquaculture is taking place (Duarte et al. 2000; Olsen et al. 2006). As shown in the conceptual graph in Figure 12.4, the CNLR is not only influenced by the rate of nutrient loading, but also by the physical characteristics of an area (current velocity, flushing, water depth).

Figure 12.4: Conceptual relationship showing the ability for the water column to assimilate nutrients as a function of nutrient inputs and water current velocity (from Olsen & Olsen (2008).



Numerical models can assist in estimating assimilative capacities, understanding potential changes in the ecosystem from various nutrient loading scenarios, and setting limits to development. Nutrient mass-balance models are the simplest to construct and can provide guidance on nutrient loading rates to a region under various scenarios and for gauging proximity to conservative CNLRs (Olsen & Olsen 2008). Mass-balance models represent the system as a series of "boxes" with nutrient inputs and outputs; data on water-column salinity and nutrients can be incorporated into the models to account for tidal and estuarine exchange (flushing). They can also be further expanded to include phytoplankton growth models, as was done to assess effects of salmon farming in Big Glory Bay (Pridmore & Rutherford 1992). The mass-balance approach has facilitated development of system-wide nutrient budgets and estimates of carrying capacity for feed-added aquaculture in Golden and Tasman Bays (Zeldis 2008b; Zeldis et al. 2011a, b) and the Firth of Thames (Zeldis 2008a; Zeldis et al. 2010).

The capacity for a system to assimilate nutrient inputs is a complex function of a system's biotic and abiotic characteristics and includes such factors as flushing rate, light and temperature regime, nutrient cycling processes (e.g. denitrification rates), grazing pressure (Tett & Edwards 2002) and native epibiota composition and biomass (e.g. macrophytes). Nutrient mass-balance models do not capture this complexity and do not provide spatial information on nutrient loading and transport, which is required to better understand and predict the cumulative effects of aquaculture (Ervick et al. 1997; Volkmann et al. 2009). More advanced models that couple three-dimensional (3D) hydrodynamic models with nutrients modelled as passive tracers can be used to generate finer-spatial resolution estimates of potential changes in nutrient concentrations and primary production. This approach reduces transport and mixing uncertainty from the simpler mass-balance models and allows for the cumulative effects of multiple farms to be visualised at local to region-wide scales.

Spatially explicit hydrodynamic models in turn provide a foundation for biophysical and biogeochemical models, which provide simulations of biological changes, such as changes in phytoplankton biomass, or shifts in phytoplankton composition as a result of changes in nutrient ratios. These types of models have been used to assess expansion of feed-added aquaculture in the Firth of Thames and Tasman Bay (Zeldis et al. 2010; 2011a, b). In a ten year study, Wild-Allen et al. (2009) used a calibrated biogeochemical model of a southern Tasmanian estuary to predict the proportion of the region that would shift to a more nourished trophic state from nutrient emissions from expanding finfish aquaculture. In order to address cumulative effects from nutrient loading, all models must consider sources of nutrients from coastal catchments. Tools such as CLUES (Catchment Land Use for Environmental Sustainability; available on the NIWA website) can feed into this process, and more advanced layering of models can provide for more complex forecasting that accounts for land-sea interactions (e.g. Nobre et al. 2012).

As discussed in Section 12.2.5, models have also been used to assist in understanding the cumulative effects of mussel farms in several regions around New Zealand. Modelling studies have primarily focused on carrying capacity in terms of sustaining farm production, rather than ecological carrying capacity. Application of food web models assist in estimating and forecasting the range of possible cumulative effects to higher trophic levels that would otherwise be too difficult to quantify based on field measurements. For example, the ECOPATH steady-state mass balance model (Christensen et. al. 2000) was applied to assess the potential of Tasman Bay for mussel aquaculture development (Jiang & Gibbs 2005). The model was used to determine an ecologically sustainable level of mussel biomass beyond which higher trophic levels of the ecosystem might be affected through competition for food resources.

12.3.3 Monitoring wider ecosystem health

As highlighted above, the complexity and limited understanding of cumulative effects occurring in the marine environment necessitates a precautionary approach to coastal development and sufficient monitoring of wider ecosystem health. Stressors from multiple developments and activities impact on the marine environment, in complex, synergistic ways (Crain et al. 2008). The monitoring of such effects is therefore best carried out within an overarching programme aimed at assessing the status of the wider environment (i.e. SoE monitoring by a regional council or coordinated group). Accessibility, consistency and coordination of datasets from regulatory monitoring programmes (i.e. compilation of monitoring data from multiple farms) and alignment with long-running shellfish sanitation programmes that focus on water quality and harmful algae (e.g. the Marlborough Sounds Shellfish Quality Programme-MSQP) would further strengthen the ability to assess the state of the wider environment over time and space.

The cumulative effects of eutrophication can occur gradually over long time periods (Armitage et al. 2011) and cascading effects to the environment (i.e. shifts in benthic communites) can last for decades (Herbert & Fourqurean 2008). Therefore, establishment of long time-series of environmental indicators is critical to establishing appropriate baselines and for understanding the variability of the wider system in response to drivers operating over long time scales (e.g. seasonal shifts in nutrient inputs versus climate/ocean processes). Possible components of long-term monitoring include sector contributions (including agriculture industries) toward region-wide field sampling programmes (e.g. expansion of the existing sanitation/seafood safety programmes) and establishment of permanent observation platforms with high frequency sampling capabilities.

Permanently established platforms with sensor arrays enable the collection of robust, time-series data for multiple purposes, including regional and national SoE monitoring, MSQP monitoring of water quality and phytoplankton blooms, validation of models, and research. Such monitoring platforms have provided valuable long-term datasets in the Firth of Thames (Zeldis et al. 2010) and more recently have been established in Nelson Bays (e.g. TASCAM buoy; www.cawthron.org.nz). A cost-effective tool for wide-scale monitoring includes the analysis of satellite imagery.

An important step in the design of a wider-ecosystem monitoring programme is the selection of appropriate indicators. In the development of finfish aquaculture in Tasmania, a range of stakeholders, including industry, were involved in the selection process, and they ranked potential indictors according to criteria such as sensitivity, applicability, correlation to actual environmental effects, cost effectiveness, social relevance, ease of measurement etc. (see Chapter 8 in Volkmann et al. 2009). Potential indicators include phytoplankton biomass and community metrics, frequency of algal blooms, concentration of dissolved oxygen and ammonium, organic enrichment of sediments and concentrations of bacteria linked with the remineralisation of the organic matter and production of hydrogen sulphide (King & Pushchak 2008; Volkmann et al. 2009). Composite indicators such as trophic state indicators (e.g. TRIX, Giovanardi & Vollenweider 2004) can combine results from several parameters into a single metric for comparison with other systems. Alternatively indicators may be based on links to values of the stakeholders in a region, such as water clarity and frequency of macroalgae blooms along shorelines and/or marine structures, or even industry data on mussel production which would reflect Sound-wide primary production (Zeldis et al. 2008). Stakeholder agreement on 'limits of acceptable change' provides an adaptive element to long-term monitoring of the wider ecosystem, and they can be linked back to a management action (Zeldis et al. 2005). For example, in the case of mussel farm development in the Firth of Thames, limits of acceptable change were agreed upon in order to avoid significant changes in the wider environment as the industry develops. Taking this approach, trigger values for indicators of phytoplankton depletion (e.g. chl a) monitored over time are linked to management actions (see Zeldis et al. 2005).

12.3.4 Mitigating cumulative effects

Using an ecosystem-based approach, efforts should be made to minimise nutrient inputs from all sources, with emphasis placed on those steps that lead to the greatest gains. As described above, knowledge of a system's carrying capacity and setting conservative limits to expansion are required to minimise eutrophication risk. Related to this are appropriate spatial planning and site selection of growing waters. A system's carrying capacity and its predisposition for eutrophication is correlated to factors such as water depth, current speeds, and flushing times of water bodies. These factors influence the degree of water column mixing and stratification, which in turn influences whether cumulative effects are more significant in the water column versus the benthos (McGlathery et al. 2007).

As covered in Chapter 2, the practice of Integrated Multi-Trophic Aquaculture (IMTA), can be an environmentally and economically efficient means of reducing nutrient emissions from feed-added aquaculture. Such practices may reduce the risks of eutrophication in the wider ecosystem. Due to transport processes and lag times in the response of plankton communities to increases in nutrients, it is possible that culture of both shellfish and finfish within the same water bodies and over larger spatial scales (e.g. Pelorus Sound) assists in mitigating effects of nutrient enrichment.

There is further potential to reduce nutrient emissions from feed-added aquaculture by decreasing the ratio between the amount of feed consumed by fish and the amount lost to the system (Feed conversion ratio: FCR). In addition, minimising nitrogen content within the feed itself can result in significant reductions in nutrient emissions. While addressing both of these aspects of feed, nutrient loading into the environment could be reduced significantly while maintaining the equivalent amount of production. Such solutions require further research and development under New Zealand conditions and among the different farmed species.

12.3.5 Knowledge gaps

Internationally, there is a very limited understanding of the cumulative effects of multiple stressors on marine ecosystems. A critical need for understanding these effects is having good information on existing environmental conditions, and access to long time-series data on indicators of these conditions from which to quantify and forecast changes occurring in the wider environment. Related to this is the need to obtain additional knowledge on the trophic status of New Zealand's coastal waters where aquaculture is underway or planned. While there is sufficient knowledge and data to classify most of New Zealand coastal waters in terms of their general trophic status, there may in some cases be limited time series data (e.g. nutrients, indicators of primary production) to describe the extent to which trophic conditions vary (spatially and temporally) within a given region.

Modelling has an important role to play in understanding, predicting and managing cumulative effects and New Zealand has access to extensive modelling capability; yet in most cases the uncertainty in model accuracy remains high due to insufficient field data for their calibration and validation. For example, underlying hydrodynamic models require sufficient time-series data on currents and water column stratification, while more advanced biogeochemical models require validated estimates of inputs (e.g. surface water, groundwater, marine) and losses (denitrification, burial rates) of nutrients more specific to New Zealand's coastal waters. Such information and data would assist in more accurately estimating the capacity for coastal systems to assimilate anthropogenic nutrients and quantifying contributions from different sources. As is the case for wider ecosystem monitoring, the collection of data useful for calibrating and validating more sophisticated models for assessing cumulative effects in the marine environment is likely to lie outside the scope of an individual farm consent applicant/holder, and would be best managed through a regional council or co-ordinated group.

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