Modelling and Technical Studies in Support of the Mid-West Aquaculture Development Zone

1051_009/1_Rev2

April 2016

Modelling and Technical Studies in Support of the Mid-West Aquaculture Development Zone

Prepared for

Department of Fisheries

Prepared by

BMT Oceanica Pty Ltd, BMT WBM and AED UWA

April 2016

Report No. 1051_009/1_Rev2

Client: Department of Fisheries

Document history

Distribution

Revision	Authors	Recipients	Organisation	No. copies & format	Date
A	G Shiell M Barry R Holmes D Paraska M Hipsey	C Hanson	BMT Oceanica	1 x docm	31/07/15
B	G Shiell M Barry R Holmes D Paraska M Hipsey	J Eyres	DoF	1 x docm	06/08/15
C	G Shiell M Barry R Holmes D Paraska M Hipsey	M. Bailey	BMT Oceanica	1 x docm	23/09/15
D	G Shiell M Barry R Holmes D Paraska M Hipsey	J Eyres L Caporn	DoF	1 x docm 1 x pdf	02/10/15
0	G Shiell M Barry R Holmes D Paraska M Hipsey	J Eyres L Caporn	DoF	1 x pdf	02/11/15
1	G Shiell M Barry R Holmes D Paraska M Hipsey	J Eyres	DoF	1 x pdf	06/11/15
2	G Shiell M Barry R Holmes D Paraska M Hipsey	J Eyres	DoF	1 x pdf	18/04/16

Review

Revision	Reviewer	Intent	Date
A	C Hanson	Technical and editorial review	06/08/15
B	J Eyres and L Caporn	Client review	28/08/15
C	M Bailey	Editorial	23/09/15
D	J Eyres and L Caporn	Client review	29/10/15
0	J Eyres and L Caporn	Client review	04/11/15
1	OEPA	Regulator review	17/02/16

Quality Assurance

BMT Oceanica Pty Ltd has prepared this report in accordance with our Health Safety Environment Quality Management System, certified to OHSAS 18001, AS/NZS 4801, ISO 14004 and ISO 9001: 2008.

Status

This report is 'Draft' until approved for final release, as indicated below by inclusion of signatures from: (i) the author and (ii) a Director of BMT Oceanica Pty Ltd or their authorised delegate. A Draft report may be issued for review with intent to generate a 'Final' version, but must not be used for any other purpose.

Approved for final release:

Glenn Shiell

Authors
Date: 18/4/16

Director (or delegate)
Date: 18/4/16

Disclaimer

This report has been prepared on behalf of and for the exclusive use of Department of Fisheries, and is subject to and issued in accordance with the agreed terms and scope between Department of Fisheries and BMT Oceanica Pty Ltd. BMT Oceanica Pty Ltd accepts no liability or responsibility for it in respect of any use of or reliance upon this report by any third party.

Copying this report without prior written consent of Department of Fisheries or BMT Oceanica Pty Ltd is not permitted.

Contents

List of Figures

List of Tables

List of Appendices

Executive Summary

Purpose

Risks associated with the Department of Fisheries (DoF) proposal to establish the Mid-west Aquaculture Development Zone (MWADZ) were assessed based on a number of technical studies, including the development and execution of an integrated environmental model. The purpose of this document is to summarise the findings of the technical studies, and to provide advice on the likely cumulative impacts of sea-cage operations on the marine environment under a range of operational scenarios. Results are presented particularly in the context of the key environmental factors identified in the Environmental Scoping Document (ESD). The findings of this Environmental Impact Assessment (EIA) feed into the broader Public Environmental Review (PER) for this project.

Methods and assumptions

Technical studies were supported by empirical and desktop procedures. Baseline water, sediment and metocean data were collected over a nine month period between May 2014 and February 2015, capturing seasonal changes in water and sediment chemistry, wave height and current speeds. Complementing the baseline assessment, single beam echo sounding and towed video methods were used to delineate key benthic habitat types and their relative proportions. The potential for impact on significant marine fauna, including marine mammals, turtles, sea-lions, finfish (sharks and rays), invertebrates and seabirds, was assessed via desktop reviews.

A key component of the assessment was to develop an integrated environmental model capable of resolving the effects of wastes on the marine environment, and the rate of environmental recovery following cessation and/or relocation of the proposed activities (fallowing). Three levels of impact; 'zone of high impact' (ZoHI), 'zone of moderate impact' (ZoMI) and 'zone of influence' (ZoI) were spatially delineated based on exceedances of predetermined environmental thresholds, following the guidance in Environmental Assessment Guideline 7 (EPA 2011). Thresholds were set differently in recognition of the diversity of receiving environments in the MWADZ. For 'sandy' habitats, thresholds were determined based on the biochemistry of the sediments and the rate at which they recovered following cessation of aquaculture activities. For the water column and mixed assemblage habitats, the impact potential was determined using a separate set of thresholds. Thresholds were developed for: nutrient enrichment, algal growth potential and oxygenation, potential for shading, smothering and stressors such as the mechanical interference, such as that produced by elevated levels of suspended particles. The latter thresholds were acute thresholds, and were based on the published literature (PIANC 2010) and the EPA's environmental criteria for high and moderate levels of ecological protection (EPA 2015).

Site description

The MWADZ is proposed to be established within the Fish Habitat Protection Area of the Houtman Abrolhos Islands. It consists of two areas: a northern area (2200 ha), located roughly halfway between the Easter and Pelsaert groups, and a southern area (800 ha), located immediately north of the Pelsaert group, for a total of 3000 ha. The waters of the MWADZ are deep (25-50 m), well flushed and experience high levels of water circulation and dispersion. Previous oceanographic work at the Easter Group islands indicated strong currents (i.e. between 2–5 cm/sec) and fast flushing times (i.e. from 0.5 to 1.5 days) in the shallow waters of the Easter Group lagoon. The MWADZ is located in more exposed waters between the Pelsaert and the Easter Group of islands, where flushing is likely higher than in the sheltered islands.

Baseline conditions

Results indicate that the waters inside the project area are clean and generally well mixed. Maximum and minimum water temperatures were achieved in autumn (23.5°C) and winter (20.8°C), respectively. Salinity and dissolved oxygen levels were fairly consistent through the water column with little evidence of stratification. The water was highly oxygenated at all times, achieving surface oxygen saturation levels between 96% and 99% and bottom oxygen saturation levels between 95% and 98%. Light attenuation in the MWADZ was lower (0.04–0.19 per m) than that obtained (1.2–1.8 per m) in the Kimberley Aquaculture Development Zone (KADZ), which is indicative of very clear water, with good light penetration. Water currents were variable, ranging between 5.8 and 14.4 cm/s. The MWADZ is an oligotrophic, or nutrient poor environment. Concentrations of ammonium (2.7 µg/L) and chlorophyll-a (0.43 µg/L) were similar to those found in Perth's coastal waters and lower than those recorded in the KADZ assessment (5.4 µg/L and 0.9 µg/L, respectively). Nitrite + Nitrate levels (12.9 µg/L) were higher than those recorded in Perth's coastal waters (6.5 µg/L) and in the KADZ (8.7 µg/L). Concentrations of inorganic nutrients and chlorophyll-a were seasonally variable, with higher concentrations in the cooler months.

The benthic environment consisted generally of a shallow layer of sand overlying rocky substrate, with mixed biotic assemblages. Higher current speeds in the northern area (northern 13.2-14.5 cm/s and southern 8.7-11 cm/s) were reflected in the tendency toward larger sediment grain sizes in the northern reaches of the MWADZ. Sediment conditions were variable, with seasonal fluctuations in nitrogen and total organic carbon with generally higher values in warmer months. Infaunal assemblages were diverse (10 phyla; 129 families), with communities dominated by polychaetes. Higher levels of infauna diversity and abundance were observed in the summer months.

Surveys indicated that much of the seafloor consisted of open sandy meadows and mixed biological assemblages, supporting macroalgae, rhodoliths, sessile invertebrates and some corals; however, all of the available data suggest that their presence may be itinerant given the observed differences between surveys. Habitats in the northern study area were more diverse relative to the southern area and comprised 59% bare sand and 34% mixed assemblages. Small patches of reef were present near the north-east boundary of the MWADZ but made up only 8% of the total habitat. The southern area by contrast comprised 96% bare sand and 5% mixed assemblage. Although ephemeral seagrass communities were observed in previous surveys of the MWADZ, no seagrasses were observed in the current assessment.

Impact assessment

Desktop assessments were undertaken to determine the likely impact of the proposal on marine mammals, seabirds and other significant fauna, including sharks, rays, other finfish and invertebrates. Several risks were identified including the potential for the sea-cages to act as a physical impediment to animal migration and water flow, a source of entanglement, an artificial food source, and as a significant artificial attractant and roosting area for seabirds. The risks were considered manageable through the use of best-practice infrastructure and management strategies. Examples of these included use of high-walled sea-cages (to limit access of sea lions), use of nets to exclude seabirds, and implementation of modern fish-feeding methods to both limit wastage and impede opportunistic feeding by sea-birds.

An integrated hydrodynamic, particle transport, water quality and sediment diagenesis model was used to simulate a total of six production scenarios (Table ES.1). Modelling scenarios were agreed in consultation with the DoF and the Aquaculture Industry Reference Group at a technical workshop held in October 2014. Scenarios were developed based on production of yellowtail kingfish (*Seriola lalandi*) using industry best-practice farming methods.

Table ES.1 Modelled production scenarios

Scenario No.	S1	S2	S3	S4	S5	S6
Total standing biomass (t)	15 000		24 000		30 000	
Standing biomass north (t)	10 000		16 000		20 000	
Standing biomass south (t)	5000		8000		10 000	
No. clusters south	3	2	3	2	3	2
No. clusters north	6	4	6	4	6	4

Note:
1. t = tonnes

The potential for impact and loss of benthic primary producing habitats (BPPH) was examined in the context of EAG 3, Category C. The assessment found that the proposal was unlikely to yield significant cumulative losses and the total cumulative loss would be restricted to <1%, which was within the Category C benchmark of 2%.

Integrated modelling examined the likely benthic footprints of the sea-cages under the range of scenarios in Table ES.1. The extent of benthic footprints was determined after two, three and five years production, and the extent of water quality impacts after one year of production. Benthic impacts were examined in the context of sediment organic enrichment and changes to sediment chemistry, with the level of impact determined by the recovery period during fallowing.

Deposition of fish faeces and waste feeds resulted in rapid changes to sediment oxygen and hydrogen sulphide concentrations beneath the sea-cages; however, the spatial extent and intensity of impacts varied significantly depending on the type and the length of the scenario modelled. Results suggested that the ZoHI would occupy 82-117 ha (S2-S1) to 139-177 ha (S6-S5) after 5 years production, but less after 3 (2-1 ha to 95-105 ha) and 2 years (0.2-0 ha to 88-91 ha) production.

Reductions in both the standing biomass and the length of production also reduced the extent of the ZoHI, as measured along the maximum radius down-current from the cage clusters. After 5 years continuous production, the ZoHI, extended to a maximum of 110 m and 70 m under S6 and S5, but less than that under other scenarios, and shorter production periods: in S4 for example, distances reduced to 60 m and 15 m after 3 and 2 years production respectively, and for S3, the distance reduced to 10 m after 3 years production. After 2 years production, the ZoHI in S3 did not breach the cage cluster perimeter.

Increasing the stocking density while maintaining standing biomass (i.e. stocking density S4 > S3; standing biomass S4 = S3) had the effect of reducing the total area occupied by the ZoHI across the zone. This effect was particularly strong after 5 years production, but less so after 3 and 2 years production. For the 24 000 t (S3-S4) and 30 000 t (S5-S6) scenarios, reducing the number of clusters from nine to six reduced the extent of the ZoHI by 15% and 22%, respectively. It was noted that while the spatial extent of the ZoHI was reduced, the effect was to increase the intensity of impacts under the sea-cages, thus extending the recovery time. Results confirmed that large standing biomasses (up to 5 000 t per sea-cage cluster (or 30 000 t spread across 6 clusters)) are achievable, while constraining the benthic impacts to relatively small areas.

Risks associated with dissolved inorganic nitrogen (DIN) and suspended particles were examined after one year of production. Suspended particles were examined in the context of smothering and interruption to filter feeding processes, and DIN in the context of algal growth potential, nutrient enrichment and shading. While modelling predicted no adverse effects to filter feeding processes, modelling predicted minor to moderate impacts (S4-S6) from smothering immediately under the sea-cages. Concentrations of DIN down-current of the sea-cages were predicted to increase with increasing biomass and increasing stocking density. However, the plumes were predicted to dissipate rapidly, with concentrations generally returning to levels commensurate with a high level of ecological protection inside the MWADZ boundary. Despite significant inputs of DIN to the system, there were no increases in chlorophyll-a or declines in light penetration attributable to fish-farming.

Conclusions

This assessment simulated the effects of finfish standing biomasses between 15 000 and 30 000 t, for periods of one year for water quality and mixed assemblages, and two, three and five years for sandy sediments. Under 30 000 t standing biomass, modelling predicted no adverse changes to water quality and only localised impacts to the sea-floor beneath the sea-cages. The most severe impacts, as represented by the ZoHI, were restricted to 110 m distance after 5 years production, and 55 m and 50 m distance after 3 and 2 years production, respectively. Further improvements were achieved by reducing the standing biomass to 24 000 (S4) under which the ZoHI was restricted to 15 m after 3 years production. Scenario 4 in particular demonstrated a capacity to maintain large volumes of finfish (4000 t per sea-cage cluster), while constraining the impacts (ZoHI) to localised areas.

Results presented here are equivalent to the 'most likely worst case' outcomes as required by the ESD for this project. The scenarios tested were designed to be (a) sufficient to support a viable finfish aquaculture industry and (b) be well within the critical assimilative capacity of the marine environment, based on an understanding of systems with similar flushing regimes and nutrient inputs. Based on this, it is recommended that 24 000 t standing biomass is set as an interim limit, pending further validation of the particle dispersion and sediment diagenesis models, using field data (sediment characteristics and water quality) collected in the first years of operation. It is also recommended that this limit is validated in the context of further metocean assessments, including the effect of significant storms, and the frequency of benthic ‘resetting’ events–both of which were not accounted for in this assessment.

1. Introduction

In late 2011, the Minister for Fisheries announced a funding package to enable establishment of two aquaculture development zones in Western Australia's (WA's) coastal waters. The Department of Fisheries (DoF) is managing the project, and is responsible for undertaking the environmental impact assessments (EIA) for zones in the Kimberley and Mid-West regions of the State.

The first of these zones, the Kimberley Aquaculture Development Zone (KADZ), was approved by the Minister for Environment on 12 May 2014 under Part IV of the *Environmental Protection (EP) Act 1986*, by way of Ministerial Statement 966. The 1993 ha KADZ, located in Cone Bay, has conditional approval to produce up to 20 000 tonnes (t) of marine finfish per year.

The second zone, the Mid-West Aquaculture Development Zone (hereafter the 'MWADZ'), is proposed to be established within the Fish Habitat Protection Area of the Houtman Abrolhos Islands (hereafter the 'Abrolhos'). The MWADZ consists of two areas: a northern area (2200 ha), located roughly halfway between the Easter and Pelsaert groups, and a southern area (800 ha), immediately north of the Pelsaert group (Figure 1-1).

The proposal to develop the MWADZ was referred to the Environmental Protection Authority (EPA) in May 2013 and the level of EIA was set at Public Environmental Review (PER), under Section 38 of the *EP Act 1986*. EIA is an orderly and systematic process for evaluating a proposal (including its alternatives) and its potential effects on the environment.

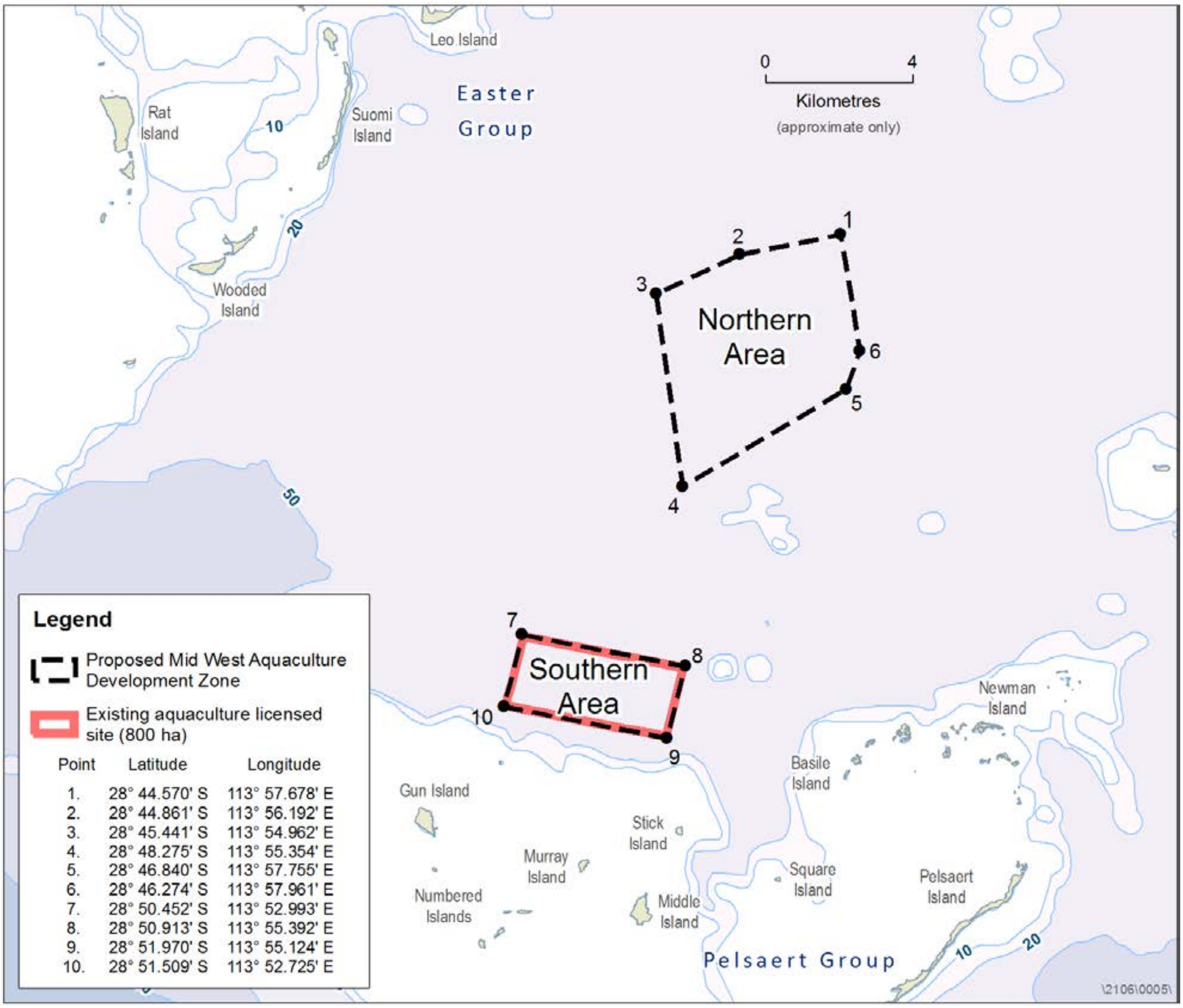
The scope of the PER is defined in the EPA-prepared environmental scoping document (ESD). A number of technical studies were required (Section 2) to assess the potential impacts of the MWADZ in the context of the key environmental factors outlined in Table 1.1. The technical studies included the development and execution of an integrated environmental model, and multiple desk top assessments.

Table 1.1 Key environmental factors and impacts identified in the Environmental Scoping Document

Key environmental factors	Key environmental impacts
• Hydrodynamics	• Alterations to hydrodynamics
• Marine water and sediment quality (including accumulation of trace contaminants)	• Degradation of marine water and sediment quality
• Marine flora and benthic primary producer habitat • Significant marine fauna • Marine benthic infauna and invertebrates	• Direct and indirect disturbance or loss of benthic communities and habitat • Direct and indirect impacts to key sensitive receptors • Impacts to marine environment and biota quality through release of pharmaceuticals, metals/metalloids and, or petroleum hydrocarbon • Direct and indirect impacts on significant marine fauna

Source: EPA (2013)

The purpose of this document is to summarise the findings of the technical studies, and to identify an upper aquaculture production level (tonnes of finfish) consistent with acceptable environmental impacts. Results are provided in the context of marine (benthic and open water) environments in and around the proposed MWADZ, and in the context of the greater Abrolhos region.



Point	Latitude	Longitude
1.	28° 44.570' S	113° 57.678' E
2.	28° 44.861' S	113° 56.192' E
3.	28° 45.441' S	113° 54.962' E
4.	28° 48.275' S	113° 55.354' E
5.	28° 46.840' S	113° 57.755' E
6.	28° 46.274' S	113° 57.961' E
7.	28° 50.452' S	113° 52.993' E
8.	28° 50.913' S	113° 55.392' E
9.	28° 51.970' S	113° 55.124' E
10.	28° 51.509' S	113° 52.725' E

Figure 1-1 **Location of the proposed mid-west aquaculture development MWADZ, showing the southern and northern areas**

2. Scope of this document

The ESD lists the EPA's objectives, the potential impacts of finfish aquaculture, and the work required (technical studies) to support the EIA (EPA 2013). The scope of the technical studies and the section where it is addressed in this document is provided in Table 2.1.

Table 2.1 Technical studies required to support the EIA and the section of this document where they are addressed

Marine environmental quality		Section
EPA objective	To maintain the quality of water, sediment and biota so that the environmental values, both ecological and social, are protected	
Potential impacts	Potential impacts include: • Impacts to water and sediment quality through release of fish feed and faeces leading to nutrient and organic enrichment of the marine environment. • Impacts to water, sediment and biota quality through release of pharmaceuticals or metals/metalloids in fish feed into the marine environment.	
Work required	• Document baseline water and sediment quality (over an approximate 12 month period) in the region of the strategic proposal area in order to effectively capture seasonal and spatial variability to the greatest extent possible, including the following parameters: • **Water** – nutrients, dissolved oxygen (DO), phytoplankton community composition, chlorophyll-a, total suspended solids (organic), hydrogen sulphide (H_2S) and light attenuation coefficient. • **Sediment** – total nitrogen, total phosphorous, total organic carbon (TOC), redox, ammonia (NH_3), DO, H_2S, sediment trace metal and organic concentrations. • **Note** – The Office of the Environmental Protection Authority (OEPA) considers that testing for baseline levels of H_2S in both sediment and water would only be required to be conducted once.	Section 5
	• Accurate and validated modelling of surrounding hydrodynamics, to understand dispersion, deposition and accumulation of nutrients, trace contaminants, organic waste material and pharmaceutical/chemical wastes from the sea cages and any other associated infrastructure. Hydrodynamic and particle transport modelling should take into account factors such as tides, meteorological and seasonal ocean conditions and should be linked to the ecological modelling.	Section 4.6 Appendix F Appendix G
	• A clear and comprehensive description of the predicted cumulative environmental effects of the future proposals within the strategic proposal area operating at maximum capacity based on professional judgement and supported by ecological models that are relevant to the locality and linked to the hydrodynamic modelling. This should include impacts to biodiversity; abundance and biomass; water, sediment and biota quality and ecosystem processes. • Predicted changes in sediment characteristics, both physically (e.g. organic content and TOC) and chemically (e.g. nutrients, H_2S, metals, DO, redox discontinuity) under the most likely or indicative cage locations and configurations to the outer boundary of the zone of reversible impact, for best, worst and most possible case.	Section 7 Section 7.3
	• The proponent must demonstrate a good understanding of the natural rates and types of ecological processes operating in the area and evaluate the possible extent and severity of any changes to the types and/or rates of processes under best case, worst case and most likely case scenarios. • This should include the development of a nutrient budget with and without the potential strategic proposal and future proposals to use as a tool to assess changes in variables such as loading, feeding regimes, assimilation capacity and FCRs etc. The assessment must address the cumulative effects of all elements of the strategic proposal.	Sections 3; 5 Section 4.4.2

	• The documentation should also include a review of the suitability and applicability of the models, and the interpreted outputs of the models, by an independent expert.	Section 4.6.3 Appendix E
	• Develop an environmental quality management framework (EQMF) for the strategic proposal, and to apply to future proposals, based on the recommendations and approaches in Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC & ARMCANZ, 2000) and State Water Quality Management Strategy Report 6 (it is an expectation that the Department of Fisheries would liaise with the OEPA regarding this framework). The framework is underpinned by defining the environmental values to be protected, identifying the environmental concerns or threats and establishing the environmental quality objectives (EQO) and levels of ecological protection to be achieved and where they apply spatially (these should be included in a detailed map). (Note that the effects on environmental quality and biota are linked.) This establishes a framework for the Environmental Impact Assessment (EIA) of the strategic proposal as well as for managing the ongoing operations from future proposals.	Developed separately
	• Develop cause/effect pathway models for nutrient and organic enrichment, sedimentation and other relevant environmental issues of concern.	Section 4.4
Benthic communities and habitat		**Section**
EPA objective	To maintain the structure, function, diversity, distribution and viability of benthic communities and habitats at local and regional scales.	
Potential impacts	Potential impacts include: • direct disturbance or loss through the installation of anchors, wire sweep (deviation to the span of cables), mooring blocks and dragging nets; • direct and indirect impacts or loss through uneaten feed and faeces causing nutrient and organic enrichment of the marine environment leading to shading, smothering, deoxygenation or potential disease of benthic communities and habitats.	
Work required	• Design and conduct a geo-referenced benthic habitat survey with the objective of mapping accurately the spatial extent of benthic habitats (including corals, macro-algae, seagrass, mangroves, filter feeders, microphytobenthos and presence of sediment infauna communities) and defining local assessment units to assess permanent loss of benthic primary producing habitats (BPPH) (in the context of EAG 3). Benthic habitat mapping should at least extend to the outer boundary of the area where both irreversible and reversible effects on biota are predicted to occur and extend into the zone of influence.	Section 4.3 Section 5.5
	• Predict and spatially define zones of high impact (irreversible loss of abundance/biomass or diversity of biota or ecological processes), moderate impact (reversible loss of abundance/biomass or diversity of biota or ecological processes within 5 years) and influence (changes in environmental quality or physiological stress, but no loss of biota or ecological processes) likely to result from the strategic proposal, and therefore the boundary beyond which there will be no effect. These zones need to be derived at maximum capacity and most likely pen configuration and accurately mapped to represent the aquaculture zones footprint. This information will inform the future proponents when selecting the locations and numbers of potential impact sites and un-impacted reference sites.	Section 7
Marine fauna		**Section**
EPA objective	To maintain the diversity, geographic distribution and viability of fauna at the species and population levels	

Potential impacts	• Potential impacts to marine fauna from disturbances such as noise (during construction and operation), lighting, vessel strike and human interaction, entanglement and physical barriers imposed by infrastructure. • Potential impacts on seabirds through changes to population levels, levels of available food and predation. • Potential impacts on wild fish populations, habitats and genetic diversity through introduction of pathogens and parasites, escaped fish and discharge of uneaten feed, faeces and pharmaceuticals. • Potential impacts on fisheries and fisheries production.	
Work required	Marine mammals, seabirds and other significant marine fauna • Identify and assess the values and significance of marine faunal assemblages within the strategic proposal area and immediate adjacent area and describe these values in a local, regional and State context. • Identify critical windows of environmental sensitivity for seabirds, marine mammals, including the Australian Sea Lion (*Neophoca cinerea*), other significant marine fauna and key fisheries in the strategic proposal area and immediate adjacent area. • Describe the presence of marine mammals, including the Australian Sea Lion (*Neophoca cinerea*), seabirds and other significant marine fauna in the proximity of the strategic proposal area and document any known uses of the area by them (e.g. foraging, migrating, calving and nursing etc). • Design, detail and conduct a targeted survey for seabirds. The survey should target the distribution, nesting and roosting habits of all locally relevant seabird species with consideration of survey timing to meet suitable weather conditions, time of day and season for presence of seabirds.	Sections 3; 8 Appendix A Appendix B Appendix C Appendix D Developed separately
	• Identify the construction and operational elements of the proposal that may affect significant fauna and fauna habitat.	Section 4.4.1 Section 8
	• Describe and assess the potential direct and indirect impacts that may result from construction and operation of the proposal to marine mammals, including the Australian Sea Lion (*Neophoca cinerea*), seabirds and other significant marine fauna and their habitat.	Section 8.3 Appendix A Appendix B Appendix C Appendix D
	• Identify measures to mitigate adverse impacts on marine mammals, including the Australian Sea Lion (*Neophoca cinerea*), seabirds and other significant marine fauna and their habitat so that the EPA's objectives can be met. • Describe possible management options to address potential impacts on marine fish populations, marine mammals, including the Australian Sea Lion (*Neophoca cinerea*), seabirds and other significant marine fauna and the surrounding environment. This must include but is not limited to: uneaten feed, marine parasites, biofouling control methods and interaction or entanglement with marine fauna (through development of a marine fauna interaction plan).	Section 8.3 Appendix A Appendix B Appendix C Appendix E Section 8 Appendix A Appendix B Appendix C Appendix D
	Biosecurity • Describe translocation, biosecurity and management arrangements addressing: fish disease/pathogen (including parasites) management and incident response, strategies for preventing outbreaks and/or preventative treatments chemicals to escape into the surrounding environment; brood stock and translocation issues; and prevention and management of escaped fish	Developed separately
	Fisheries • Describe commercial and recreational fishing activity in the Northampton region and Abrolhos Islands that may be affected by the proposal. • Describe and assess the potential direct and indirect environmental impacts on recreationally and commercially important marine species, including impacts to migratory patterns, spawning areas and nursery areas.	Section 8.2 Appendix C

3. Site description

3.1 Climate

The Abrolhos Islands are a group of islands located approximately 60 km west of Geraldton, Western Australia (WA). The islands are clustered into three main groups – Wallabi, Easter and Pelsaert, and are approximately 100 km in length from the northern to the southern tip. In the warmer months (January–April), the Abrolhos Islands experience strong south to south easterly winds in the morning and generally stronger south to south westerly winds in the afternoon (Webster et al 2002). High wind speeds are consistently recorded in the afternoons from September through to March, with the months of strongest wind being December, January and February. During the cooler months, winds tend to be weaker and more variable in direction.

The MWADZ is also characterised by frequent storms and squalls. In the winter months, storms to the south of the region can bring gales and strong winds up to 35 m/sec (Webster et al. 2002). Squalls can also occur in the summer months of December–April, and can generate wind speeds between 25 and 30 m/sec that can occur in any direction (Webster et al. 2002). The majority of rainfall (average 272 mm) occurs between April and September. Mean air temperatures range between 21 to 27°C and 16 and 22°C in the warmer and cooler months, respectively.

The Abrolhos region is occasionally subject to cyclonic activity during the cyclone season from December to May, with more than half the recorded cyclones occurring between March and May. Since 1915, a cyclone has passed through coastal waters within 400 km of the region approximately every 2.5 years on average.

3.2 Oceanography

The waters of the MWADZ are deep (25-50 m), well flushed and experience high levels of water circulation and dispersion (Figure 3-1). The MWADZ is located on the edge of the WA continental shelf between 28°S and 29°S, in the pathway of the warm poleward-flowing Leeuwin Current (Pearce 1997). It is also situated in the Zeewijk Channel, one of three breaks in the Houtman Abrolhos archipelago (Maslin 2005). The region surrounding the Abrolhos is a dynamic system influenced by large-scale regional currents (e.g. Leeuwin Current, Capes Current), wind stresses, upwelling and wave dynamics (Pearce & Pattiaratchi 1999, Feng et al. 2007, Waite et al. 2007, Woo & Pattiaratchi 2008, Rossi et al. 2013). The Leeuwin Current is a well-studied oceanic flow of warm, low salinity tropical water (originating in the Timor Sea) that travels southwards along the Western Australian coast. It is driven by a southwards pressure gradient, and under the influence of Coriolis deflections, hugs the coastline as it travels from near North West Cape to Cape Leeuwin (south of Perth) and then onwards to the Great Australian Bight (Cresswell 1991).

The Leeuwin Current flow is strongest in autumn, winter and early spring, raising sea surface temperatures. The flow is greatest and most consistently south along the shelf break, a relatively short distance to the west of the Abrolhos Islands (Webster et al. 2002). The currents through and inshore of the islands vary spatially and temporally. During the late spring and summer months, the current through and inshore of the islands tends to set to the north, driven by the prevailing southerly winds with occasional current reversal to the west along the shelf break (Pearce et al. 1999). During the winter months strong westerlies and north-westerlies can generate southward setting currents through and inshore of the Abrolhos Islands (Pearce et al. 1999).

The waters of the MWADZ are well flushed and experience high levels of water circulation and dispersion. Their position within the Zeewijk Channel means that the area is exposed to significant westerly currents, which expel large volumes of water out of the zone toward the continental shelf slope (Maslin 2005). Differences in the hydrodynamics between the surface and bottom of the Zeewijk channel have been shown to affect particle transport times (Maslin 2005). Particles in the surface waters are expected to be flushed out of the system rapidly (within 24 hrs), while particles at the bottom of the water column are expected to be retained in the system for longer periods, due to the recirculation of bottom currents (Maslin 2005).

In addition, previous oceanographic work completed by (Sukumaran 1997) at the Easter Group islands indicated fast flushing times (i.e. from 0.5 to 1.5 days) in the shallow waters of the Easter Group lagoon (Sukumaran 1997). The proposed MWADZ is located in a more exposed area north of the Pelsaert Group and east of the Easter Group of islands. Currents speeds through the MWADZ are expected to be higher than that reported in Easter Group lagoon, leading to lower retention times and enhanced flushing capacity.

Wave heights in the open ocean near the south westerly margins of the Abrolhos Islands average ~2 m, and can exceed 4 m during storm events. Wave heights are substantially lower on the eastern leeward sides of the Abrolhos Islands and in the areas near the MWADZ, with average wave height reaching ~1.2 m (Webster et al. 2002). The majority of the swell approaches the islands from the south and west 78% of the time (Department of Fisheries 2000).

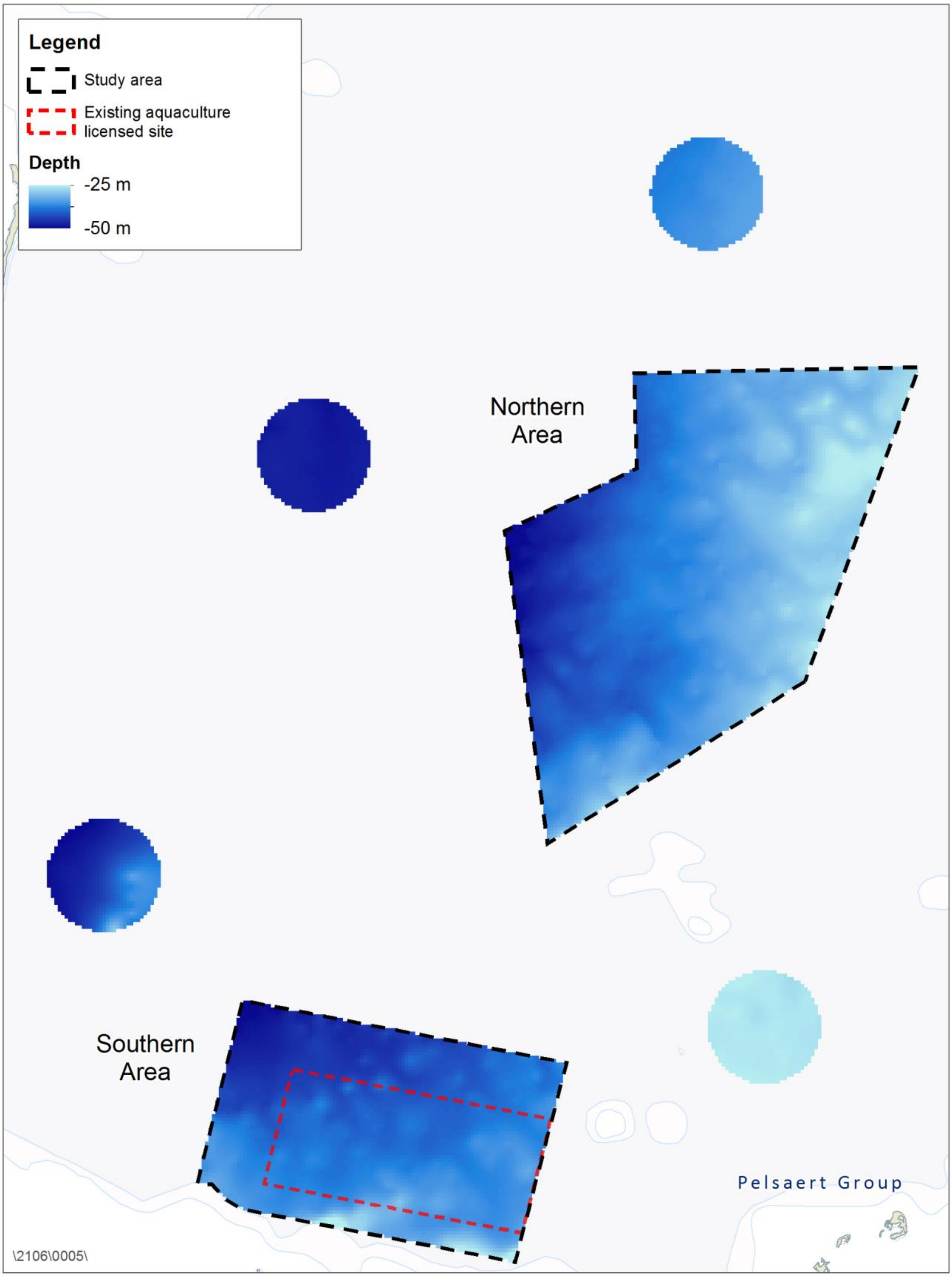


Figure 3-1 Bathymetry of the proposed MWADZ and reference areas

3.3 Sediment biochemical processes

Sediment characteristics of the Abrolhos Islands vary with depth and space (Section 5.4). Sediments in the MWADZ are sandy with grain sizes ranging <0.06 to 2 000 µm. Concentrations of nutrients and organic material are very low and anecdotal observations suggest that much of the MWADZ consists of shallow sediments (15 m thick) overlying rock. Attempts to retrieve consolidated cores for model validation failed owing to the depth of the water (beyond diving depth) and the shallow nature of the sediments which impeded the coring process. Sediment grabs were not appropriate for discerning natural biochemical processes, because of difficulties in retaining consolidated and unmixed samples. Biochemical processes were therefore assumed to be consistent with shallow, well oxygenated sediments. The characteristics of sediments matching these criteria were ground-truthed with the relevant literature (Berner 1980, Boudreau 1997, Fossing et al. 2004).

3.4 Benthic marine fauna and flora

The reefs of the Abrolhos are unusual in that they support both rich coral and macroalgal communities, with corals dominant on the leeward reef sections and macroalgae dominant on the more windward reef sections (Wells 1997).

The corals of the Abrolhos Islands are diverse, with 184 species from 42 genera recorded (Veron & Marsh 1988). While being at the extreme southern limit of their latitudinal range, the Abrolhos Islands coral populations are considered to be reproductively active, with 60% of the 184 species recorded to spawn in late summer (Babcock et al. 1994). As such, the Abrolhos Islands support extensive coral cover despite their southerly location, and the growth rates and calcification rates of *Acropora formosa* and *Porites* spp. from the Abrolhos Islands have been reported to be within the range reported for their tropical counterparts (Smith 1981, Harriott 1998). The family Acroporidae (*Acropora* and *Montipora*) dominates the coral communities at the Abrolhos Islands, and a marine heat wave in 2010/2011 (Pearce & Feng 2013) resulted in coral bleaching and subsequent coral mortality (~12% decline in coral cover) at the Abrolhos (Abdo et al. 2012, Moore et al. 2012). The sea surface temperatures at the Abrolhos Islands were once again above seasonal averages in the 2011/2012 summer period (NOAA 2015), with additional coral bleaching and mortality likely due to the extent of the thermal anomaly.

Besides corals, the Abrolhos has rich and diverse macroalgal communities, with 295 macroalgal species recorded – 13.6% are considered to be endemic to the Abrolhos, and only ~10% have a tropical affinity (Phillips & Huisman 2009). The macroalgal abundance in the lagoonal reefs at the Abrolhos is high in comparison to other tropical coral reefs (Wilson & Marsh 1979) and includes large stands of fucoid algae and kelp, *Ecklonia radiata*, not found on coral reefs (Womersley 1981). It appears that the grazing rates of invertebrates and fish at the Abrolhos are less than on tropical reefs (Hatcher & Rimmer 1985). As such, little of the macroalgal production is consumed by grazers, but rather the macroalgae are removed by storms carried into the lagoons as a nutrient subsidy of particulate carbon (Wells 1997). The lagoons therefore include large aggregations of unattached macroalgae and macroalgal fragments that contribute to a rich detritus-based food web, which includes the Western Rock Lobster fishery – of which ~19% of the WA catch is taken from the Abrolhos region (Abdo et al. 2012).

One of the dominant macroalgae in the Abrolhos is the kelp *Ecklonia radiata*, which can reach densities of 8.2 plants/m^2 at ~12 m depth in lagoonal area (Hatcher et al. 1987). Besides *Ecklonia*, fleshy macroalgae form a major component of the benthic communities of the Abrolhos, where the high-energy outer reef slopes support rich and dense macrophyte communities characterised by large brown algae (e.g. *Dictyota*, *Glossophora*, *Sargassum*) mixed with fleshy red and green algae (e.g. *Asparagopsis*, *Hypnea*, *Laurencia*, *Plocamium*, *Caulerpa*; Crossland et al. 1984). The protected reef areas within the lagoon vary seasonally, whereby large phaeophytes (e.g. *Caulocystis*, *Cystophyllum*, *Hormophysa*, *Sargassum*, *Turbinaria*) are common in summer, and other fleshy algae (e.g. *Eucheuma*, *Laurencia*) are more common in spring (Crossland et al. 1984).

Besides the dominant coral and macrolgal communities, ten seagrass species have been recorded at the Abrolhos (Brearley 1997). Seven of these species (*Amphilbolis antarctica*, *A. griffithii*, *Thalassodendron pachyrhizum*, *Posidonia angustifolia*, *P. australis*, *P. coriacea*, *P sinuosa*) are predominantly temperate species, and three (*Syringodium isoetifolium*, *Halophila decipiens*, *H. ovalis*) have a tropical affinity (Brearley 1997). However, the seagrass communities at the Abrolhos are sparse and species poor compared to the mainland locations of Shark Bay and Geraldton (Brearley 1997).

Wilson and Marsh (1979) originally considered the non-coral fauna of the Abrolhos to be relatively impoverished and unstable in comparison to the corals. However, diverse molluscs (492 species; Wells & Bryce 1997), echinoderms (172 species; Marsh 1994), oligochaetes (Erseus 1997), polychaetes (Hutchings 1997), and hydroids (Watson 1997) have been recorded, indicating that the known diversity of benthic marine biota in the Abrolhos is substantially higher than that suggested by Wilson and Marsh (1979). In terms of the subtidal molluscs at the Abrolhos, >65% of the bivalves have a tropical affinity, whereas ~45% of the gastropods have a tropical affinity (Glover & Taylor 1997). Moreover, while no literature is available on the diversity of sponges at the Abrolhos, they did comprise a major component of the dredge samples used for the mollusc surveys (Glover & Taylor 1997), and given the high diversity of sponges recorded at Ningaloo (Heyward et al. 2010), the sponges are therefore expected to be relatively diverse at the Abrolhos.

The benthic habitats of the Abrolhos also support rich fish communities, with up to 389 fish species recorded (Hutchins 1997). The majority of these species (~60–65%) are tropical species, ~15% are subtropical, and ~20–25% are temperate species (Hutchins 1997, Watson et al. 2007). Moreover, the structure of the fish assemblages differ between fished and non-fished areas (Watson et al. 2007), and there is a greater relative abundance of many of the targeted fish species in areas protected from fishing (Watson et al. 2009, Nardi et al. 2004).

In addition to the reefal areas, the lagoons and areas east of the Abrolhos Islands are comprised of large open sandy habitats – areas of which are commercially trawled for the scallop *Amusium balloti*. Areas sampled for molluscs over the scallop grounds were generally characterised by fine carbonate sand with shell debris, with patches of coralline algal rubble with attached sponges (Glover & Taylor 1997). The molluscan community was dominated by suspension feeding bivalves (particularly pectiniids), a suspension feeding gastropod (*Monilea lentiginosa*), an algal grazing gastropod (*Calthalotia mundula*), echinoderms (*Prionocidaris bispinosa*, *Luidia australiae*, *Astropecten preissi*), and sponges (Glover & Taylor 1997).

3.5 Marine mammals and turtles

The Abrolhos Islands and surrounding waters provide important habitat for an array of marine mammals, comprising mainly whales, dolphins and sea lions. Thirty one cetacean and two pinniped species are known to occur with a 50 km radius of the MWADZ (DoE 2014a). Some species occasionally transit through the area at low densities, but there is insufficient information to confirm a definitive presence. Species that are likely to occur within a 50 km radius include: blue whale, humpback whale, Australian sea lion, Indo-Pacific bottlenose dolphin and the common bottlenose dolphin. Species with a low likelihood of occurring include: the blue whale, southern right whale, Bryde’s whale, killer whale and the dugong. Four marine turtles may occur within a 50 km radius, including the loggerhead turtle, flatback turtle, leatherback turtle and green turtle, with the last two species more likely.

3.6 Finfish, sharks and rays

The benthic habitats of the Abrolhos support rich fish communities, with up to 389 fish species recorded (Hutchins 1997). The majority of these species (~60–65%) are tropical species, ~15% are subtropical, and ~20–25% are temperate species (Hutchins 1997, Watson et al. 2007). The structure of the fish assemblages differs between fished and non-fished areas (Watson et al. 2007) and there is a greater relative abundance of many of the targeted fish species in areas protected from fishing (Watson et al. 2009, Nardi et al. 2004).

These rich communities host a number of threatened, endangered and protected species. These comprise sharks, rays, Queensland grouper and syngnathid (pipefish, seahorses and seadragons). Most syngnathid species inhabit shallow, sheltered coastal waters, well away from the proposed MWADZ. While Queensland grouper possibly exist at the Abrolhos Islands the likelihood of an interaction with the proposed sea-cage operations was consider remote (DoF 2015b). However, interaction between the sharks/rays and the proposed sea-cages is considered more plausible (DoF 2015b). The significant finfish of the Abrolhos are considered in detail in DoF (2015a, 2015b).

3.7 Seabirds

The Houtman Abrolhos is the most significant seabird breeding location in the eastern Indian Ocean. Eighty percent (80%) of the brown (Common) noddies, 40% of sooty terns and all lesser noddies found in Australia nest at the Houtman Abrolhos (Ross et al. 1995). It also contains the largest breeding colonies in Western Australia of wedge-tailed shearwaters, little shearwaters, white-faced storm petrels, white-bellied sea eagles, osprey, caspian terns, crested terns, roseate terns and fairy terns (Storr et al. 1986, Surman and Nicholson 2009). The Houtman Abrolhos also represents the northernmost breeding islands for both the Little Shearwater and White-faced Storm Petrel.

Components of the avifauna at the Abrolhos are protected under three National and State Acts:

- Environment Protection and Biodiversity Conservation (EPBC) Act 1999;
- Conservation and Land Management (CALM) Threatened and Priority Fauna Database and
- Western Australian Wildlife Conservation (Specially Protected Fauna) Notice 2014.

Migratory species are protected under the EPBC Act (1999), and are included in the Japan Australia Migratory Bird Agreement (JAMBA), the China Australia Migratory Bird Agreement (CAMBA) and the Republic of Korea-Australia Migratory Bird Agreement (ROKAMBA). Of these, all migratory waders recorded in Surman and Nicholson (2009), as well as the eastern reef egret and seabirds including the bridled tern, caspian tern, crested tern, osprey and white-breasted sea eagle, are listed under migratory bird agreements with Japan, China or Korea. Birds covered by these agreements are listed in Schedule 3 under the Wildlife Conservation Act 1950 (WA).

Eight bird species found at the Abrolhos are also listed under the CALM Threatened and Priority Fauna Database, although only one of these species, the lesser noddy, is likely to interact with the aquaculture lease area.

Five seabird species occur in the vicinity of the aquaculture leases that are listed under the Western Australian Wildlife Conservation (Specially Protected Fauna) Notice 2014, Schedule 1: Fauna that is rare or likely to become extinct. These are the:

- lesser noddy
- Hutton’s shearwater
- fairy tern
- Indian yellow-nosed albatross, and
- black-browed albatross

Both the lesser noddy and fairy tern breed at the Abrolhos, whereas the Hutton’s shearwater migrates through the region in late spring, with up to 50 birds occurring in flocks off Eastern Passage (Easter Group) and The Channel (Pelsaert Group) (Surman and Nicholson 2009). Albatrosses in contrast are winter visitors (Surman pers. obs). Hutton’s shearwaters forage with wedge-tailed shearwaters on small pelagic fishes and squids, including species that are likely to congregate near sea-cages.

Seventeen species use the Abrolhos as breeding regular breeding grounds. These are the white-bellied sea eagle, osprey, wedge-tailed shearwater, little shearwater and white-faced storm Petrel, pacific gull, silver gull, caspian tern, crested tern, bridled tern, roseate tern, fairy tern, brown noddy, lesser noddy, eastern reef egret, pied oystercatcher, and pied cormorant (Surman and Nicholson 2009).

Three species of seabird are considered most at risk due to interaction with the proposed MWADZ, including the Pacific gull, silver gull and the pied cormorant. Approximately 356 pairs of silver gulls were recorded nesting during an Abrolhos wide survey conducted in 2006 (Surman and Nicholson 2009). The largest colonies were observed on Long Island in the Wallabi Group (142 pairs), Pelsaert Island (43), Leo’s Island (34) and Wooded Island (33).

Pied cormorant, silver gull and Pacific gull populations at the Houtman Abrolhos are currently reliant upon natural food sources only. The establishment of finfish farms in either of the proposed areas could potentially lead to in changes in the size of these species populations (or changes in colony location) that could result in increased competition with, or predation of other seabirds or alteration in breeding habitat (Surman 2004). Adult silver gulls are particularly at risk given their propensity for rapid population growth in response to opportunistic food sources. These aspects of breeding biology allow silver gulls to respond rapidly to seasonal changes in food availability.

4. Methods and assumptions

Section 4 of this document summarises the methods and assumptions that underpin the technical studies. The section first provides a technical overview of the methods and experimental design supporting the baseline data collection process. It then goes on to describe the approach to identifying the relevant cause–effect / pressure response–relationships, before describing the approach to model development. All of the work described in this Section is the work of BMT Oceanica, BMT WBM and UWA AED, unless otherwise specified.

4.1 Metocean data collection

4.1.1 Data collected for this project

Metocean data, consisting of conductivity-temperature-depth (CTD), wave height and current speeds, were collected over a 10 month period at a total of four sites, and captured each of the calendar seasons. Metocean data were collected using bottom–mounted data loggers in conjunction with Acoustic Doppler Current Profilers (ADCP). Four ADCPs were deployed in total: one in each of the northern and southern areas), and one in each of two regional locations (north-east and south-east, respectively) (Figure 4-1). A total of 6 deployments were made over a 10 month period (Table 4.1).

Table 4.1 Timing of ADCP deployments

Northern and southern MWADZ	Regional sites
16 May 2014 – 19 June 2014	17 July 2014 – 19 November 2014
17 August 2014 – 18 September 2014	19 November 2014 – 18 March 2015
9 November 2014 – 10 December 2014	-
9 February 2015 – 11 March 2015	-

4.1.2 Historical data

In addition to the above data, some historical data were also utilised including:

- Wave data from the Outer Channel at Geraldton which were provided by the Mid West Port Authority for a ten year period to 1 May 2014
- ADCP data collected in October 2002 and September 2003 from a location within the Pelsaert Group just west of the northern area of the proposed MWADZ
- Tide gauge data from Geraldton port from 1 Jan 2014 to present.

4.2 Baseline water and sediment quality

Coinciding with the metocean data collection period, a baseline water and sediment quality monitoring program was also undertaken between May 2014 and March 2015. The purpose of the monitoring program was to effectively capture the seasonal and spatial variability in a range of water and sediment parameters, as per the requirements of the ESD. Field work associated with the baseline program was undertaken by the DoF research division. Data analysis and interpretation was undertaken by BMT Oceanica, BMT WBM and UWA AED.

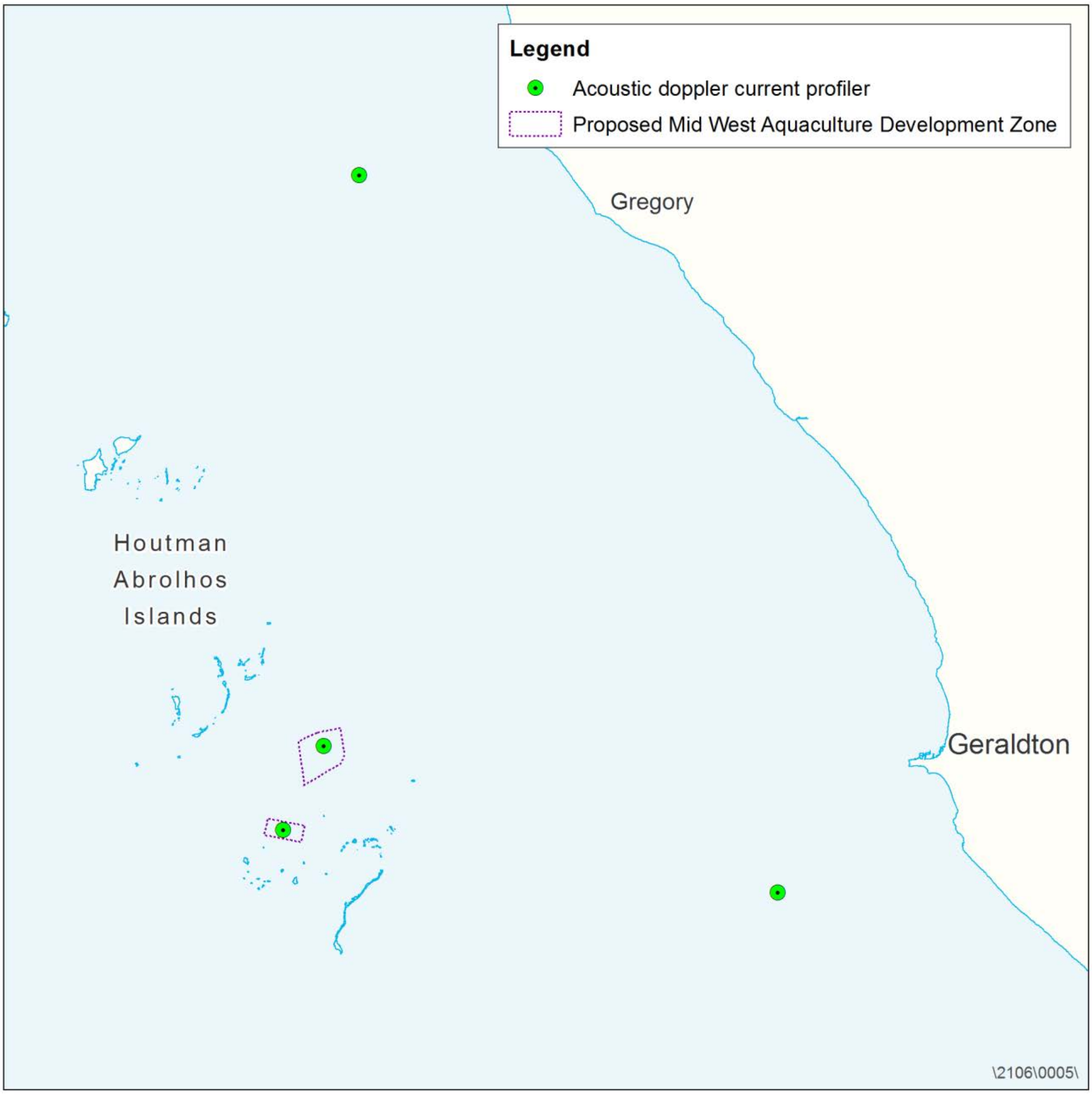


Figure 4-1 Location of acoustic doppler current profilers for metocean data collection

4.2.1 Monitoring program design

Water quality

Water samples were taken at a total of 28 sites comprising of 9 sites in the northern area and 6 sites in the southern area, and 12 reference sites located at the perimeter of the MWADZ (Figure 4-2). Several of the water quality sites were positioned at the boundary of the northern and southern areas of the MWADZ, while others were positioned so as to co-located with sediment sampling sites (Figure 4-3).

Sites were also positioned to allow for future Multiple-Before-After-Control-Impact (MBACI) framework of Keogh and Mapstone (1997). In line with this framework, the design includes multiple impact locations (north and south locations), multiple reference locations and multiple data sets, each collected over multiple seasons.

Phyisco-chemical readings were taken using a Hydrolab Datasonde 5 Multiparameter Probe. The measured parameters (and associated units) were:

- temperature (°C)
- pH/oxidation/reduction potential (pH units, mV)
- conductivity/salinity (mS/cm, ppt)
- dissolved oxygen (DO, mg/L)
- turbidity (NTU)
- depth (m)
- incident irradiance (photosynthetically active radiation [PAR])

Profiles of the above parameters were logged through the water column using a field computer running the Hydras 3 LT data logging software. In addition, incident irradiance at the sea surface was measured using a JFE Advantech ALW-CMP PAR logger installed in an open (unshaded) area on Rat Island at the DoF research station for a period of 12 months. Two identical PAR loggers were deployed ~1 m from the bottom, within each of the northern and southern area of MWDAZ in the same locations as the ADCPs. The PAR loggers were fixed to the deployment frame of the ADCP's, and the data downloaded with the metocean data.

At each water quality monitoring site, water samples were collected and analysed for the following

- ammonium (NH_4)
- nitrate + nitrite (NO_X)
- chlorophyll-a
- total suspended solids (TSS), including loss on ignition
- total phosphorus (TP) + total nitrogen (TN)
- orthophosphate (FRP)
- total organic carbon (TOC) + dissolved organic carbon (DOC)
- hydrogen sulphide (H_2S)–subset of sites and bottom sample only from summer & winter
- polycyclic aromatic hydrocarbons (PAH) (ultra-trace level)
- total petroleum hydrocarbons (TPH)
- phytoplankton community

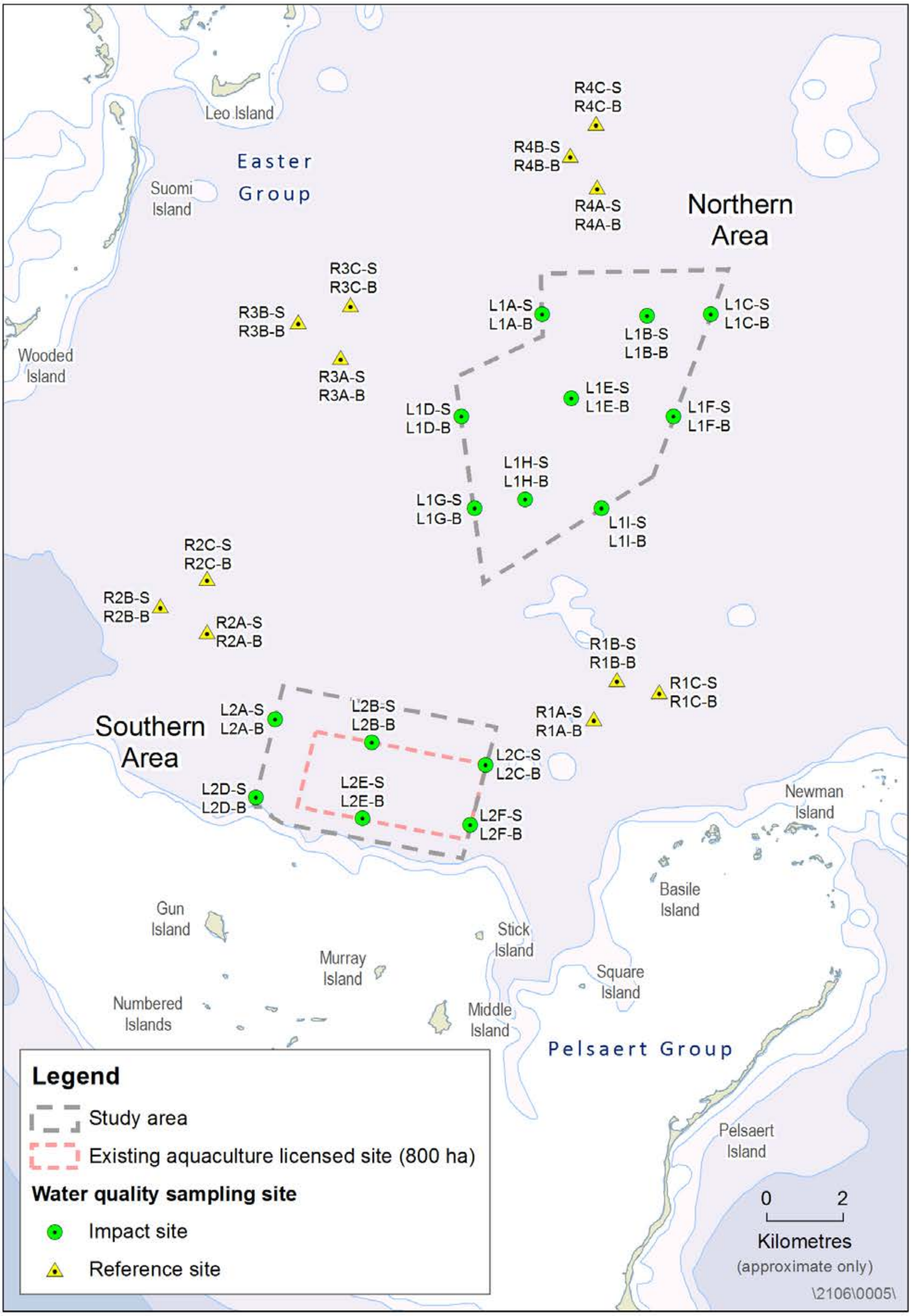


Figure 4-2 Baseline water quality sampling sites

Water samples for chemical analyses were collected using a 4.2 L Van Dorn sampler deployed at each of the 27 water quality sampling sites (Table 4.2), twice within each season, and from the surface (0–1 m) and bottom (~1m from seafloor) of the water column. Once retrieved, the water samples were divided into the aliquots required for each analysis. Once each required sub-sample was obtained, the respective sample bottles were placed into an esky with ice or ice bricks. Once back on land, samples were appropriately stored or post-processed prior to transportation to the laboratory.

For phytoplankton community samples, three discrete water samples were taken using the Van Dorn Sampler (4.2 L each) at the surface, mid and bottom of the water column. The samples were then combined and homogenised in a clean 20 L bucket. This equated to an integrated water column sample of 12.8 L, from which the 250 mL aliquot was obtained.

Table 4.2 Timing of baseline sampling

	Autumn				**Winter**				**Spring**				**Summer**			
	May		Jun		Aug		Sep		Nov		Dec		Feb		Mar	
	S	B	S	B	S	B	S	B	S	B	S	B	S	B	S	B
Light intensity																
In situ PAR data loggers	In		Out		In		Out		In		Out		In		Out	
Water quality sampling																
Physical water quality profiling	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Ammonium / Nitrite + Nitrate / FRP	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Total nitrogen / Total phosphorus	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Total organic carbon	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Total suspended solids	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Chlorophyll-a	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
PAH/TPH	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Hydrogen sulphide						✓								✓		
Phytoplankton	✓				✓				✓				✓			
Sediment quality sampling																
Total nitrogen / Total phosphorus						✓								✓		
Total organic carbon / Dissolved organic carbon						✓								✓		
Trace metals						✓								✓		
PAH/TPH						✓								✓		
pH / oxidation–redox potential						✓								✓		
Particle size diameter						✓								✓		
Infauna						✓								✓		
Habitat mapping																
Single beam hydro-acoustic mapping		✓														
Metocean																
ADCP (Department of Fisheries)	In		Out		In		Out		In		Out		In		Out	
ADCP (BMT WBM)			In[1]						Out/In						Out	

Notes:

1. First deployed in mid July

Sediment quality

Sediment samples were obtained at a total of 33 sites comprising of 12 sites in the northern area and 9 sites in the southern area, and an additional 12 reference sites, located at the perimeter of the MWADZ. As with the water quality sites, sites were positioned to allow for future MBACI style analyses, and stratified to capture the presence of sediment quality gradients, if present (Figure 4-3).

Sediment samples were collected for the determination of:

- total phosphorus (TP)
- total nitrogen (TN)
- total organic carbon (TOC)
- trace metals: silver (Ag), arsenic (As), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), antinomy (Sb), selenium (Se), zinc (Zn), iron (Fe), manganese (Mn), lithium (Li), and mercury (Hg)
- polycyclic aromatic hydrocarbons (PAH) (ultra-trace level)
- total petroleum hydrocarbons (TPH)
- pH / redox–oxidation potential (ORP)
- particle size distribution , including wet/dry weight ratio
- infauna community composition

Initially sediment sampling was attempted using a modified sediment corer. However, the depth of the water column and the presence of an underlying rocky platform prevented effective sampling. All subsequent sampling was undertaken using a Petite Ponar sediment grab.

Three replicate samples were collected at each sample site. Each of the three replicates were then combined and homogenised, and aliquots were obtained from the homogenised sample. Samples were analysed for the parameters listed in Table 4.3. Samples were stored on ice in the field before being frozen and transported to the laboratory for analysis.

Infauna samples were collected using the Petite Ponar grab. The content of each grab was carefully rinsed through a series of graded sieves (to a minimum of 1 mm). Any material greater than 1 mm was fixed in formalin prior to transportation to the laboratory. Infauna were carefully picked from the samples and retained for identification to species level.

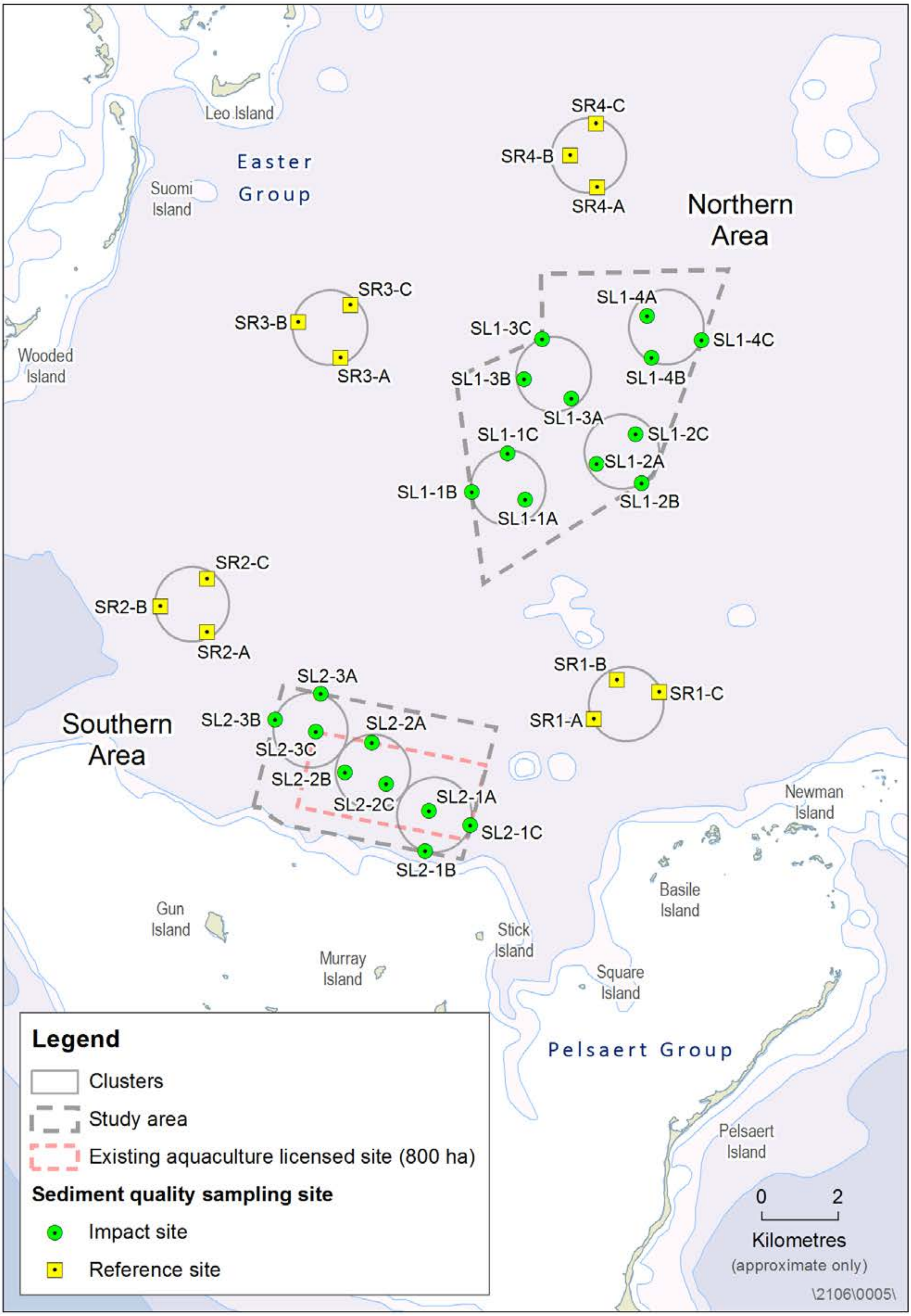


Figure 4-3 Baseline sediment quality sampling sites

Table 4.3 Sediment quality sample vessel and preservations requirements

Analyte	Details	
Total organic carbon (TOC)	Sample volume	125g
	Sample bottle	Polyethylene bottle
	Preservation technique	Fill sample bottle ¾ full.
	Maximum sample holding time and storage conditions	1 month, frozen sample
	Reporting limit	0.05%
Total nitrogen (TN) Total phosphorus (TP)	Sample volume	125g
	Sample bottle	Polyethylene bottle
	Preservation technique	Fill sample bottle ¾ full.
	Maximum sample holding time and storage conditions	1 month, frozen sample
	Reporting limit	10 mg/kg (TP), 0.005% (TN)
Trace metals (Ag, As, Cd, Co, Cr, Cu, Ni, Pb, Sb, Se, Zn, Hg, Fe, Li, Mn)	Sample volume	250g (250g for Hg)
	Sample bottle	Acid washed Polyethylene bottle Hg – plastic jar with Teflon lid
	Preservation technique	
	Maximum sample holding time and storage conditions	1 month, chilled sample 6 months, frozen sample
	Reporting limit	0.001 (Ag, As, Cd, Co, Cu, Pb, Se, Sb); 0.005 (Cr); 0.01 (Ni, Zn); and 0.0001 (Hg) mg/L
Polycyclic aromatic hydrocarbons (PAH) (ultra trace) Total petroleum hydrocarbons (TPH)	Sample volume	100 g
	Sample bottle	Glass jar
	Preservation technique	None
	Maximum sample holding time and storage conditions	14 days, chill sample and keep in dark
	Reporting limit	0.001 mg/kg
Particle size distribution	Sample volume	200 g
	Sample bottle	Ziplock bag (triple bagged)
	Preservation technique	None
	Maximum sample holding time and storage conditions	Chill sample and keep in dark
	Reporting limit	0.02µm and greater (binned by size classes)
Infauna community composition	Sample volume	200mL
	Sample bottle	Plastic Jar
	Preservation technique	Sieved to 1mm
	Maximum sample holding time and storage conditions	Preserved with 10% Formalin
	Reporting limit	Lowest recognisable taxonomic unit and associated abundance

4.2.2 Statistical analysis

The following section describes the statistical procedures used to analyse the baseline dataset. It includes a technical overview of the approaches to the transformation, interrogation and interpretation of the data. The description is necessarily technical to ensure the approaches used are as transparent as possible.

Water quality

All water quality data were analysed statistically using PERMANOVA. Separate univariate analyses tested the relative importance of three main sources of variance, known as factors: (1) Time (fixed factor, orthogonal with ten levels [months]; (2) Zone vs Reference [ZvR] (fixed factor, orthogonal with two levels [zone & reference]; and (3) Location (fixed factor, nested within ZvR, with six levels). The six levels nested in Location included: northern area; southern area, reference 1; reference 2; reference 3 and reference 4. Data obtained at the surface and bottom of the water column were analysed separately.

For all univariate tests, a Euclidean resemblance matrix was applied on untransformed data prior to analysis with PERMANOVA (non-parametric analysis of variance, Version 1.0.1, Primer-E Ltd) (Anderson et al. 2008). Post-hoc pair wise comparisons were then used to test for differences among levels within significant factors. Results from univariate analyses were presented using graphs of means and standard errors for either time or location.

Phytoplankton

For phytoplankton counts, biovolume and total counts analyses, PERMANOVA routines tested the relative importance of three main factors: (1) Time (fixed factor, orthogonal with four levels: May 2014, Aug 2014, Dec 2014, Feb 2015); (2) Zone vs Reference [ZvR] (fixed factor, orthogonal); and (3) Location (fixed factor, nested within ZvR). All statistical analyses, including post-hoc pair-wise comparison tests on significant factors, were undertaken using PERMANOVA.

Multivariate phytoplankton count data were fourth-root transformed prior to analysis. This transformation down-weighted the contribution of dominant phytoplankton taxa and allowed intermediate or rarer groups to play a part in the analyses (Clarke 1993). The Bray-Curtis dissimilarity measure was used prior to analysis with PERMANOVA. If any of the three factors were significant, they were interpreted using post-hoc pair-wise comparisons to test for differences among levels within each factor. Results of multivariate analysis were presented graphically using a non-parametric, multi-dimensional scaling plot (nMDS), which plotted the centroid (average) of each location by averaging over replicates. Vector overlays of the phytoplankton counts were plotted on the MDS to show correlations with the patterns in the multivariate data.

For multivariate phytoplankton biovolume data and total counts, a Bray-Curtis dissimilarity measure was applied to square-root transformed data to create the resemblance matrix for analysis. Data were zero-adjusted prior to creating resemblance matrix by adding a dummy variable of one to all samples (Clarke et al. 2006). This was undertaken to address the high proportion of blank samples and samples with only one species recorded. Without the use of a dummy variable, a Bray-Curtis matrix would have produced undefined similarities where no species were recorded in two compared samples, and highly varied similarities where only one species was recorded in the two samples. The inclusion of a dummy variable moderates these effects (Clarke et al. 2006). If any of the factors were significant following a PERMANOVA, they were interpreted using post-hoc pair-wise comparisons to test for differences among levels within each factor.

Irradiance and light attenuation

Incident irradiance at the sea surface was measured in an open (unshaded) area on Rat Island. Two further identical PAR loggers were deployed ~1 m from the bottom, one in the centre of the southern area and the other in the centre of the northern area of the MWADZ (Figure 4-1). The loggers were deployed for the periods shown in Table 4.4.

Table 4.4 Dates of light logger deployment

Deployment phase	Season	Month/Year of deployment	Dates of deployment duration
1	Autumn	May–June 2014	16/05/2014–20/06/2014
2	Winter	August–September 2014	17/08/2014–19/09/2014
3	Spring	November–December 2014	09/11/2014–11/12/2014
4	Summer	February–March 2015	09/02/2015–11/03/2015

Data were processed as per Chevron (2012). All data collected between 1000 and 1400 each day was retained for analysis. Data collected by the terrestrial light logger unit was multiplied by 0.96 to estimate the intensity just below the water surface (Chevron 2012). Light attenuation coefficient (K_d) was calculated according to the following equation:

$$Kd = \frac{-Ln\left(\frac{Intensity_{depth}}{Intensity_{Surface}}\right)}{Depth\ (m)}$$

Light intensity (as radiance) was calculated for the 1st, 5th, 20th and 50th percentiles for each of the four logger deployments.

Physical-chemistry

Dissolved oxygen measurements were grouped by location (northern area, southern area and reference locations) and by season (summer, autumn, winter and spring. Summary statistics were then produced for the surface (top 50 cm measured) and the bottom (bottom 50 cm measured) of the water column:

- mean surface
- mean bottom
- 20th percentile bottom
- 5th percentile bottom
- 1st percentile bottom

Sediment quality

All sediment quality parameters were analysed to identify potential patterns between four factors: (1) Season (fixed factor, orthogonal with two levels: winter and summer); (2) Future lease vs Reference [ZvR] (fixed factor, orthogonal); (3) Location (fixed factor, nested within ZvR with six levels: SL1, SL2, SR1, SR2, SR3, SR4); and (4) Site (random factor, nested within Location). All statistical analyses, including post-hoc tests on significant factors, were undertaken using PERMANOVA (Anderson et al. 2008). This method enabled analysis of univariate and multivariate datasets, while not explicitly requiring normalised data or homogeneous variances. All analyses were run using permutations of residuals under a reduced model (n = 9 999 permutations).

For percent particle size distribution, data were square-root arcsine transformed following Underwood (1997). A Bray-Curtis dissimilarity matrix was generated and the data were analysed using PERMANOVA. Multivariate statistical outputs were presented graphically using a canonical analysis of principle coordinates (CAP). The CAP routine was used as there were differences among a priori groups in multivariate space that could not be seen in an unconstrained ordination such as a PCO or MDS plot (Anderson et al. 2008). Vector overlays of the particle size groups were plotted on the CAP to show correlations with the patterns in the multivariate data.

Separate univariate analyses were performed on sediment nutrient concentrations. For percent nitrogen and TOC, data were square-root arcsine transformed prior to analysis as this is a standard transformation for proportional datasets that are often binomially distributed (Underwood 1997). No transformations were necessary in the cases of the ammonium and phosphorus data were. Euclidean distance was used as a dissimilarity measure for all univariate analyses. By using the Euclidean measure, PERMANOVA returns an equivalent test statistic to a standard ANOVA (Anderson et al. 2008). If location were significant, they were interpreted using post-hoc pair-wise comparisons to test for differences among levels within locations. Results from univariate analyses were presented using graphs of means and standard errors.

Trace metal data were analysed using both univariate and multivariate techniques. For the multivariate component, data were initially square-root transformed to down-weight the contribution of dominant trace metals and to allowed intermediate or rarer groups to play a part in the analyses (Clarke 1993). A Bray-Curtis dissimilarity matrix was generated and the data were analysed using PERMANOVA. Results of multivariate analysis were presented graphically using nMDS, which plotted the centroid (average) of each location by averaging over replicates. Upon detection of a significant difference among levels within a factor for the multivariate data, vector overlays were plotted on the MDS. This enabled the top five trace metals that had the strongest correlations with the patterns in the multivariate data to be determined.

The trace metals with the highest concentrations (top 5) as identified by the vector overlay were further explored with separate univariate PERMANOVAs. A Euclidean distance measure was applied on untransformed data, allowing PERMANOVA to return an equivalent test statistic to a standard ANOVA (Anderson et al. 2008). Post-hoc pair wise comparisons were used to test for differences among levels within significant factors. Results from univariate analyses were presented using plot of means and standard errors for each location.

For the analysis of infauna, benthic infauna assemblages (multivariate dataset) were first sorted to species level, before being consolidated to the family level. Multivariate assemblage data were square-root transformed to down-weight the contribution of dominant infauna and to allow intermediate or rarer groups to play a part in the analyses (Clarke 1993). A Bray-Curtis dissimilarity matrix was generated and the data were analysed using PERMANOVA.

Results of multivariate analysis were presented graphically using nMDS. This enabled the top ten benthic infauna families that had the strongest correlations with the patterns in the multivariate data to be determined. The top ten benthic families were then presented using pie charts to represent the overall percentage contribution for each season and location. For univariate analyses of infauna abundance and family richness, a Euclidean distance measure was applied on untransformed data, allowing PERMANOVA to return an equivalent test statistic to a standard ANOVA (Anderson et al. 2008). Post-hoc pair wise comparisons were used to test for differences among levels within significant factors. Results from family richness and abundance analyses were presented using bar graphs of means and standard errors for each location.

To examine the relationship between infauna community assemblage and sediment parameters (grain sizes, trace metals, nutrients), a Canonical Analysis of Principal Coordinates (CAP) ordination plot of the community assemblage were graphed with vectors overlayed on the CAP ordination plot of sediment parameters. This enabled the top sediment parameters that had the strongest correlations with the patterns in the multivariate infauna data to be determined.

4.2.3 Program sensitivity

Both the water and the sediment monitoring programs were designed according to the MBACI (Multiple-Before-After-Control-Impact) framework of Keogh and Mapstone (1997). The sensitivity of MBACI designs is generally constrained by the number of locations (both impact and reference) and in some cases, the number of sites nested in locations (Underwood and Chapman 2003). The statistical power of MBACI designs cannot be calculated directly, but can be estimated using Monte Carlo simulations (Underwood and Chapman 2003). While the power of these designs was not tested during the EIA, the use of up to four impact locations and four reference locations compares well with other studies with reasonable levels of sensitivity (capable of detecting changes of between 20-40%) and acceptable levels of statistical power (~0.8; BMT Oceanica, unpublished data).

4.3 Baseline benthic habitat surveys

This assessment utilised two sources of benthic habitat data: historical and publically available data sets captured in 2003, 2006 and 2008 (by the University of Western Australia Marine Futures Project) and more recent data captured by DoF during the baseline assessment between May 2015 and March 2015 (Section 4.3.2). The habitat descriptions and proportional estimates in Section 5.5 are for the MWADZ study area which incorporates an area of 4750 ha (Figure 4-4). These differ from the descriptions in Section 6, which are based on a Local Assessment Area (LAU) of 6735 ha, determined in consultation with the OEPA.

4.3.1 Historical assessments

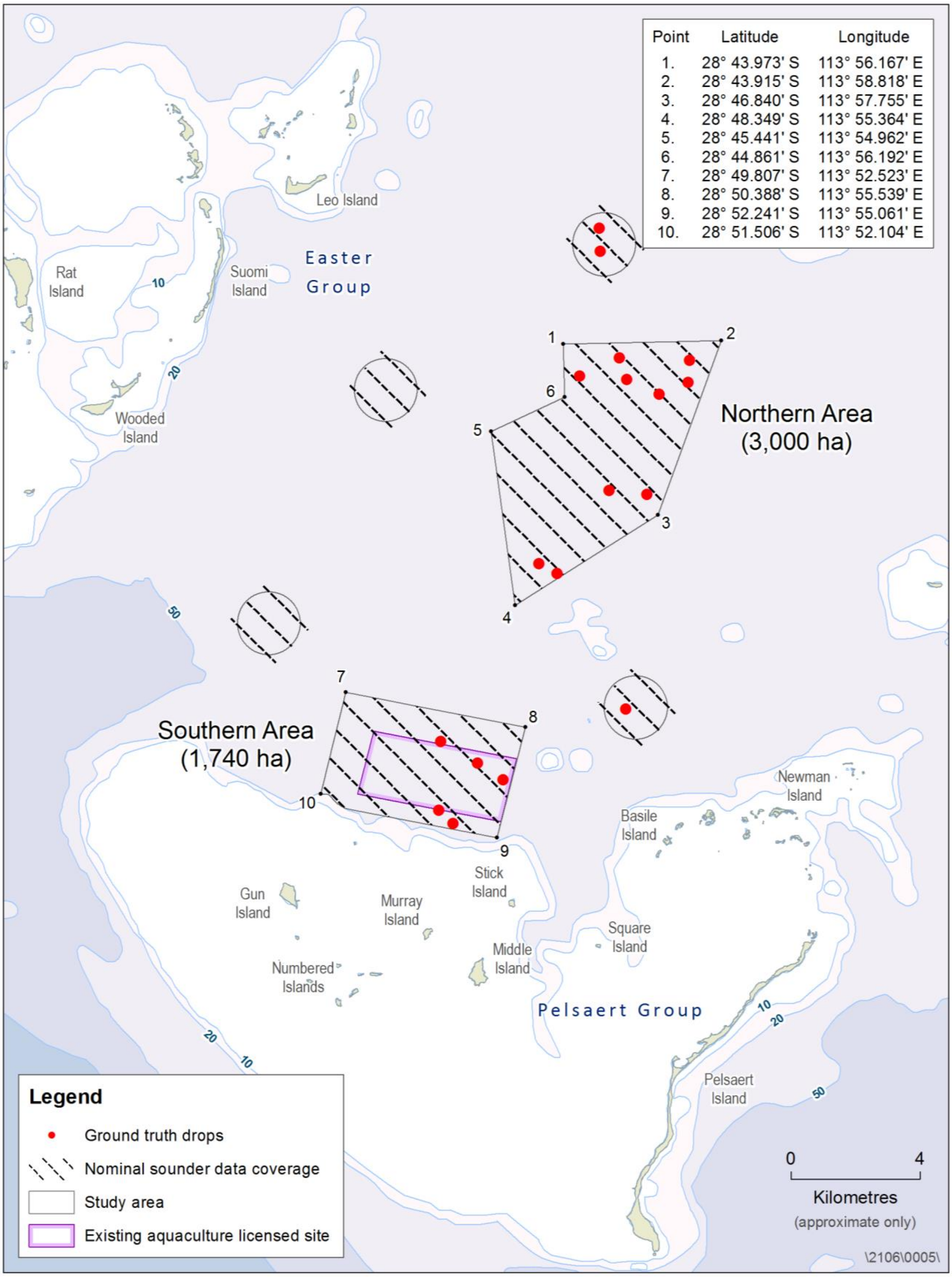
The 2003 surveys utilised sidescan sonar to map habitat in the southern group of the Abrolhos and the 2006 and 2008 surveys habitats north of the Pelsaert Group. The signal from the sidescan sonar was digitised using SonarWiz equipment and software from Chesapeake Technologies. Processing of the sidescan sonar data consisted of bottom tracking, beam angle correction and slant range correction and mosaiking. The data was analysed to classify benthos into broad categories, which were further defined by a total of 22 subcategories. All data was compiled in ArcView 8.2 GIS.

4.3.2 Surveys undertaken for this project

The current assessment utilised a Biosonic MX digital single beam echosounder and covered both the northern and the southern areas of the MWADZ and the reference locations. The sounder was fixed to the hull of the operational vessel and linked to a differential Global Positioning System (DGPS). The DGPS system provided sub metre accuracy through corrections via the OmniSTAR satellite service.

Depth data were collected 16–19 May 2014 along a xyz configuration of latitude, longitude and depth. East to west transect lines, spaced ~1 km apart were surveyed through both of the MWADZ locations and four reference areas. Sounding data was collected at a rate of 5 sounding records per second, with the boat travelling at approximately 5 knots.

The hydroacoustic surveys were conducted along approximately east-west lines through each area, based on the prevailing conditions, in an effort to minimise the pitch/roll of the vessel during the May 2014 sampling period. The first phase of soundings were spaced ~1 km apart (Figure 4-4) to capture a minimum level of hydroacoustic data for each area. The total linear distance covered was 7 900 meters for the first phase. The second phases of surveys involved infilling the 1 km spaced survey lines (Figure 4-4).



Point	Latitude	Longitude
1.	28° 43.973' S	113° 56.167' E
2.	28° 43.915' S	113° 58.818' E
3.	28° 46.840' S	113° 57.755' E
4.	28° 48.349' S	113° 55.364' E
5.	28° 45.441' S	113° 54.962' E
6.	28° 44.861' S	113° 56.192' E
7.	28° 49.807' S	113° 52.523' E
8.	28° 50.388' S	113° 55.539' E
9.	28° 52.241' S	113° 55.061' E
10.	28° 51.506' S	113° 52.104' E

Figure 4-4 Nominal sounder data tracks and location of ground truth sites

The resulting data was used to create an ‘unsupervised’ classification of the benthos to broad categories of benthos in the surveyed areas.

The unsupervised classification was used to select ground truthing sites for verification via drop video in the field during the June 2014 sampling period (Figure 4-4). The underwater video was a 'live feed' system consisting of a progressive scan camera in an underwater housing attached to weighted frame with legs (the weighted frame keeps the system directly below the vessel, while the legs provide protection and also a scale reference in the image). The system was connected to the vessel by 10 mm rope and a reinforced video umbilical cable. The live feed video, with DGPS overlay, was recorded onto a hard drive recording device or progressive scan HandyCam.

The video data were processed by using the point intercept method to identify the benthic habitats at each sampled site. The benthos was classified into several broad categories, encompassing reef, mixed assemblages (sparse, mixed) and sand. Within these broad categories, the percentage cover of macroalgae, sponges, hard corals and rhodoliths was determined.

Percentage cover of each habitat type, latitude, longitude and depth were recorded for each video drop site. These data were then analysed to determine homogenous habitat types to provide the basis for the supervised classification of the habitat. A classification of 'mixed assemblage' consisted of two or more biotic categories within one location (e.g. filter feeders, macroalgae and rhodoliths).

Data Analysis

All depth data was exported from the 'Biosonic MX digital single beam echo sounder' into Microsoft Excel. All data was collected and analysed in spatial reference datum WGS84. For analysis, depth data was averaged over 50 sounding records (~ every 30 m). Averaged depth data were then corrected to lowest astronomic tide (LAT) using tide information from the Bureau of Meteorology (BoM; see http://www.bom.gov.au/australia/tides) for Geraldton. The Geraldton tide data were used for tide correction as it is measured data, where the Pelsaert and Easter group tide data predicted and may not be accurate. However, variation in tide within a 30 minute period (the longest predicted tide variation at the Abrolhos Islands) at the Geraldton real time tide station fluctuates up to ±0.05 m at Geraldton. Therefore tidal difference between Geraldton and the MWADZ were expected to be minimal.

Digital Elevation Map

The digital elevation model for bathymetry of the MWADZ was developed using the averaged tide corrected depth data in the ArcGIS program ArcMap© using the spatial analyst extension. The 'Topo to Raster' tool was chosen as it is a proven best-practice interpolation method which is specifically designed for the creation of hydrologically correct digital elevation models. An individual model was run for each of the northern and southern areas of the MWADZ and the reference locations, with an output cell size of 50 m. The outputs provided are three interpolated surface rasters of bathymetry for the MWADZ northern, southern and reference areas. Each surface raster has cells with a pixel size of ~50 m, providing a depth data point for each cell within each location.

4.4 Pressure-response relationships

A key component of the EIA was to accurately identify and describe the cause-effect-response pathways relevant to the proposed MWADZ. The oceanographic and ecological components of the MWADZ are described in Section 3. Section 4.4 follows on from Section 3 to provide an overview of the ecological changes which may result from the proposal. To fully appreciate the risks posed by the MWADZ, it was first necessary to understand the types of pressures (and their magnitude) imparted by the proposal, and their likely effect (Section 4.4.1). This understanding, together with a desktop risk evaluation, was subsequently used to identify the key cause-effect-response pathways (Section 4.4.3), and to select thresholds for model interrogation (Section 4.5).

4.4.1 Identification of relevant pressures and risks

Noise

Noise generated by anthropogenic activities has the potential to disturb marine and terrestrial fauna, causing temporary or long-term avoidance of an area that may be important for feeding, reproduction or shelter. Underwater sounds may interfere with communication systems of fish and marine mammals, masking important biological cues or causing behavioural disturbance (Richardson et al. 1995, National Research Council 2005, Southall et al. 2007). Depending on the duration and intensity of underwater noise, an animal may avoid the source of the disturbance completely, thereby altering the overall use and ecology of that marine environment.

Construction and demolition of aquaculture facilities may, in rare circumstance, involve the use of pile-drivers or explosives (Olesiuk et al. 2012). These generate intense sounds, as well as shock waves that may affect critical behaviours and functions, such as feeding, migration, breeding and response to predators (National Research Council 2005; Yelverton et al. 1973; Yelverton 1981; Richardson et al. 1995; Dzwilewski and Fenton 2003; Madsen et al. 2006).

Acoustic Harassment Devices (AHDs) used to deter seal and sea lion attacks at salmon farms have been shown to have far ranging effects on non-target cetaceans, such as harbour porpoise and killer whales, which can be displaced large distances from where AHDs have been deployed. In contrast, pinnipeds (seals and sea lions) appear to habituate to these devices and may experience hearing loss through prolonged exposure or very close approach, such that AHDs are largely ineffective as long-term predator deterrents. AHDs could potentially disrupt the behaviour patterns of some fish that have specialized hearing apparatus, particularly clupeids like herring, but these effects have not been documented (Olesiuk et al. 2012).

Less intense sounds, such as those associated with vessel movements (i.e. movement of feeding barges and/or service vessels) would likely be in similar frequency and intensity ranges as those of commercial fishing and transport operations. For marine mammals at least, the effects of the sounds from these sources are usually transitory, or the animals can habituate to such sounds with regular exposure. However, the range of effects can be large, and the cumulative effects of the frequent exposure to louder vessels is largely unknown (Olesiuk et al. 2012).

Physical presence

Finfish will be grown in circular sea-cages (cages) of 38 m diameter and 18 m height (volume ~20 000 m^3). The design, construction and materials of cages will incorporate modern technology and best-practice to minimise environmental impacts. Cages will be anchored to the sea floor using equipment and techniques appropriate to marine conditions in the MWADZ. Where possible, anchoring on the sea-cages is undertaken with helix 'auger like' anchors which screw into the sea-floor. However, larger anchors, or weighted substrates (i.e. concrete blocks) might be required if the nature of the seafloor prevents penetration by the auger type anchors. Permanent losses of small areas of benthic habitat may occur in this instance.

The project infrastructure may act as a physical barrier to migrating marine life, an artificial substrate for attraction and roosting of seabirds (Section 8.4), and as a barrier to ambient water currents. The presence of large networks of sea-cages may in some circumstances act as a barrier or deterrent to cetacean migration (Section 8.3). Placement of sea-cage structures should proceed based on a review of the significance of the region as a migration corridor, as well as the likelihood that the configuration and placement of the infrastructure may act as a barrier. Ideally cage and/or lease placement should be organised to avoid such interactions.

Networks of floating sea-cages act as fish attractants and artificial substrates for marine invertebrates and sea-birds. For seabirds, direct disturbances may result from adverse interactions while foraging, attraction to, or avoidance of, aquaculture vessels and marine infrastructure, or exposure to contaminants. Direct interactions with finfish farming operations could include:

- supplementary feeding from stock predation, fish food, waste material or food scraps
- collisions with sea cages, other structures or vessels moored at night
- attraction and disorientation due to inappropriate lighting on service vessels, pens or navigation markers at night
- entanglement in cage mesh, predator nets or protective bird netting
- attraction of prey to vessel or sea cages due to “FAD” effects.
- attraction to the fish stock
- use of vessel or sea cages as roosting sites

In addition, floating sea-cages may affect local hydrodynamics. Model results show that the presence of fish cages restricts water flow and reduces the velocity in the surface layer occupied by the cages, but enhances the water velocity in the bottom layer beneath the cages. Increases in current speeds beneath sea-cages are dependent on distance between the bottom of the sea cages, and the seafloor. Bottom currents are maximised where the height of the cages is roughly half of the maximum water depth (Wu et al. 2014).

Organic wastes

The cause-effect-response pathways relevant to inputs of organic waste are a key consideration in this assessment. 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, regional eutrophication. Gray (1992) emphasises that the critical effects of eutrophication are experienced when water column oxygen concentrations become depleted as total community respiration increases due to increased organic loads to the sediments. Increased nutrient loadings are generally associated with increased episodes of hypoxia or anoxia, particularly in stratified waters, with subsequent detrimental effects on the fauna (Baden et al. 1990, Schaffner et al. 1992). Hypoxia may cause 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, Josefson & Jensen 1992, Hargrave et al. 2008; Hargrave 2010). Changes in communities are typically driven by the sensitivities of infauna, with rare and more sensitive species disappearing first. More resilient species such polychaetes are known to be resistant to hypoxic or near-hypoxic conditions (Pearson & Rosenberg 1978, Gray 1992, Dauer et al. 1992).

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, Dimitriadis & Koutsoubas 2011). Impacts to infauna commonly occur along a gradient of sediment organic enrichment (Pearson and Rosenberg 1978, Hargrave 2010), as evidenced by numerous studies demonstrating a correlation between the level of organic enrichment and the level of infauna community degradation. Cromey et al. (1998) reviewed the fate and effects of sewage solids added to mesocosms. Organic loading rates less than 36 g C/m^2/yr had little effect, rates between 36 and 365 g C/m^2 /yr enriched the sediment community, and a loading over 548 g C/m^2/ yr produced degraded conditions (Kelly & Nixon 1984, Frithsen et al. 1987, Oviatt et al. 1987, Maughan and Oviatt 1993, all cited in Cromey et al. 1998). Eleftheriou et al. (1982) showed that the addition of 767 g C/m^2/yr to unpolluted sediment enriched the fauna, whereas addition of 1 498 g C/m^2/yr caused degraded conditions. Deposition rates

>700 g C/m^2/yr are widely believed to represent a critical value, such that sediments exposed to this rate of deposition are considered degraded, i.e. diversity of benthic fauna is significantly reduced (Cromey et al. 1998). Although useful in terms of predicting the magnitude of effect of infauna, these thresholds give no indication of recovery times (also known as remediation) following removal of the source of the contaminants.

Although finfish farming has the potential to impact sediments beneath, and immediately adjacent to sea-cages (Carroll et al 2003), case studies of finfish aquaculture systems in Tasmania and Europe found that impacts are generally restricted to within 10–100 m of sea-cages and that the magnitude of impact depended largely on the depth of the water and the rate of water movement through the site (Carroll et al. 2003, Crawford 2003, Borja et al 2009). Average current velocities through the proposed MWADZ are 8.7–14.1 cm/s in the summer months, and 10.5–14.5 cm/s in the winter months (Table 4.5). This range of average current speeds is conducive to conditions described as either 'moderately' or 'not sensitive' to impact. Currents speeds >10 cm/s are widely considered 'ideal' for sea-cage aquaculture, and current speeds <6 cm/s are generally considered 'not ideal' for sea-cage aquaculture (Table 4.6).

Table 4.5 Average surface and bottom water current speeds through the MWADZ

	Current speeds (cm/s)			
	Northern area		Southern area	
Month	Surface	18 m water depth	Surface	18 m water depth
Summer	13.2-14.1	10.4-11.0	8.7-9.4	5.8-7.0
Winter	14.0-14.5	9.0-11.5	10.5-11.0	6.1-8.0

Table 4.6 Increasing suitability of potential aquaculture sites based on current speed

Suitability	Current speed (cm/s)	Reference
Not sensitive to impact / desirable	10-25	Carroll et al. (2003)
	>15	Borja et al. (2009)
	13–77	Benetti et al. (2010)
	5–20	Halide et al. (2009)
	10–60	Beverage (2004)
Moderately sensitive to impact	5–15	Borja et al. (2009)
Sensitive to impact / unsuitable	3–6	Carroll et al. (2003)
	<5	Borja et al. (2009)

Inorganic nutrients

Finfish aquaculture in open water sea-cages may, in some instances, cause deterioration in local water quality due to inputs of inorganic nutrients from fish faeces and uneaten food. Aquaculture may contribute inorganic nutrients to the water column either directly through secretion of ammonia by fish, or indirectly through organic matter deposition and remineralisation. Inorganic nutrients in the form of ammonia, nitrite + nitrate and orthophosphate may lead to adverse environmental effects via a number of cause-effect pathways, all of which contain BPPHs as key receptors. As with the cause-effect-response pathways relevant to organic wastes (described above), the cause-effect-response pathways relevant to inorganic nutrients are also considered key in this assessment.

Habitat studies in the MWADZ have revealed a diverse array of benthic habitats, including the presence of vast swathes of mixed assemblages comprising macro-algal, rhodolith, filter feeding, coral and other invertebrate communities (Section 5.4.5). Macroalgae and corals in particular are known to be sensitive to sources of inorganic nutrients, and may in worst-case examples undergo phase shifts. For example, prolonged exposure to nutrients may lead to conditions where living corals are slowly replaced by macroalgae. Some authors believe that phase shifts are dependent on the degree of herbivory on a reef system (e.g. Littler & Littler 1984, Jackson et al. 2001, Bellwood et al. 2004, Hughes et al. 2010, Rasher et al. 2012). The paradigm is that in the absence of herbivores, algae have been able to proliferate even at low nutrient concentrations (~1 µmol/L).

Metals and other contaminants

Toxic effects on marine organisms are likely when metal concentrations reach threshold levels, or increase via biomagnification (Parsons 2012). Sources of metals include contaminated sites, agricultural and urban runoff, discharges from sewage treatment plants, and copper-based antifoulants sometime used on sea-cage infrastructure (Parsons 2012).

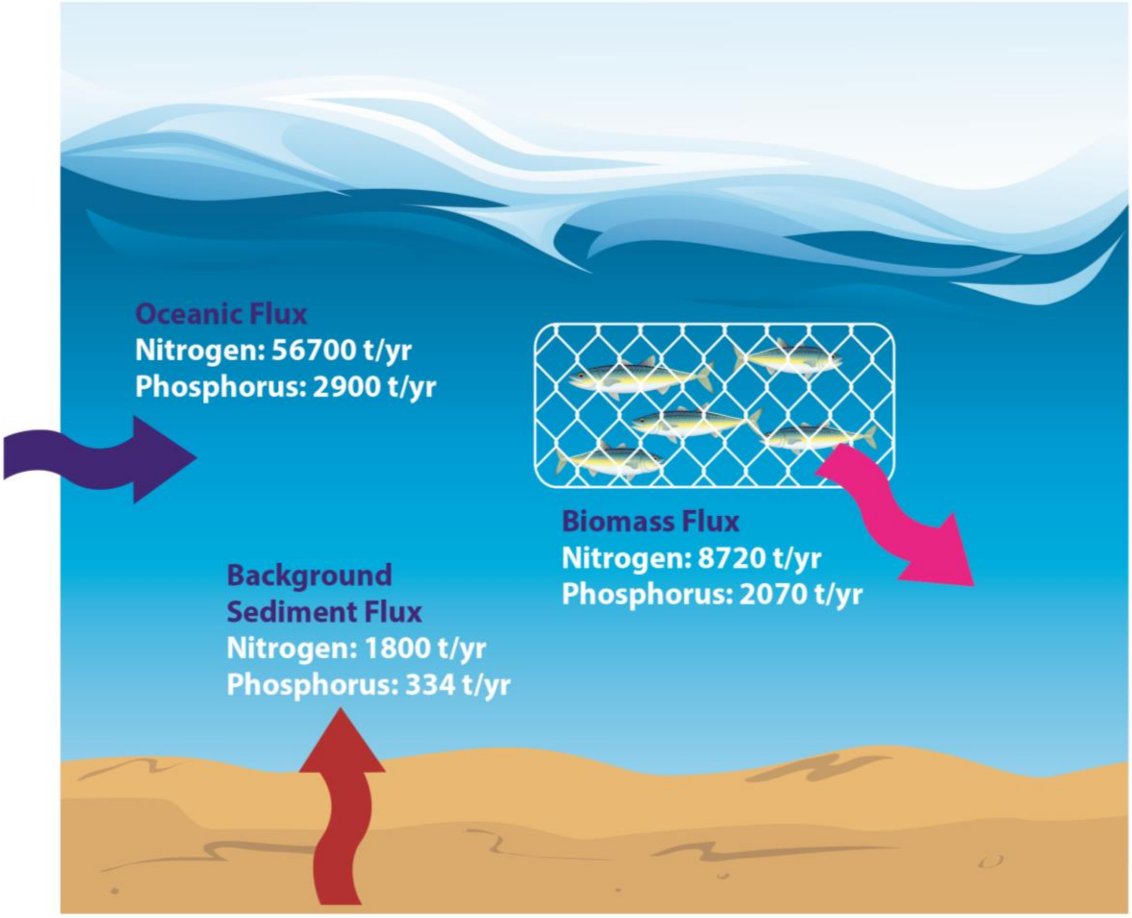
Metals form a small constituent of commercial aquaculture feeds as trace elements. The metals are consumed by finfish and excreted in the faeces. A study of the metal content of trout faeces by Moccia et al. (2007) found that Zn and Fe were present in the highest concentrations, with relatively low proportions of copper (see Section 7.3.3). Despite the very low concentrations in commercial feeds, monitoring in Tasmanian waters has recorded copper and zinc sediment values at concentrations higher than the ANZECC/ARMCANZ (2000) ISQG-low and ISQG-high guideline values at some sea-cage sites (DPIPWE 2011).

Antibiotics are sometimes used to treat bacterial disease occurring in farmed finfish and are generally administered in feed. Antibiotics may impart pressure on the marine environment by reducing or changing numbers of sediment bacteria, which in turn may affect broader ecological processes. In the treatment of farmed salmon in Tasmania, oxytetracycline is the most common antibiotic used, accounting for more than 70% of total antibiotic use during 2006–2008 (Parsons 2012). A strong seasonal component to the use of antibiotics has been noted in Tasmania, with the greatest requirement in the summer months when water temperatures are elevated and pathogens tend to be most virulent.

4.4.2 Ecosystem nutrient budget

The nutrient budget of the region is relatively simple in that it comprises (presently) only advective oceanic fluxes and sediment nutrient fluxes. These are both considered small in that the existing environment is essentially oligotrophic. Supporting this, it is noted that the monitoring data collected as part of this study showed that water column nutrient concentrations were generally very low (Section 5.3.3).

The addition of the proposed fish cages adds a considerable nutrient perturbation to the system, and has been a key subject of investigation in this study. This perturbation takes the form of both an immediate nutrient load to the water column (via waste and feed excess) and a delayed load via impacted sediment nutrient remineralisation. A graphical representation of existing and impacted conditions, with approximate annual nutrient fluxes is included in **Error! Reference source not found.** and Table 4.7. Fluxes have been computed from measurements and model predictions.



Notes:

1. Biomass flux includes both solid and liquid waste nitrogen and phosphorus
2. Sediment flux is the background flux for the southern Abrolhos region (~3,000 km^2); sediment flux is based upon the average sediment nutrient content measured during the baseline sampling program
3. Oceanic flux is the total nutrient flux in and out of the southern Abrolhos region (~3,000 km^2)

Figure 4-5 Conceptual diagram of the baseline and post operation nutrient budget under scenario 1

Table 4.7 Baseline and post operation nutrient budgets

Scenario	Source (t/yr)		
	Aquaculture (biomass)	Oceanic	Background sediment
1-2	Nitrogen 8720 Phosphorus 2070	Nitrogen 56 700 Phosphorus 2900	Nitrogen 1800 Phosphorus 10700
3-4	Nitrogen 13950 Phosphorus 3310		
5-6	Nitrogen 17440 Phosphorus 4130		

4.4.3 Cause-effect-response pathways

Cause-effect-response pathways were developed following the step-wise approach of Gross (2003). The approach included development of two models: a control model and a stressor model. The control model (Figure 4-6) is hierarchical in nature, with the stressors and their sources shown in the upper strata of the model, and the indicators (receptors) and effects shown in the middle to bottom strata of the model. The control model remains relatively simple in that it makes no attempt to account for the magnitude and/or the duration of the stress.

The stressor model is a refined version of the control model focussing on the cause-effect pathways of most concern (Figure 4-7). It articulates the relationship between stressors, ecosystem components, effects and biological receptors and is a succinct account of the major cause effect pathways, from which the indictors and thresholds were ultimately derived.

The objective of this approach was to identify the cause-effect-response pathways most likely to be affected by the MWADZ, and those likely to exhibit measurable changes in response to stressor inputs. The understanding gained by this process was used to develop the thresholds described in Section 4.5.

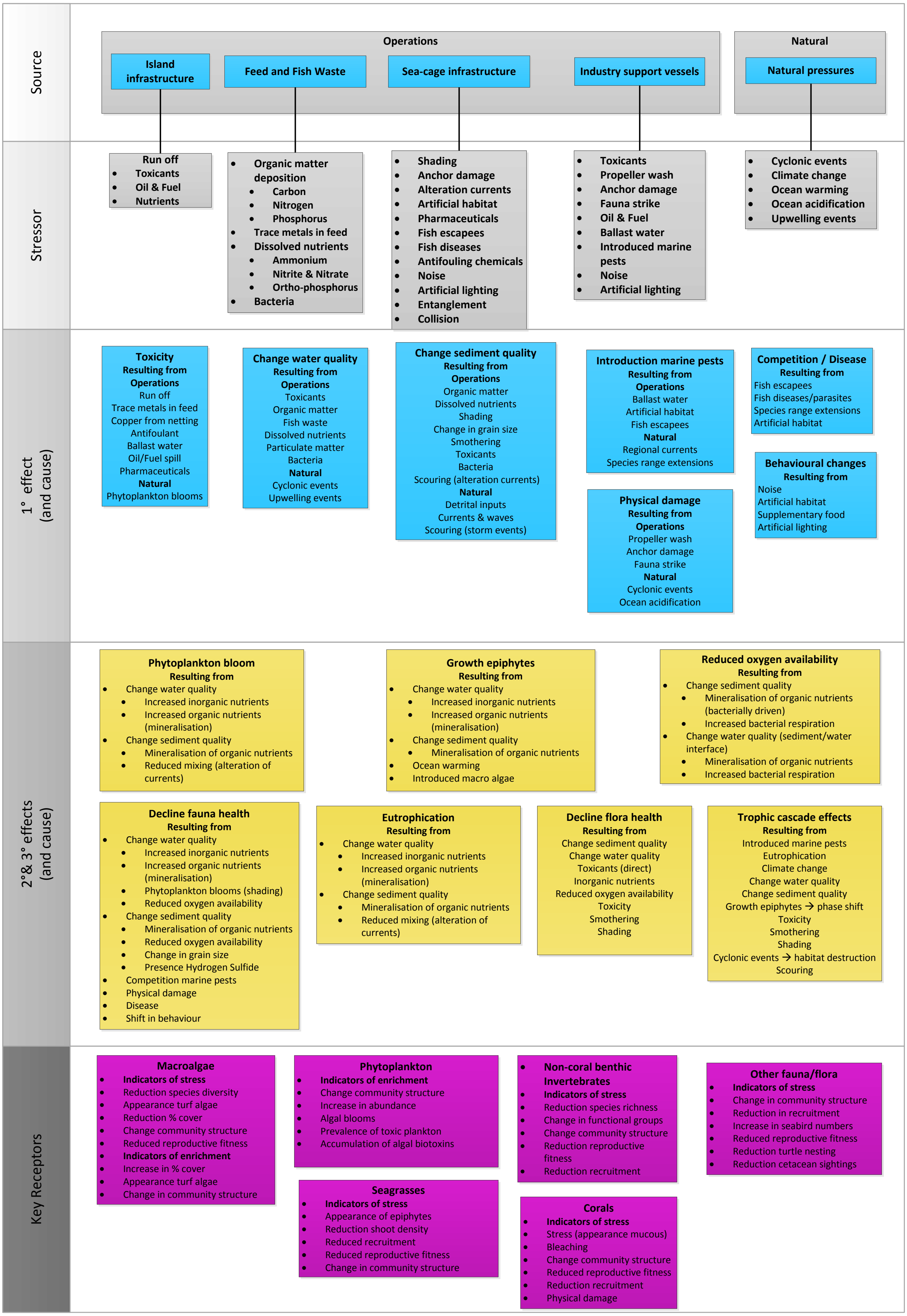
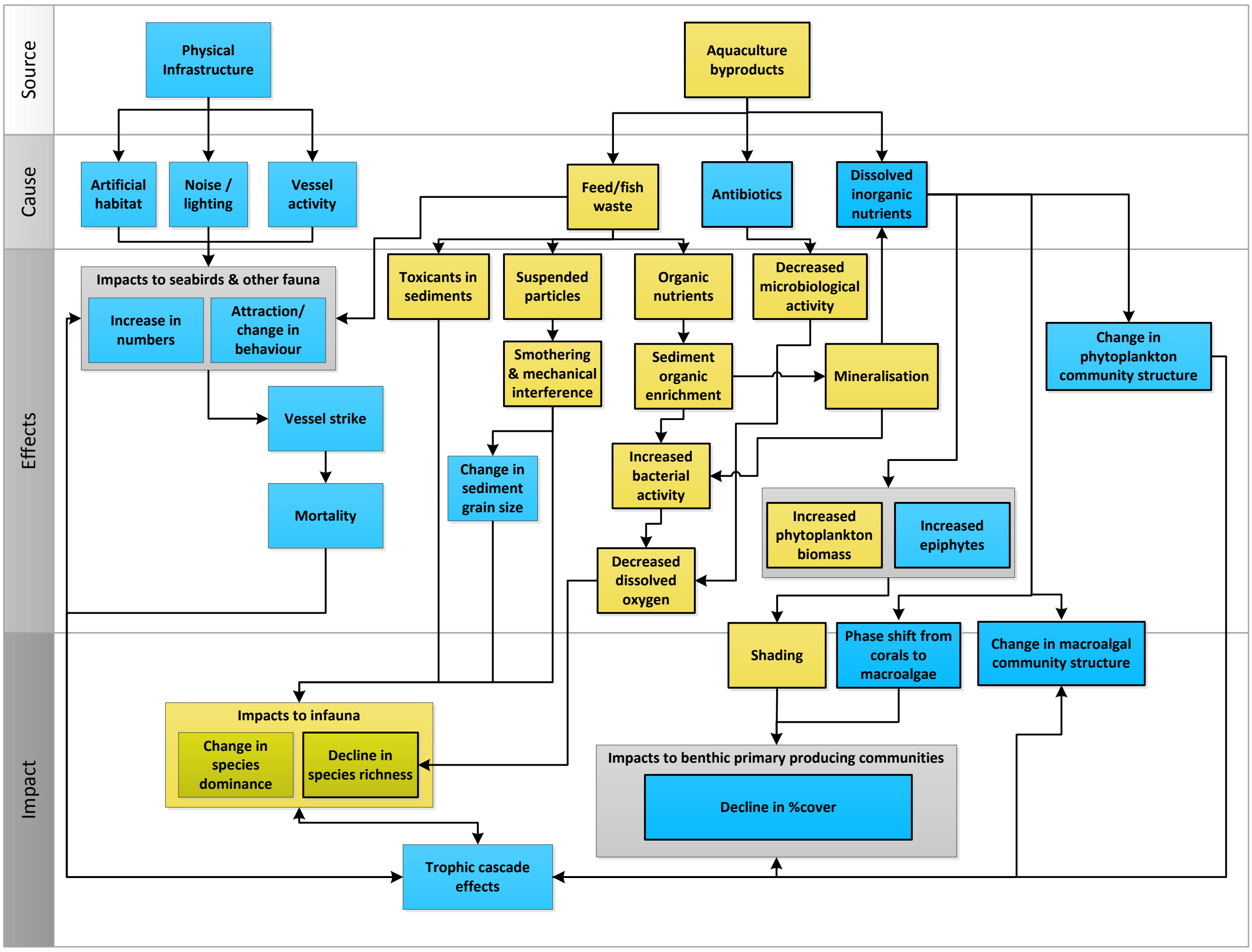


Figure 4-6 Hierarchical control model showing natural and anthropogenic stressors and key cause-effect-response pathways



Notes:

1. Key cause-effect-response pathways. Pathways shown in yellow represent those captured by the modelling and those for which thresholds were developed.

Figure 4-7 Hierarchical stressor model showing the key cause-effect-response pathways and those chosen for model interrogation

4.5 Thresholds for model interrogation

4.5.1 Application of EAG 3

EAG 3 is concerned with the protection of ecological integrity and biodiversity through a framework for assessing the cumulative loss of, and/or serious damage to benthic primary producer habitats (BPPH) in WA. BPPHs are seabed communities within which algae (e.g. macroalgae, turf and benthic microalgae), seagrass, mangroves, corals or mixtures of these groups are prominent components. BPPHs also include areas of seabed that can support these communities (EPA 2009).

'Irreversible loss' of benthic primary producer habitats is commonly associated with excavation or burial. Such activities modify BPPH so significantly that the impacted community would not be expected to recover to the pre-impact state and therefore the loss is considered irreversible. ‘Serious damage’ is also intended to apply to damage to BPPH that is effectively irreversible or where recovery, would not occur for at least 5 years (EPA 2009).

Applicable category

EAG 3 was applied here given the potential for sea-cage aquaculture to cause both permanent loss and serious damage. Both are hereafter termed cumulative loss.

EAG 3 provides guidelines which outline cumulative losses of BPPHs that may be acceptable, provided all other options have been exhausted. The waters of the Abrolhos Islands, including the MWADZ, are gazetted as a Fish Habitat Protection Area (FHPA) under section 115 of the *Fish Resources Management Act 1994*. The FHPA has the following purposes:

1. conservation and protection of fish, fish breeding areas, fish fossils or the aquatic ecosystem
2. culture and propagation of fish and experimental purposes related to that culture and propagation, or
3. management of fish and activities relating to the appreciation or observation of fish.

The Management Plan for the FHPA does not identify any areas of high conservation value that would be category A; therefore the proposed MWADZ should be category C. The Cumulative Loss Guidelines (EAG 3) recommend that cumulative loss of BPPH within areas deemed to be Category C do not exceed a benchmark of two percent of the BPPH within the LAU (Table 4.8).

Table 4.8 Cumulative loss guidelines for benthic primary producer habitat within defined local assessment units

Category	Description	Cumulative loss guideline[1]
A	Extremely special areas	0%
B	High protection areas other than above	1%
C	Other designated areas	2%
D	Non-designated area	5%
E	Development areas	10%
F	Areas where cumulative loss guidelines have been significantly exceeded	No net damage

Note:

1. Defined as a percentage of the original area of benthic primary producer habitat within a defined local assessment unit

4.5.2 Application of EAG 7

The potential for the MWADZ to impart adverse effects on the benthic marine environment (particularly soft sediments) were described in the context of EAG 7. EAG 7 includes three predefined levels of impact: zone of high impact (ZoHI), zone of moderate impact (ZoMI) and zone of influence (ZoI) (Table 4.9). EAG 7 was developed to assess the impacts of capital dredging activities to benthic habitats in the State's Northwest, and its application to aquaculture EIA is new (see DHI 2013).

Table 4.9 Zone of impact criteria from EAG 7

Zone	Criteria
Zone of high impact (ZoHI)	The area where impacts on benthic organisms are predicted to be irreversible. The term irreversible is defined in accordance with EPA (2009) as 'lacking a capacity to return or recover to a state resembling that prior to being impacted within a timeframe of five years or less'. Areas within and immediately adjacent to proposed dredge and disposal sites are typically within zones of high impact. The irreversible loss of the benthic primary producer habitats within these zones should be considered in the context of Environmental Assessment Guideline No. 3 (EPA, 2009), unless a defensible case for recovery of the impacted benthic primary producing habitat can be presented.
Zone of moderate impact (ZoMI)	The area within which predicted impacts on benthic organisms are sub-lethal, and/or the impacts are recoverable within a period of five years following completion of the dredging activities. This zone abuts, and lies immediately outside of, the zone of high impact. Proponents should clearly explain what would be protected and would be impacted within this zone, and present an appraisal of the potential implications for ecological integrity of the impacts over the timeframe from impact to recovery (e.g. through loss of productivity, food resources, shelter). Where recovery from the impact predicted in this zone is likely to result in an 'alternate state' compared with that present prior to development, then this outcome should be clearly stated in environmental assessment documents, along with justification as to why the predicted impacts should be included within this zone (rather than the zone of High Impact) and an appraisal of the potential consequences for ecological integrity. The outer boundary of this zone is coincident with the inner boundary of the next zone, the zone of Influence.
Zone of influence (ZoI)	The area within which changes in environmental quality associated with dredge plumes are predicted and anticipated during the dredging operations, but where these changes would not result in a detectible impact on benthic biota. These areas can be large, but at any point in time the dredge plumes are likely to be restricted to a relatively small portion of the zone of Influence. The outer boundary of the zone of Influence bounds the composite of all of the predicted maximum extents of dredge plumes and represents the point beyond which dredge-generated plumes should not be discernable from background conditions at any stage during the dredging campaign. Furthermore, this provides transparency for the public regarding where visible plumes may be present, albeit only occasionally, if the proposal receives approval. Reference sites for monitoring natural variability would ideally be located outside of the zone of Influence of the dredging activities.

Soft sediments

The recovery of sediments at the point of fallowing was determined directly using a sediment diagenesis (biogeochemical) model, linked to a hydrodynamic and a particle transport model. The period of recovery was determined across a range of scenarios. Conditions were simulated in which sediments, beneath and near the sea-cages, received inputs of waste for a period of two, three and five years. At the completion of the two, three and five year periods, the cages were fallowed, and the sediments allowed to recover.

Oxygenation

Recovery was deemed to have occurred when sediment chemical conditions, represented by the concentration and depth of oxygenation and hydrogen sulphide, returned to pre-aquaculture conditions (Table 4.10). Three zones were defined based on threshold criteria for recovery (defined in more detail in Appendix G). This included consideration of oxygen and sulphide concentrations within the top 5 cm of sediment. The ZoHI was applied when sediment conditions took greater than 5 years to recover; the ZoMI was applied when sediment conditions took less than 5 years to recover, and the ZoI was applied when sediments received waste material, but not in proportions great enough to alter the sediment chemistry. Chemical recovery was used over biological recovery, as its trajectory is more reliable and it has readily identifiable beginning and end points. Biological recovery, in contrast, may never occur completely as guilds of infauna inhabiting similar ecological niches may replace each another, leading to subtle differences in post remediation community structures – meaning the end point is difficult to quantify.

Metals
Recovery thresholds were based on the time taken for sediment metal concentrations to return to values lower than the EPA's Environmental Quality Guideline (EQG) trigger values (EPA 2014). The ZoHI was applied when sediment conditions took greater than 5 years to recover and the ZoMI was applied when sediment conditions took less than 5 years to recover. The ZoI was applied when sediments received waste containing metals, but not in concentrations great enough to exceed the EQG trigger values.

Table 4.10 Thresholds applied to soft sediments

Parameter	Zone of high impact (ZoHI)	Zone of moderate impact (ZoMI)	Zone of influence (ZoI)
Hydrogen sulphide	Concentrations deteriorate and do not recover to baseline levels within a 5 year period	Concentrations deteriorate but recover to baseline levels within a 5 year period	Concentrations not to exceed baseline levels Top 5 cm of sediment remain oxygenated
Oxygenation			
Metals (Zn and Cu)1	Sediment concentrations of Zn and Cu do not recover to values lower than the EPA EQGs with a period of 5 years	Sediment concentrations of Zn and Cu recover to values lower than the EPA EQGs within a 5 year period	Sediment concentrations of Zn and Cu not to exceed the EPA EQGs

Notes:
1. Zinc (Zn) and Copper (Cu) are the metals present in feeds in the highest proportion and those with EPA (2015) triggers.
2. EQG = Environmental Quality Guideline

4.5.3 Application of other impact criteria

Mixed assemblages and the water column

Unlike soft sediments, for which it was possible to model recovery directly, the development of impact criteria for mixed assemblages and the water column required a different approach. The thresholds for smothering are based on PIANC (2010), and the thresholds for water column oxygenation, suspended particles, algal growth potential, nutrient enrichment and shading are based on EPA (2015). The EPA's criteria were used in lieu of the uncertainty regarding the lethal and sub-lethal thresholds of endemic species, and equal uncertainly regarding their timing of recovery, particularly following exposure to aquaculture stressors (i.e. organic material and inorganic nutrients).

Smothering
Thresholds for smothering are based on lethal and sub-lethal end-point triggers for corals published in PIANC (2010), and are the same as those used in the KADZ assessment (Oceanica 2013) (Table 4.11). The thresholds correspond to the levels of impact described in Table 4.12 which are based on the sensitivities of coral. These thresholds were originally developed for inorganic materials, but in the absence of comparative information, these thresholds were used as a best estimate.

Table 4.11 Thresholds based on PIANC (2010)

Effect	Major impact (ZoHI)	Moderate impact (ZoMI)	No impact (ZoI)
Smothering[1]	Sedimentation rate not to exceed 500 g/m^2/day	Sedimentation rate not to exceed 100 g/m^2/day	Sedimentation rate not to exceed 50 g/m^2/day

Notes:
1. Thresholds based on those developed for sensitive coral species by the PIANC Working Group 108 (2010)

Table 4.12 Impact assessment categories for the effects of smothering

Severity of impact	Description
Minor impact	Changes are likely to be detected in the field as localised mortalities, but to a spatial scale that is unlikely to have any secondary consequences.
Moderate impact	Changes are detectable in the field. Moderate impacts are expected to be locally significant.
Major impact	Changes are detectable in the field and are likely to be related to complete habitat loss. Major impacts are likely to have secondary influences on other ecosystems.

Suspended particles

Thresholds for suspended particles were developed to be consistent with the moderate and high levels of marine ecological protection described in EPA (2015) (Table 4.13). The thresholds are respectively based on the 95th and 80th percentile values obtained during baseline studies. In this context, the 80th percentile is in alignment with the criteria used for a high level of ecological protection and the 95th percentile a moderate level of ecological protection. For contextual purposes, Table 4.13 also outlines the limits of acceptable change under a low level of ecological protection. Low ecological protection areas are typically applied to ocean outfalls, where moderate and high levels of ecological protection are not always achievable.

Table 4.13 Levels of ecological protection

Level of ecological protection	Limits of acceptable change
Low	To allow for large changes in the quality of water, sediment and biota (e.g. large changes in contaminant concentrations causing large changes beyond natural variation[1] in the natural diversity of species and biological communities, rates of ecosystem processes and abundance/biomass of marine life, but which do not result in bioaccumulation/biomagnification in near-by high ecological protection areas).
Moderate	To allow moderate changes in the quality of water, sediment and biota (e.g. moderate changes in contaminant concentrations that cause small changes beyond natural variation in ecosystem processes and abundance/biomass of marine life, but no detectable changes from the natural diversity of species and biological communities).
High	To allow small changes in the quality of water, sediment or biota (e.g. small changes in contaminant concentrations with no resultant detectable changes beyond natural variation* in the diversity of species and biological communities, ecosystem processes and abundance/biomass of marine life).

Note:

1. Detectable change beyond natural variation nominally defined by the median of a test site parameter being outside the 20th and 80th percentiles of the measured distribution of that parameter from a suitable reference site

Water column

Oxygenation

The thresholds for oxygenation (dissolved oxygen; DO) are based on EPA (2015). The thresholds are equivalent to the Environmental Quality Guidelines (EQG) for achieving moderate and high levels of ecological protection (Table 4.13), which require that DO levels are maintained at 80% and 90% saturation respectively for a period greater than six weeks duration.

Table 4.14 Thresholds based on EPA (2015)

	Moderate ecological protection	High ecological protection
Oxygenation[1]	DO saturation in the bottom half of water column not to fall below 80% for a period exceeding 6 weeks	DO saturation in the bottom half of water column not to fall below 90% for a period exceeding 6 weeks
Suspended particles[2]	TSS concentration not to exceed 8.4 mg/L more than 50% of the time	TSS concentration not to exceed 2 mg/L more than 50% of the time
Algal growth potential[2]	DIN concentration not to exceed 40 µg/L more than 50% of the time	DIN concentration not to exceed 29 µg/L more than 50% of the time
Nutrient enrichment[2]	Chlorophyll-a not to exceed 0.45 µg/L more than 50% of the time	Chlorophyll-a not to exceed 0.30 µg/L more than 50% of the time
Shading[2,3]	Light intensity at the benthos not to fall below the 5th percentile more than 50% of the time	Light intensity at the benthos not to fall below the 20th percentile more than 50% of the time

Notes:

1. Thresholds for the ZoHI/ZoMI and the ZoI are based respectively on the EPA's EQGs for moderate and high ecological protection (EPA 2005). Threshold assumes continuous exceedance for a period exceeding six weeks.
2. Thresholds for the Zone of moderate impact (ZoMI) and Zone of influence (ZoI) are based respectively on the EPA's EQGs for moderate (95th percentile baseline data) and high (80th percentile baseline data) ecological protection (EPA 2015). The threshold for the Zone of high impact (ZoHI) is based on the 99th percentile of baseline data.
3. During daylight hours (8am–6pm).

Algal growth potential and shading

Thresholds for inorganic nutrients were developed to address the effects of algal growth potential, nutrient enrichment and shading (Figure 4-8). The thresholds for algal growth potential and nutrient enrichment are based on the 95th and 80th percentile values obtained during baseline studies (Section 5.3). The thresholds for shading by contrast are based on the 5th and 20th percentile values obtained during baseline studies. In this context, the 20th and 80th percentiles (ZoI) are in alignment with the criteria used for a high level of ecological protection; and the 5th and 95th percentiles, a moderate level of protection.

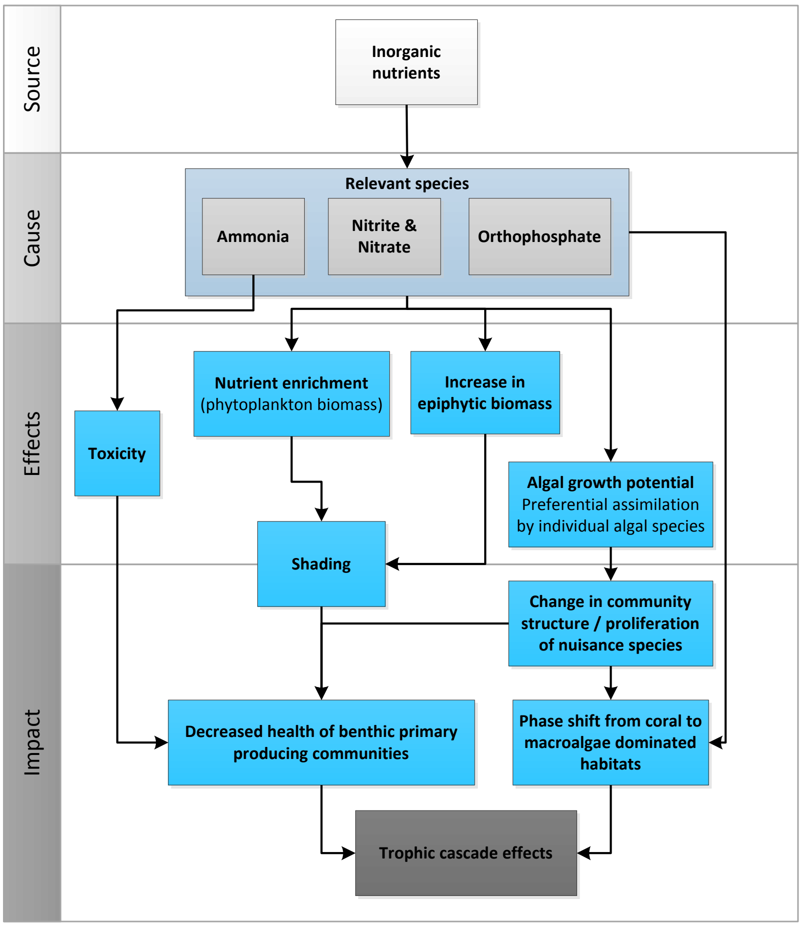


Figure 4-8 Cause-effect-response pathways relevant to inorganic nutrients

4.5.4 Aquaculture scenarios chosen for modelling

Modelling scenarios were agreed in consultation with the DoF and the Aquaculture Industry Reference Group at a technical workshop held in October, 2014. Scenarios were developed based on production of yellowtail kingfish (*Seriola lalandi*) using industry best-practice farming methods, including use of the standard infrastructure as described in Table 4.15.

Table 4.15 Aquaculture infrastructure assumptions

Infrastructure component	Details
Cage diameter (m)	38
Cage circumference (m)	120
Cage depth (m)	18
Cage volume (m^3)	20 641
No. cages per cluster	14
Other assumptions	• Two to three clusters in the southern location • Four to six clusters in the northern location • Percentage of uneaten feed = 1%

Six production scenarios were modelled in total (Table 4.16). All scenarios assumed constant stocking of between 15 000 and 30 000 tonnes standing biomass, and static Food Conversion Ratio (FCR) and Specific Growth Rate (SGR) values of 3.1 and 0.29% respectively (Section 4.6.1). No allowances were made for annual fluctuations in standing biomass due to growth and/or harvesting of stock. Feed inputs and waste outputs were also assumed to be constants in time. The effect on the benthic environment of increasing and decreasing stocking densities was examined by manipulating the number of cage-clusters between six and nine. This was undertaken in recognition of the economic-environmental trade-offs between infrastructure requirements and the aquaculture industries desire to maintain higher stocking densities, wherever resources and/or the biology of the target species allows. It is noted however, that the choice of cluster numbers was intended to balance the infrastructure proportionally across the two areas making up the proposed MWADZ, and not one intended to constrain the industry to that specific number.

Table 4.16 Modelled production scenarios

Scenario No.	S1	S2	S3	S4	S5	S6
Total standing biomass (t)	15 000		24 000		30 000	
Standing biomass north (t)	10 000		16 000		20 000	
Standing biomass south (t)	5000		8000		10 000	
No. clusters south	3	2	3	2	3	2
No. clusters north	6	4	6	4	6	4

Note:

1. t = tonnes

4.6 Approach to modelling

The ESD required development of an ecological/environmental model to predict the cumulative environmental effects of the proposal, operating across a range of production scenarios. To meet this objective, several models were developed, all of which were integrated to address the requirements of the ESD. The fully integrated model was capable of resolving the regional hydrodynamics, the deposition and dispersal of wastes from sea-cages, the effects of these wastes on the marine environment, and the rate of environmental recovery following cessation and/or relocation of the aquaculture activities. The approach to integrating the individual modelling components is summarised in Section 4.6.1, below, and the assumptions underpinning the modelling are summarised in Section 4.6.2. Full details, including the approach to calibration, are included in Appendix F and Appendix G.

4.6.1 Model integration

Hydrodynamic

The primary aim of the hydrodynamic model was to provide a realistic representation of currents and wave dynamics in the northern and southern areas, for determining the fate of wastes released from aquaculture activities (e.g. waste feed, inorganic nutrients and faecal material), and also to inform the sediment diagenesis and the water quality simulations. The model was calibrated against metocean and water quality data collected during the May 2014 to December 2014 period of the baseline sampling program. Validation was then undertaken by comparing model results against observations made during the December 2014 to March 2015 period of the baseline monitoring program (results of these processes are detailed in Appendix F). TUFLOW FV was used as hydrodynamic modelling engine (http://www.tuflow.com/Tuflow%20FV.aspx). It is capable of solving Non-Linear Shallow Water Equations (NLSWE) on a 'flexible' (unstructured) mesh comprising triangular and quadrilateral cells.

A digital elevation model (DEM) was developed using a regional bathymetry dataset from Geosciences Australia with 250 m resolution, and a higher-resolution dataset of the Abrolhos Islands from the WA Department of Transport. This was interrogated to provide bathymetry values to the model mesh. The model mesh covers an overall area of 2.7 million ha, with a single open boundary of ~413 km stretching from Kalbarri in the north to Leeman in the south. It includes 23 093 horizontal cells, ranging from resolution of ~3.5 km at the open boundary to approximately 40 m resolution within the proposed lease areas Figure 4-9 and Figure 4-10). A variety of cage configurations were included in the mesh to ensure that processes adjacent to cage clusters are highly resolved by the model. Sub-sets of these cage configurations were used developing the modelled scenarios (Section 4.5).

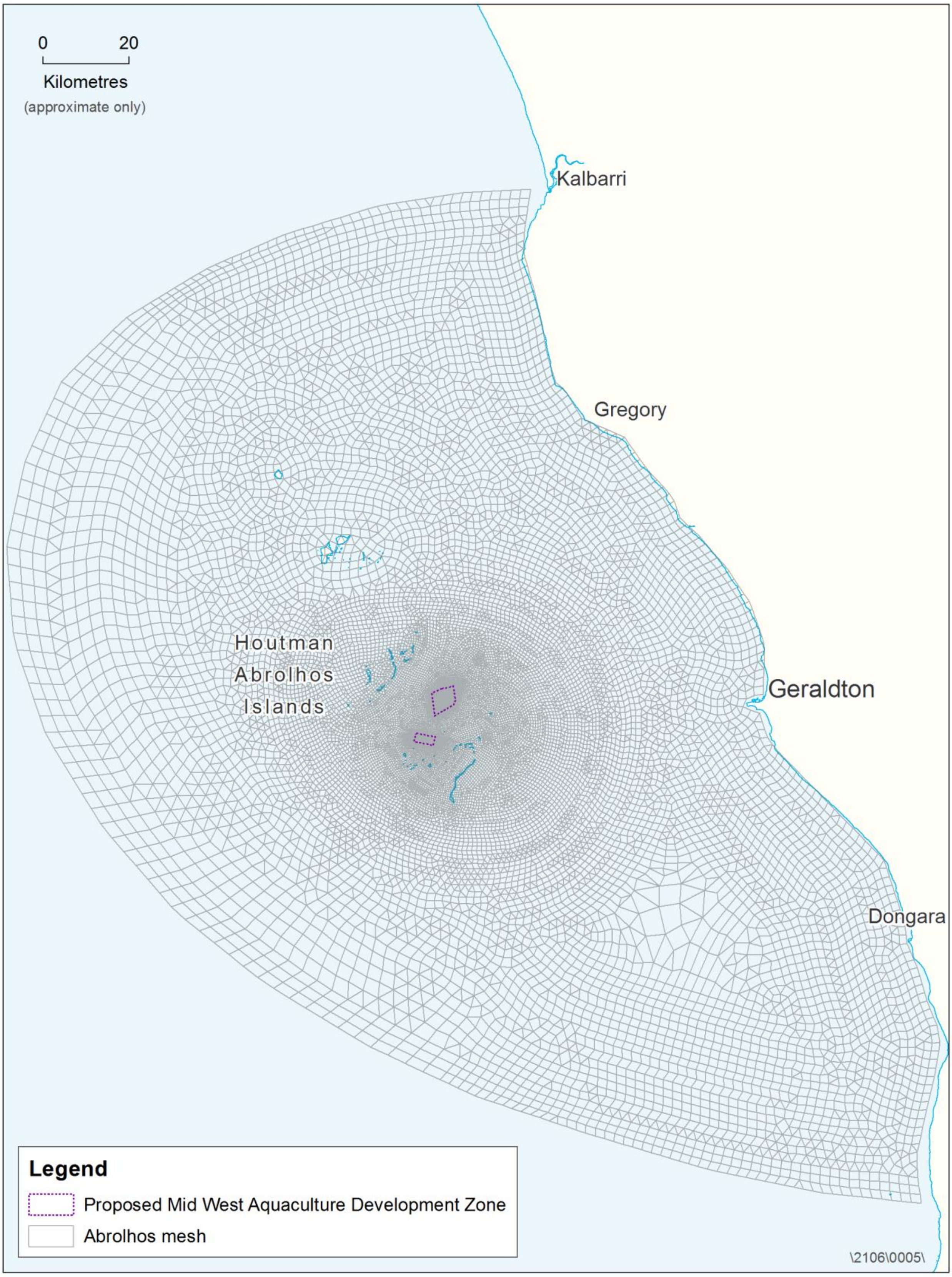


Figure 4-9 Full extent of the model mesh

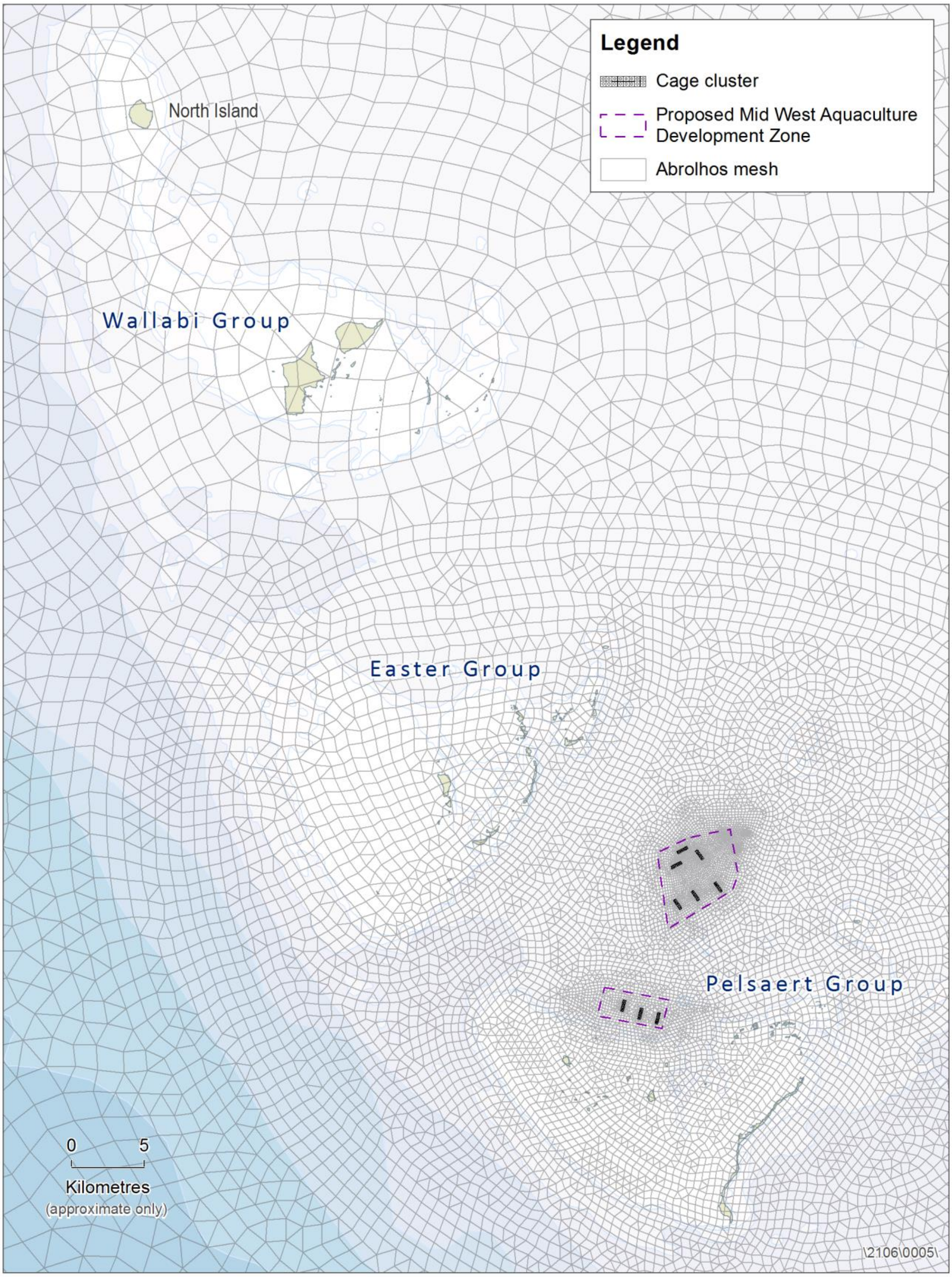


Figure 4-10 Zoomed in view of the model mesh

Wave model

To resolve potential wave-driven currents plus wave-induced drift and to capture suspension/deposition dynamics driven by waves, a wave field was applied to TUFLOW FV using the model SWAN. SWAN is a third-generation wave model, developed at Delft University of Technology, which computes random, short-crested wind-generated waves in coastal regions and inland waters. In addition to wind data (as provided to TUFLOW FV), SWAN also requires swell to be provided on the boundaries. This was sourced from WAVEWATCH III, which is a global wave prediction model developed by the National Oceanic and Atmospheric Administration (NOAA). The SWAN model was run, using default parameters, on a regular grid with 500 m resolution.

Fish waste model

A fish waste model was developed based on the collective works of Tanner et al. (2007), Fernandes and Tanner (2008) and Tanner and Fernades (2010). The model assumes an average fish size of 1.5 kg and an average water temperature of 20°C, representing Abrolhos winter temperatures. Respiration and FCR/SGR values are based on Tanner et al. (2007), respectively. For the purposes of modelling, the SGR and FCR values reported in these papers were averaged to produce values of 0.29% and 3.1, respectively (Table 4.17).

Table 4.17 Specific Growth Rate and Food Conversion Ratio values

Value	SGR	FCR
1	0.25%	3.0
2	0.32%	3.2
Mean	0.29%	3.1

Source: Tanner et al (2007)

The model predicted the volume of waste for a given volume of fish, including the proportional nitrogen, phosphorus and carbon (the solid and dissolved fractions). Outputs from the fish waste model were fed into the particle transport model to predict the fate of the organic particles once discharged from the sea-cages.

Particle transport model

The Particle Transport Model (PTM) was used to resolve both the vertical and horizontal transport of aquaculture wastes, while accounting for differing size fractions and settling velocities of waste particles (i.e. waste feed and faecal material). The PTM was based on a Lagrangian particle tracking scheme driven by three-dimensional currents and wave fields described above. The Lagrangian particle movements included a deterministic component derived from the modelled currents and a stochastic 'random walk' component to represent vertical and horizontal dispersive processes due to unresolved turbulence scales. The processes of deposition and resuspension from the seabed due to wave and current induced shear stresses were also resolved using standard boundary layer and sediment transport calculations. A very large number of Lagrangian particles (~1 million) were released over a 12 month simulation period in order to integrate over a broad ensemble of environmental conditions, including stochastic dispersion processes.

The PTM calculated the transport of particles away from the cages, and quantified the rate of waste deposition near and far from the sea-cages. The Lagrangian PTM approach allowed for high resolution 'meshless' representation of the particle advection, dispersion, deposition and resuspension dynamics. The particle size, settling rates, ratio of nitrogen, phosphorus and carbon in the waste material was held at a constant, based on the outputs from the fish waste model described above. Particles that had settled out of suspension were tracked on the seabed and remained available for resuspension when wave and current induced shear stresses exceeded prescribed thresholds. No particle breakdown or burial processes were considered in the PTM simulations.

The science of particle transport through the water column is complex, with the bulk of studies focussing on inorganic particles and phytoplankton, with few that address the specifics of fish faeces (but see Chen et al. 1999, Felsing et al. 2005, Moccia et al. 2007, Moran et al. 2009). The settling velocity of fish waste leaving a sea-cage varies depending on an exhausting array of variables: feed type, fish health, species, fish size, and general farming practices (Chen et al. 1999, Felsing et al. 2005, Moccia et al. 2007, Moran et al. 2009). In addition, the difference between the volume of waste leaving a cage and the volume reaching the seafloor is also complex, and depends on biological and physical factors (e.g. current speeds and the extent of secondary consumption by scavengers beneath the cages; Felsing et al. 2005). For this study, fish waste was partitioned into waste feed (commercial aquaculture pellets) and waste faecal material. Faecal material was further partitioned into three size fractions following Chen et al. (1999), Cromey et al. (2002) and DHI (2013; Table 4.18).

Table 4.18 Waste particle fractions and settling velocities

Waste fractions	% of total input	Settling velocity (cm/s)	Source / assumptions
Feed (pellets)	1%	12.1	Tanner et al. (2007)
Faecal fraction 1	43%	1.5	DHI (2013)
Faecal fraction 2	32%	3.5	DHI (2013)
Faecal fraction 3	25%	5.5-6.3	Cromey et al. (2002), Chen et al. 1999.

Deposition of waste in this study was based on the Farmér concept (Tanner et al. 2007), where the largest proportion of particles falls beneath or close to the cages, with increasingly smaller proportions falling further from the cages. Modifications were made to include a total of five release points across the 38 m diameter sea-cages, and to account for the prevailing currents, which tended to skew the distribution of the finer particles in one direction over another. This concept is illustrated in Figure 4-11, which shows the rate of particle deposition over one year of production, but at differing stocking densities. Higher volumes are depicted directly under the cages (red to orange shading), with decreasing volumes depicted further from the cages (yellow to blue shading).

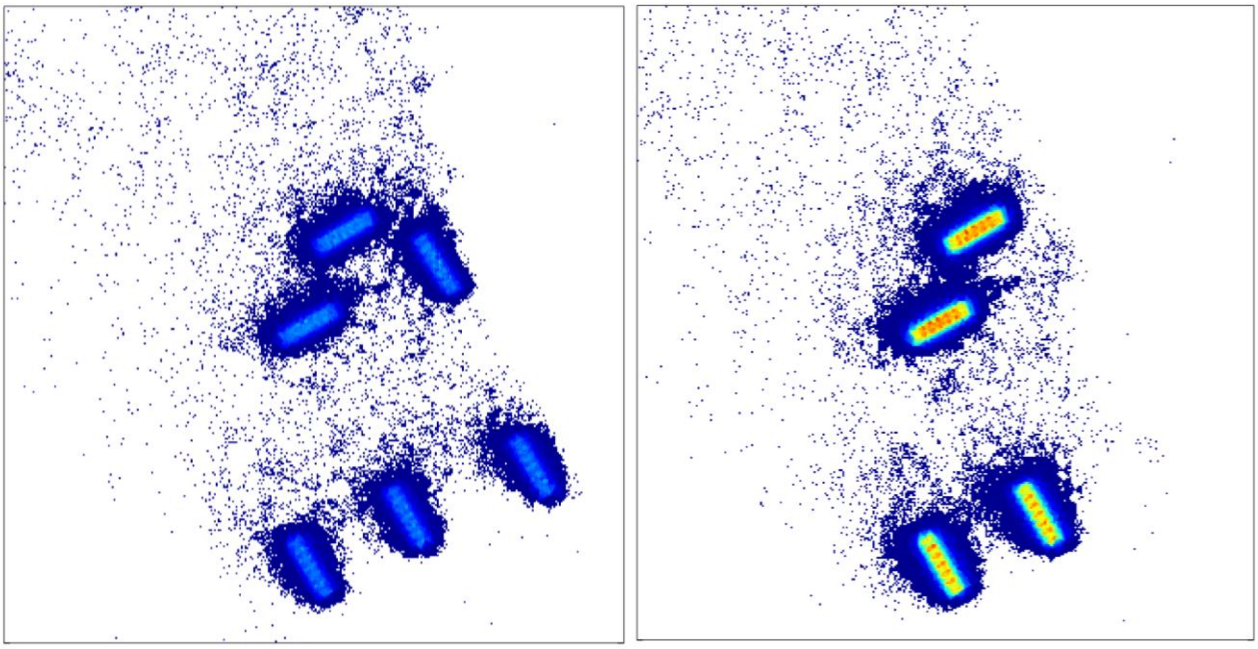


Figure 4-11 Deposition of waste material following twelve months of aquaculture production under differing stocking densities

Water quality model

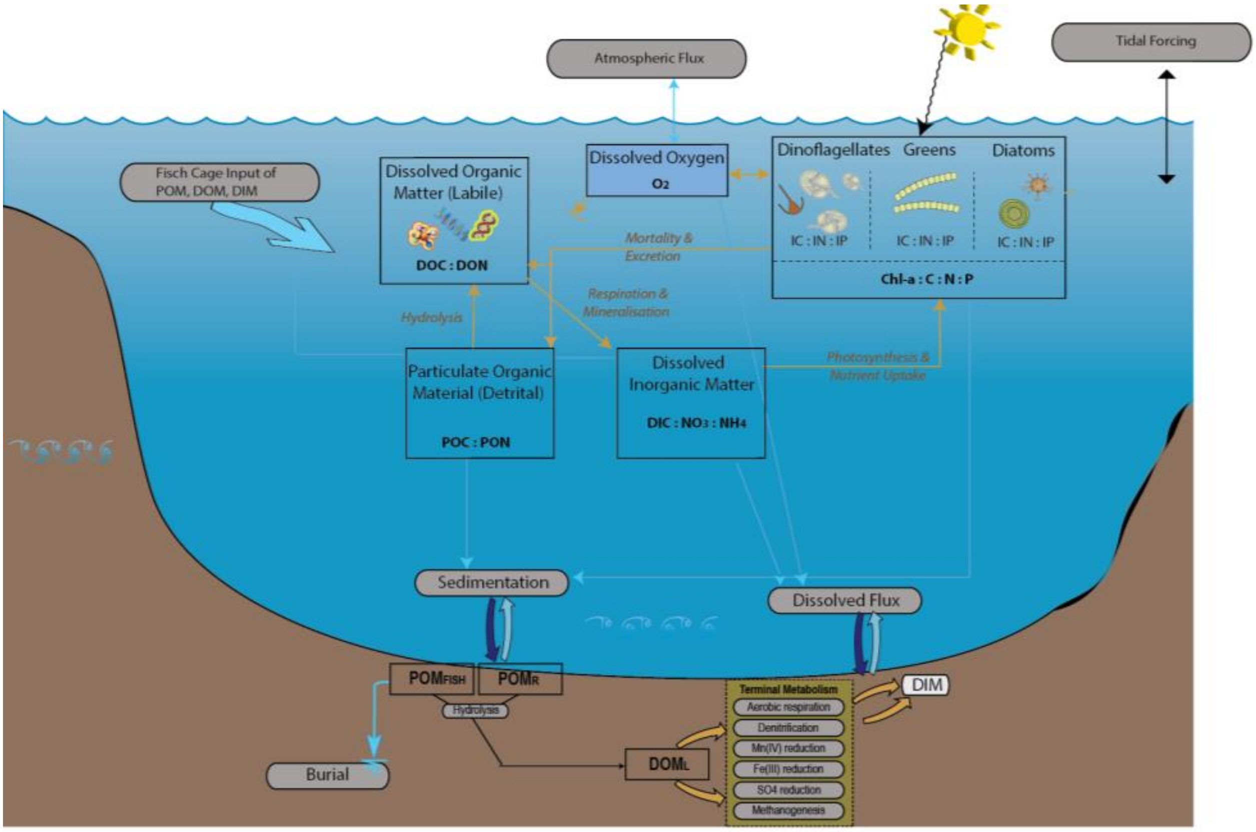
The water quality model utilised the Aquatic Ecodynamics (AED2) model library developed at UWA (http://aed.see.uwa.edu.au/research/models/AED/). It is capable of simulating a number of biogeochemical pathways relevant to water quality, including nutrient, sediment and algal dynamics. In this study it was configured to include organic matter, inorganic nutrients and phytoplankton (Figure 4-12).

The specific suite of parameters AED used in this study were:

- dissolved oxygen
- nutrients (nitrogen, phosphorus and associated species and cycles)
- organic matter (carbon, nitrogen and phosphorus, both particulate and dissolved)
- algae (one generic species in this study).

Boundary conditions for AED were derived from observations collected as part of the sampling program (Section 5.3) and parameters were chosen to represent a typical oligotrophic region.

Working with the hydrodynamic model, the water quality model was used to resolve the release, dispersion and dilution of inorganic nutrients from the sea-cages, and subsequent uptake and growth of phytoplankton. The model was also used to resolve the potential for changes in dissolved oxygen and light attenuation at the bottom of the water column.



Notes:

1. POM (particulate organic matter); DOM (dissolved organic matter); DIM (dissolved inorganic matter); DOC (dissolved organic carbon); DON (dissolved organic nitrogen); IC:IN:IP (inorganic carbon:inorganic nitrogen:inorganic phosphorus); C:N:P (carbon:nitrogen:phosphorus); NO_3:NH_4 (nitrate:ammonia)

Figure 4-12 Carbon and nutrient processes simulated in CANDI-AED

Sediment diagenesis model

Biogeochemistry

The diagenesis[1] model was first used to resolve the biogeochemistry of the seafloor and to estimate the nutrient flux into and out of the sediments under a range of waste deposition scenarios (Appendix G). It was then coupled to the hydrodynamic and water quality models to ensure the phytoplankton response was based on the cumulative sources of nutrients, both directly from fish respiration and indirectly via sediment mineralisation processes. Importantly, the diagenesis model was also used to determine the recovery of sediments beneath the sea-cages, and then from this, to map the spatial distribution of the zones of aquaculture influence (ZoHI, ZoMI and the ZoI).

The diagenesis model adopted in this EIA was the CANDI-AED model, which is an extension of the numerical code written by Boudreau (1996), and widely used across a range of marine and coastal environments (Paraska et al. 2014). The configuration of the model was guided by a previously published sediment biogeochemical model application to finfish aquaculture (Brigolin et al. 2009). Additional sources used for guidance in the development of diagenesis model setup and parameters are given in Table 4.19. For an overview of the theory and applications of sediment diagenesis models refer to the review by Paraska et al. (2014).

Table 4.19 Sources of literature informing the development of the diagenesis model

Reference	Study location
Macleod & Forbes 2004	Salmon farms in Tasmania
Tanner & Fernandes 2007	Yellowtail kingfish farms in Fitzgerald Bay in Spencer Gulf, South Australia
Fernandes &Tanner 2008	
Brigolin et al. 2009	Salmon farms in Loch Creran, Scotland
Volkman et al. 2009	Salmon farms in the Huon Estuary and D'Entrecasteaux Channel, Tasmania

Based on field observations, it was assumed that a generalisation for the sediment physical properties was a highly porous and permeable sediment of approximately 15 cm depth, with hard rock beneath. In previous diagenesis modelling studies, a shallow depth of sediment with hard rock underneath has not been specifically simulated (Paraska et al. 2014). Therefore much of the model was derived from Van Cappellen and Wang (1996), which is a well-established study and based in a marine study site. In order to simulate the vertical mixing of the sediment, a relatively high bioturbation rate of 20 cm^2/y was used, with a constant value from the sediment-water interface to the deepest layer at 15 cm.

Chemical concentrations at the sediment-water interface are subject to a mix of competing forces at different spatial and temporal scales, for example: solid particles are deposited via gravity and resuspended by currents in the water column; particles are buried following further deposition and ultimately form rock; chemicals diffuse between the water and the sediment, and within the sediment, following concentration gradients; benthic animals and plants cause mixing or binding of the sediment particles, as well as non-local transport of chemicals; bacteria use chemical reactions to fuel their metabolism; benthic animals, plants and bacteria thrive or die depending on the chemicals present in the sediment (Berner 1980, Boudreau 1997, Fossing et al. 2004). The chemical reactions simulated in the model can be broadly defined as primary and secondary reactions; these are summarised in Figure 4-13. Primary reactions are the microbially-driven breakdown reactions of organic matter via a series of oxygen reduction (redox) pathways (Figure 4-14). Primary reactions are the driving force of most of the other chemical reactions that occur in the sediment. Inputs of fish feed and faecal matter serve to quickly shift chemical

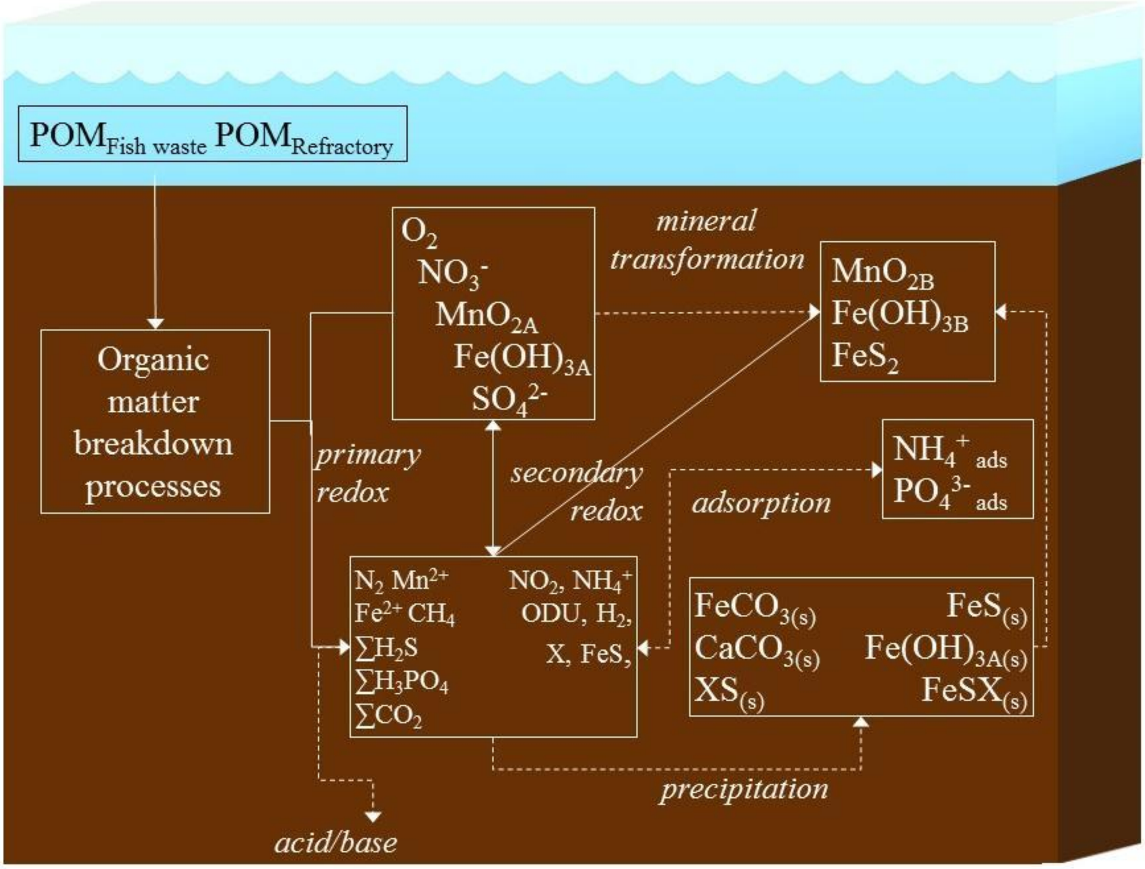
---

[1] *Diagenesis is the term used for all of the changes sediments undergo following inputs of organic material*

concentrations away from the equilibrium that occurs in marine waters, especially those which are naturally nutrient poor (i.e. waters of the Abrolhos Islands).

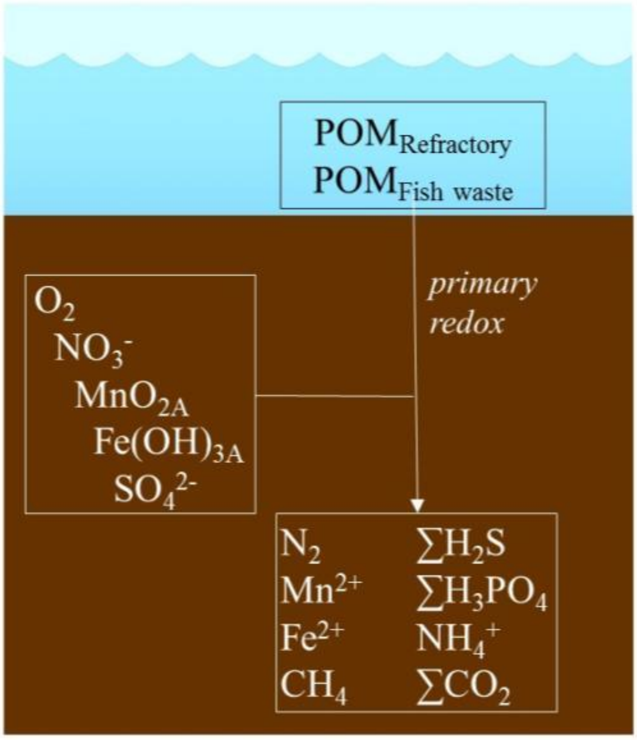
One guiding principle used to understand how the competing pathways of primary organic matter reactions interact is the sediment redox sequence. There is an assumption that there are six major terminal electron accepting pathways for the degradation of organic matter, and that bacteria will use these pathways in order of decreasing free energy yield: aerobic, then denitrifying, manganese reducing, iron reducing, sulfate reducing and finally methanogenic respiration. Since the source of fresh sediment organic matter is always the top of the sediment, each terminal electron accepting pathway corresponds with a depth zone (Van Cappellen et al. 1995).

The diagenesis model was applied to MWADZ sediment, firstly under background conditions, then with 2, 3 and 5 years of organic matter deposition from fish-waste, then 7+ years with no deposition (post fallowing) to simulate a recovery period. The simulation was calibrated against available field data, primarily total organic carbon (TOC), total nitrogen (TN) and total phosphorus (TP) (Section 5.3.3). The resulting chemical concentration profiles were then assessed against a spectrum of organic matter deposition fluxes, from 1×10^2 to 5×10^6 mmol/m^2/y to explore how the sediment would respond to a range of stocking densities, near and far from the cages. The resulting recovery time in sediment concentrations, and absolute concentrations of key sediment variables were then assessed, and used to define the zones of high and moderate impacts, and the zones of influence, as per EAG 7.



Note:
1. POM = particulate organic material, ads = adsorbed.

Figure 4-13 Processes simulated in the CANDI-AED sediment diagenesis model



Note:

1. POM = particulate organic material.

Figure 4-14 Organic matter degradation processes simulated in the diagenesis model

Metal accumulation and recovery

In addition to its capacity to simulate the biogeochemistry of the sediments, the diagenesis model simulated the chemical processes leading to the accumulation and compound-forming transition of metals (Zn, Cd and Cu). The purpose of the modelling was to determine the potential for metal accumulation in the sediments beneath sea-cages and the time required for recovery after fallowing. The chemistry is such that the concentrations of metals correlate strongly with the presence of sulphides. A simple approach was simulated in which accumulation occurred under conditions of low oxygen and high sulphide concentrations, and flux (out of sediments) occurred as oxygen and sulphides returned to baseline conditions.

The potential for impacts relating to the metal content of commercial feeds was assessed based on metal concentrations in fish faeces and its potential to accumulate in the sediment. The metal content of the fish faeces was based on the analysis by Moccia et al. (2007; Table 4.20) and then converted to a molar ratio compared with carbon (Table 4.21). Modelling undertaken for this study focussed on the metals in greatest supply (Zn and Cu) and on the metals for which there are EPA triggers (EPA 2014). Concentrations are for total metals in mg/kg. The thresholds used to determine the spatial extent of contamination, and thus the zones of impact are outlined in Table 4.10, Section 4.5.1.

Table 4.20 Elements measured in fish faeces fed on commercial aquaculture feeds

Element	Average (mg/kg)	Standard deviation
As	<1.0	0.0
Cd	<1.0	0.0
Co	<1.5	0.0
Cr	5.01	2.09
Cu	42.22	30.53
Fe	1003.56	296.30
Hg	<0.05	0.0
Mn	695.94	279.79
Mo	<2.5	0.0
Ni	<4.0	0.0
Pb	<5.0	0.0
Se	<1.0	0.0
Zn	620.56	238.47

Source: Moccia et al. 2007

Table 4.21 Fish waste organic matter converted from values in Moccia et al. (2007) to a molar C:metal ratio

	Mass per mass	Molar ratio	Exceedance concentration
Zn	620 mg Zn/kg faeces	2.79×10^{-4} mol Zn/mol C	7.7 mmol Zn/L
Cu	42 mg Cu/kg faeces	1.89×10^{-5} mol Cu/mol C	2.5 mmol Cu/L
C	0.41 kg C/kg faeces	–	–

Source: Moccia et al. 2007

The chemical reactions that metals are subject to are summarised in Table 4.22. Over reactions (1) to (6), organic metals are released from the organic matter upon microbial oxidation and then diffused as a free solute, or precipitated out as a metal sulphide; then metal sulphides can be oxidised by oxygen to release the free metal again. The criteria for metal contamination were 200 and 65 mg/kg dry weight for Zn and Cu respectively, or 7.7 and 2.5 mmol metal/L (Table 4.21).

Table 4.22 Major reaction equations for metal release

$$OM.Zn + oxidant \rightarrow CO_2 + NH_4^+ + PO_4^{3-} + Zn^{2+} \quad (1)$$
$$Zn^{2+} + S^{2-} \rightarrow ZnS \quad (2)$$
$$ZnS + 2O_2 \rightarrow SO_4^{2-} + Zn^{2+} \quad (3)$$
$$OM.Cu + oxidant \rightarrow CO_2 + NH_4^+ + PO_4^{3-} + Cu^{2+} \quad (4)$$
$$Cu^{2+} + S^{2-} \rightarrow CuS \quad (5)$$
$$CuS + 2O_2 \rightarrow SO_4^{2-} + Cu^{2+} \quad (6)$$

4.6.2 Model assumptions

The modelling approach adopted here was to build an integrated hydrodynamic, water quality, particle transport and sediment diagenesis model, which captured the key environmental processes and their interactions. A conservative approach was adopted to ensure the outputs of modelling were equivalent to 'most likely worst case' outcomes, as required by the ESD (EPA 2013) (Table 2.1). As such, the impacts predicted in this document are more extensive than might be expected on average, but are nevertheless within the upper range of impacts reported in the literature (i.e. Brooks et al. 2004). The assumptions underpinning the development and execution of the integrated model are summarised below:

- The hydrodynamic and the wave models were calibrated and validated against metocean data collected over a 10 month period, encompassing each of the calendar seasons. Climatic conditions during the data collection phase were considered normal, and captured the normal seasonal pattern of changing winds, waves and oceanographic currents. Although the metocean data collection period captured the normal pattern of winter storms, no significant storm events were captured. For example, since 1915, a cyclone has passed through coastal waters within 400 km of the region approximately every 2.5 years on average (Bureau of Meteorology).
- The predicted zones of impact shown in Section 7 are based on rates of waste deposition and resuspension averaged over the period of operation (examples for 5 years of operation are given in Section 7.3.2). If viewed as an animation, rather than a static image, the actual area occupied is subject to short-term changes depending on the levels of shear stress operating at the time.
- Rates of recovery (Section 7.3.2) as predicted by the sediment diagenesis model were assumed to proceed free of major disturbances. A constant rate of bioturbation of 20 m^2/y was simulated across all strata of the sediment to a depth of 15 cm, thus simulating some capacity for reoxygenation. However, despite capturing some capacity for biodiffusion and irrigation, neither of these account for the potential ‘resetting’ of the sediment during major scour events i.e. such as those which may occur during storm events. As such there is a strong conservative factor in the results for longer time frames.
- The Food Conversion Ratio (FCR) and Specific Growth Rate (SGR) values used in the development of the fish waste model (Section 4.6.1) are based on the collective works of Tanner et al. (2007), Fernandes and Tanner (2008) and Tanner and Fernandes (2010). These studies are the only peer reviewed source of information on the respiration, metabolism, energetics and the nutrient and carbon outputs of yellow tail kingfish, and were used here as the basis of the model. The outputs produced by the model are conservative, and likely greater than the outputs that will be achieved once the farms are established. Aquaculture proponents have a vested interest to achieve food conversion ratios better than 3.1, with ratios in the range 1.5–2.0 being standard across the industry.
- Modelled estimates of the total volume of fish waste expected to reach the seafloor are based on the physical and hydrodynamic properties of several different waste fractions: pelletised feed, and three faecal size fractions. The two largest fractions were assumed to settle rapidly (Table 4.23), and the smallest, slowly. Smaller particles tended to settle further from sea-cage infrastructure, and larger particles settled closer. The dispersion of fine particles was enhanced under higher current speeds, and retarded under lower current speeds.
- It was also assumed that fish wastes (faecal material) exhibited cohesive (‘sticky’) properties, increasing its propensity for ‘clumping’ and limiting its potential for resuspension relative to inorganic particles (following Nowell et al. 1981; Masalo et al. 2008). The carbon in the material was also assumed to be highly labile, meaning much of it was consumed and oxidised relatively quickly by resident microbiological flora (following deBruyn & Gobas 2004). Hence, much of the material deposited from cages was assimilated quickly resulting in rapid changes to sediment chemistry.
- Notwithstanding the generally assumed cohesive and ‘sticky’ properties of the waste, the smallest size fraction simulated demonstrated high capacity for dispersion. It was conservatively assumed that these fine particles, which might ordinarily be expected to dissolve over the periods simulated (12 months), remained in suspension indefinitely. This resulted in outputs showing widespread and highly distant dispersion of particles, albeit not in densities/volumes expected to result in impacts to sediment biology.

Table 4.23 Time for modelled particles to reach the seafloor

Distance to sea floor from bottom of cage	Settling time	
	Medium particles	Large particles
5m	2.3 min	1.3 min
30m	14.2 min	7.9 min

4.6.3 Peer review

The approaches to developing the integrated hydrodynamic, particle transport, water quality and sediment digenesis models were subjected to independent peer review. All aspects of the approach, including the collection of baseline metocean data, the development of thresholds and the assumptions underpinning the development of the models were assessed. The peer review process and response is detailed in Appendix E.

5. Baseline Conditions

5.1 Hydrodynamics and wave climate

Currents around the Abrolhos Islands are dominated by the Leeuwin Current system, primarily consisting of the Leeuwin Current (a poleward-flowing, boundary current which is usually stronger in winter and weaker in summer) and the returning Capes Current (a northward-flowing current on the continental shelf, which is strongest in summer; see review by Pattiaratchi & Woo, 2009).

Current speeds and wave heights were measured in the northern and southern areas of the MWADZ and at two regional sites to the east of the MWADZ (Figure 4-1). As illustrated in Figure 5-1, the ADCPs deployed at the regional sites between November 2014 and March 2015 captured the Capes Current, which had typical flows of approximately 0.1-0.2 m/s northwards. The hydrodynamic model captured the Capes Current in summer, with similar velocities (Figure 5-1), and also captured the Leeuwin Current adjacent to the continental slope, with southward velocities ranging between ~ 0.1-0.3 m/s (Figure 5-2).

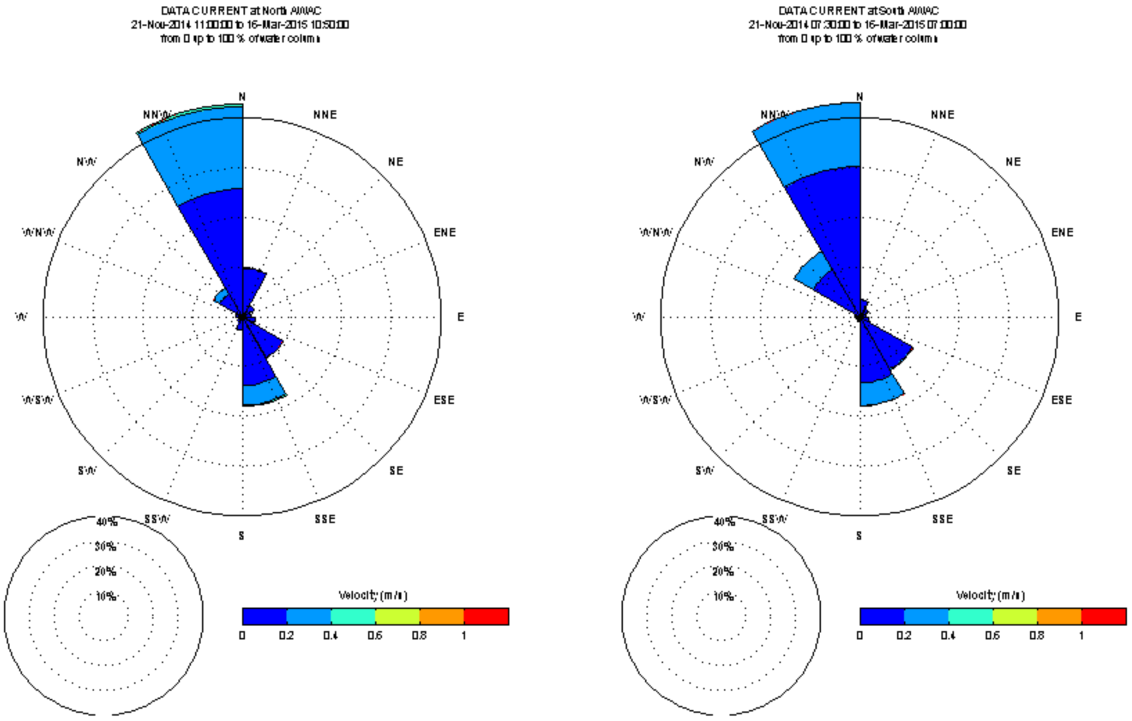


Figure 5-1 **Current directions and speeds at regional sites between November 2014 and March 2015**

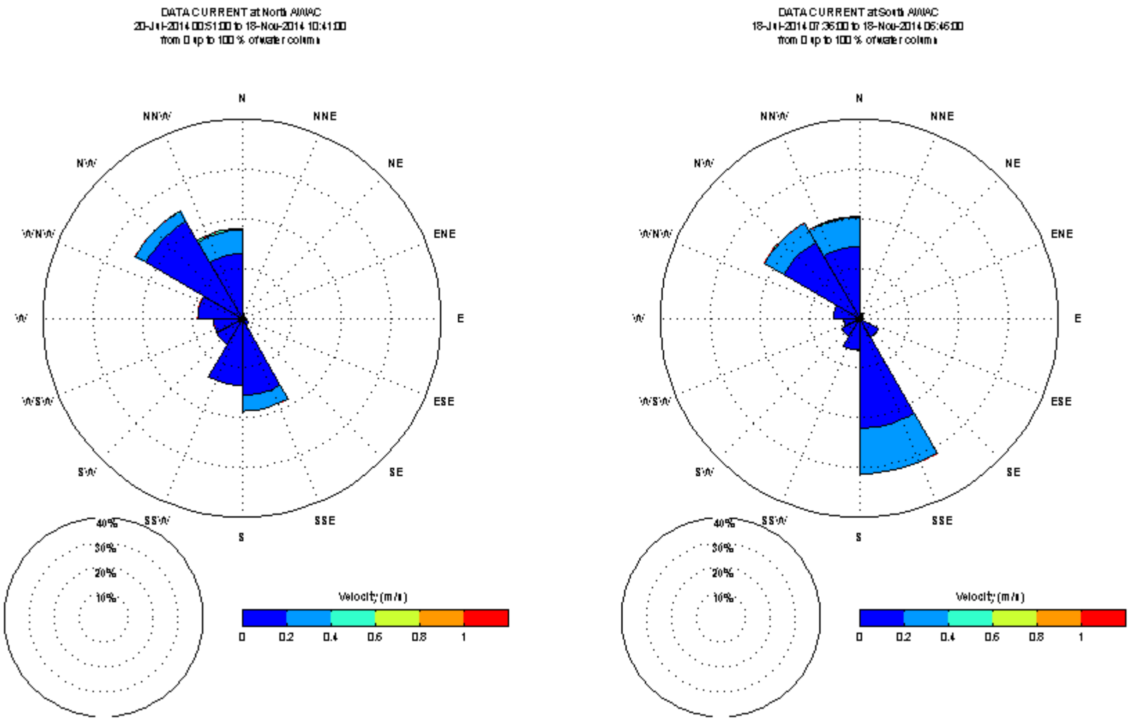


Figure 5-2 Current directions and speeds at regional sites between July 2014 and November 2014

Rose plots of depth-averaged velocity measured by the MWADZ ADCPS are presented in Figure 5-3–Figure 5-4. The currents in the southern area (L2) flowed primarily along the east-west axis, as north-south flow was hindered by the presence of the adjacent islands of the Pelsaert group. Measured flow was predominantly westward during the May-June deployment, switching to eastward during the November-December deployment, with no dominant current direction during the August-September or February-March deployments.

Currents in the northern area (L1) are typically had higher velocities than those in the south, but with no dominant direction of flow during the May-June (Figure 5-3) and August-September deployments. During the summer deployments, the direction of flow was typically to the northwest, with velocities of approximately 0.1-0.3 m/s (Figure 5-4). The hydrodynamic model simulated similar conditions (Appendix F).

The regional sites had somewhat similar wave climates, although with lower significant wave height at the northern site. Mean significant wave height was 1.6 m (northern site) and 2.2 m (southern site) during the July-November deployment, and 1.5 m (northern site) and 2.1m (southern site) during the November-March deployment. Mean wave periods were approximately 11-12 s during the July-November deployment and 8-10 s during the later deployment at both sites, while peak wave direction was from the SSW. At the northern lease site, significant wave heights were lower (means of approximately 1 m during each deployment bar Aug-Sep, which was 1.3 m), periods were similar (approximately 10s) and the peak wave direction was from the WSW.

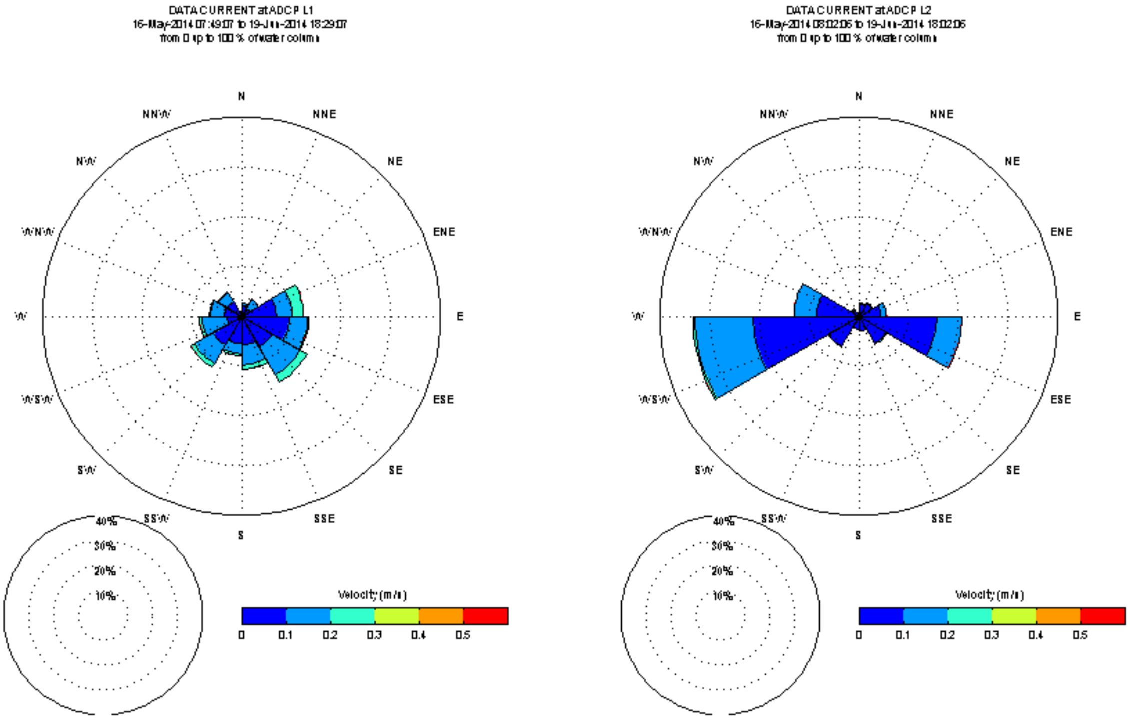


Figure 5-3 Current directions and speeds in the northern (L1) and southern (L2) areas of the MWADZ between May and June 2014

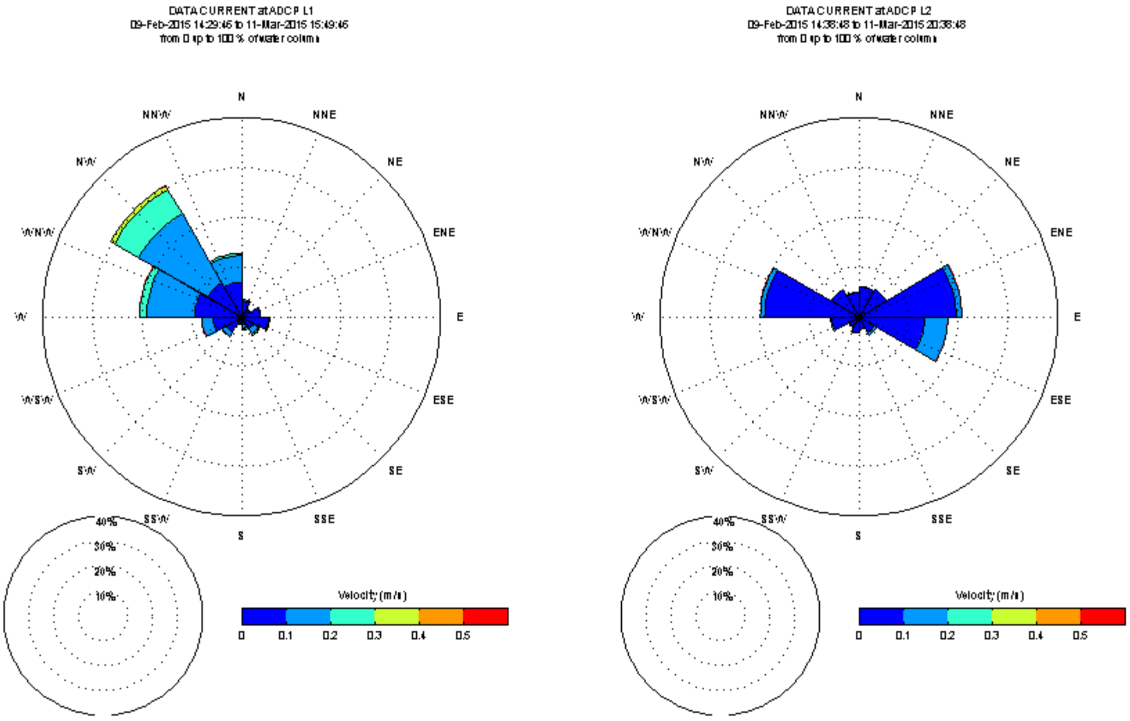


Figure 5-4 Current directions and speeds in the northern (L1) and southern (L2) areas of the MWADZ between February and March 2014

5.2 Biogeochemical processes

Natural biochemical processes were not empirically measured in the MWADZ sediments. Attempts to obtain consolidated sediment cores for this purpose failed due to the deep water (beyond diving depth) and the characteristics of the sediments–consisting of a shallow coarse layer of sand (of ~15 cm depth) overlying a rocky substrate. Given the depth, porosity and coarseness of the sediments, it was assumed that sediments were naturally well oxygenated, and free of sulphides probably throughout the sediment column (i.e. ~15 cm). For further context see Section 3.3.

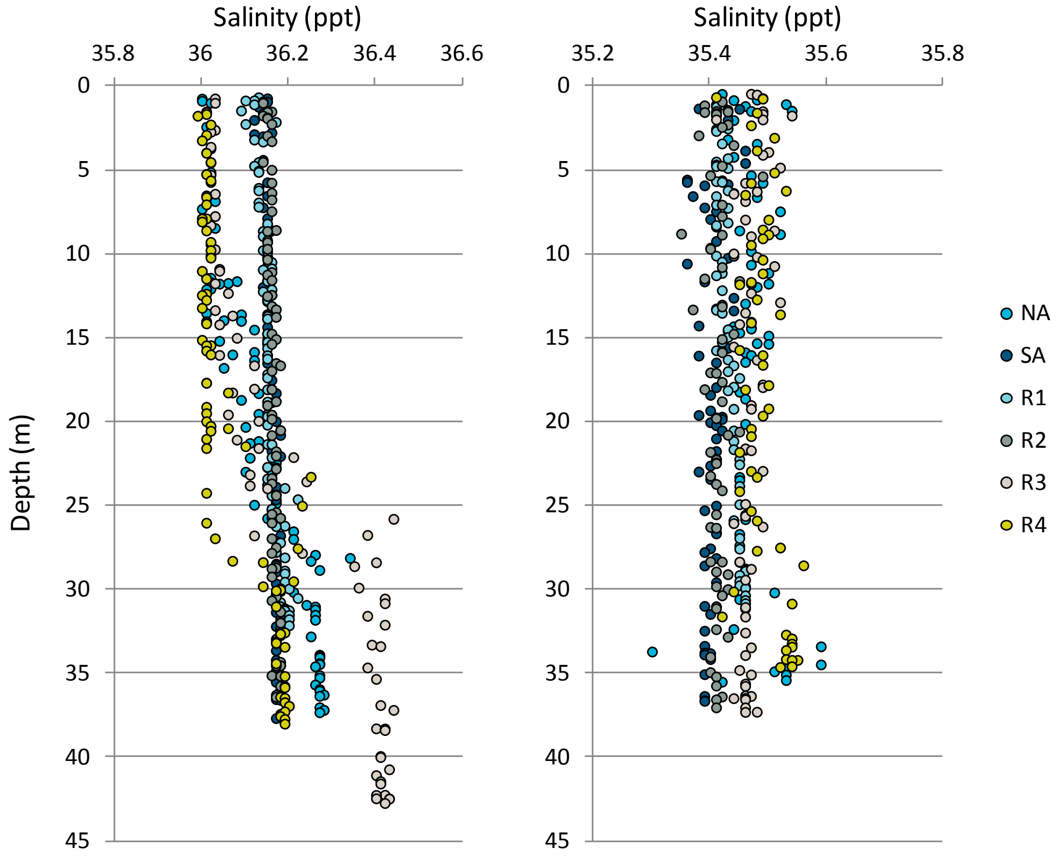
5.3 Water quality

5.3.1 Physical and chemical

Salinity readings confirmed that there was no significant stratification at any location across the seasons, indicating a well-mixed water column. However, salinity readings during autumn 2014 at the northern area (NA) and reference locations R3 and R4 increased from 36 ppt to 36.14–36.43 ppt at 29 m water depth (Figure 5-5). During winter 2014, the northern and southern (SA) MWADZ areas and reference locations had slightly lower salinities throughout the water column (~35.35–35.59 ppt) than autumn 2014 (~35.99–36.44 ppt) and summer 2015 (~35.74–36.16 ppt; Figure 5-5).

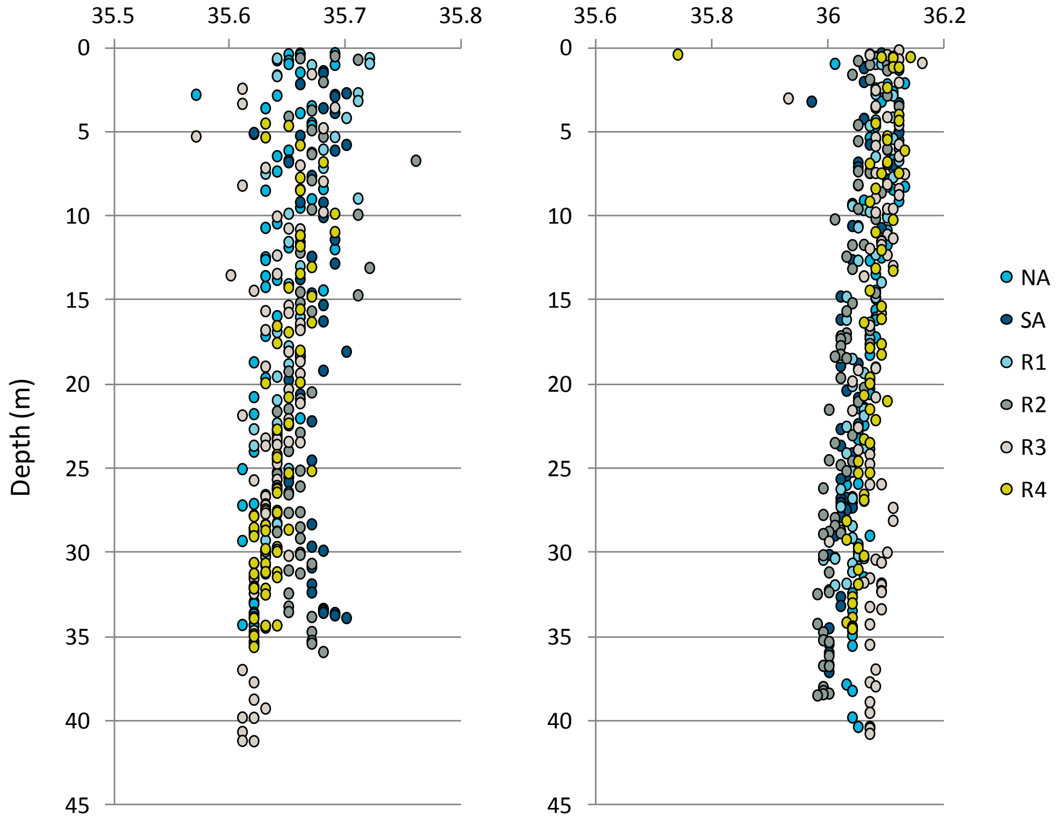
A temperature gradient was observed at the deeper reference location R3 (~43 m deep) particularly during autumn and summer, were temperatures dropped ~0.36–1.31°C between 15 m and 25 m (Figure 5-6). The three most northern locations (northern area [NA], R3 and R4) displayed similar decreasing trends in temperatures during autumn and winter (Figure 5-6), possibly a result of cooler water delivered to this area during periods of increased water movement. Across all locations, surface temperatures (0–10 m) were typically lower during spring (21.09–21.71°C) than summer (23.31–23.48°C; Figure 5-6).

DO concentrations showed no clear trend between the northern, southern and reference locations over the year (Figure 5-7). Across all sites and sampling periods, mean surface DO saturation was always >96%, while mean bottom DO saturation was always >95% (Table 5.1). Mean bottom DO saturation was slightly lower than mean surface DO saturation during the autumn and winter sampling periods. There was a slight decreasing trend in DO saturation with increasing depth across all locations over all four seasons (Figure 5-7). Across all locations and seasons, mean surface (0–10 m) DO saturation values were always >~94.6%, while mean bottom DO saturation values were >95% (Figure 5-7).



a) Autumn 2014

b) Winter 2014



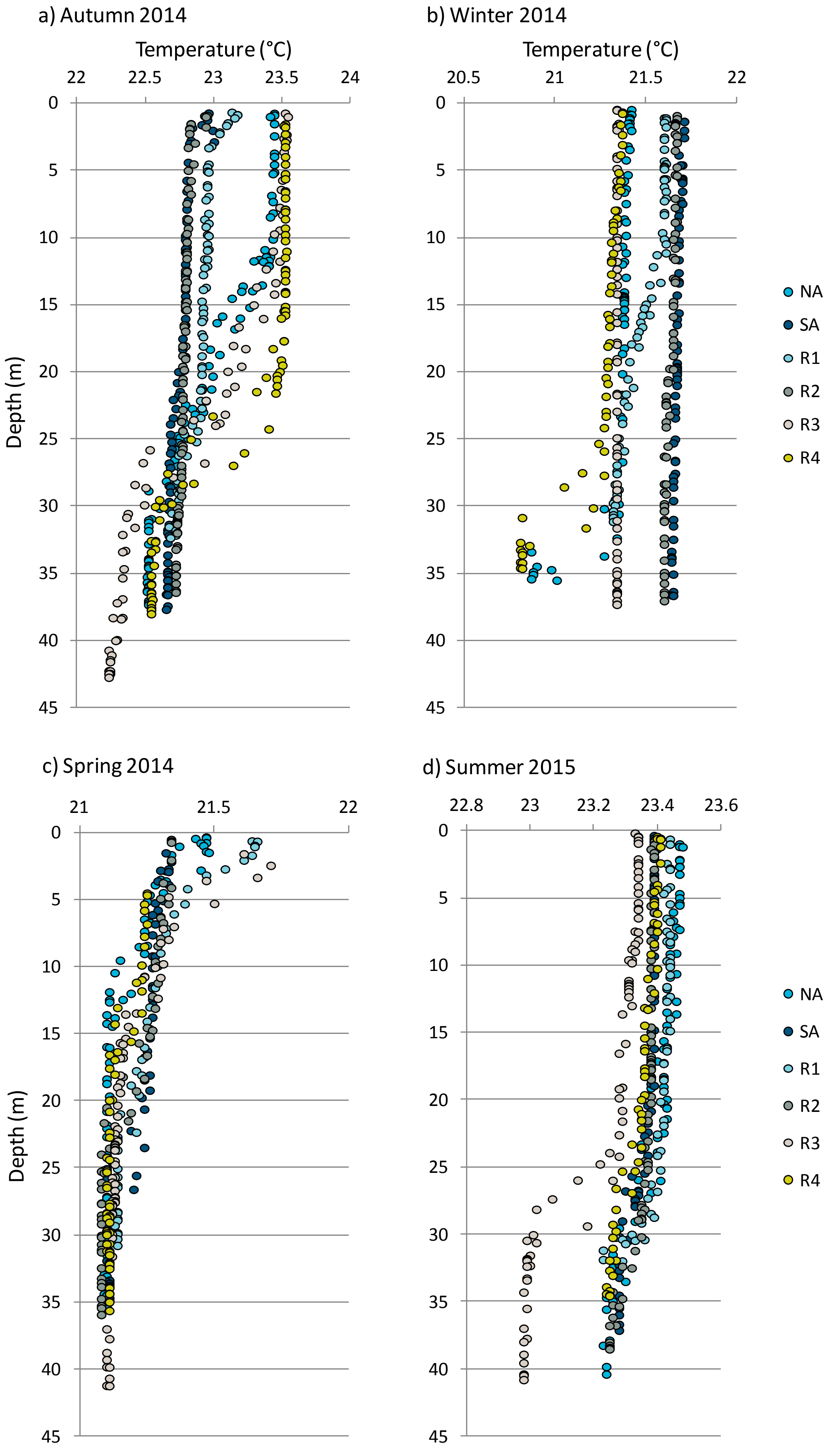
c) Spring 2014

d) Summer 2015

Note:

1. NA = northern area, SA = southern area, R1–R4 = reference areas.

Figure 5-5 Salinity measured in autumn, winter and spring 2014, and summer 2015 at all locations



Note:
1. NA = northern area, SA = southern area, R1–R4 = reference areas.

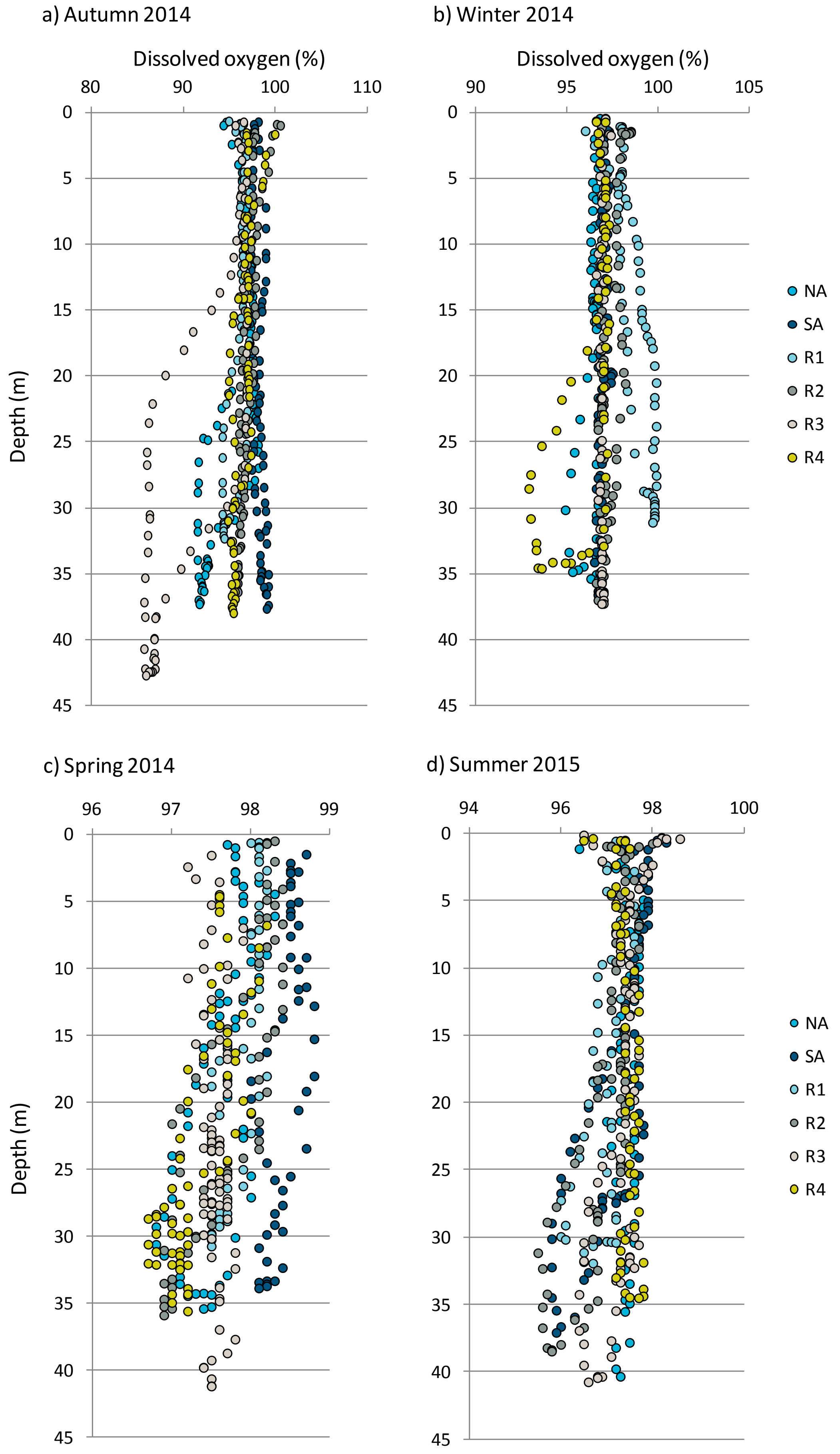
Figure 5-6 **Temperature measured in autumn, winter and spring 2014, and summer 2015 at all locations**

Table 5.1 Dissolved oxygen statistics at all locations

Season	Autumn			Winter			Spring			Summer		
MWADZ	**N**	**S**	**R**	**N**	**S**	**R**	**N**	**S**	**R**	**N**	**S**	**R**
Mean surface DO (%)	98	98	98	97	96	98	98	99	98	97	98	97
Standard deviation	2	1	2	1	1	2	1	1	1	0	1	1
Mean bottom DO (%)	96	97	95	95	96	96	98	98	97	97	97	97
Standard deviation	3	1	4	1	2	2	1	1	1	0	1	1

Notes:

1. MWADZ = Mid-west aquaculture development zone; N = northern MWADZ, S = southern MWADZ, R = reference
2. DO = dissolved oxygen



Note:

1. NA = northern area, SA = southern area, R1–R4 = reference areas.

Figure 5-7 **Dissolved oxygen measured in autumn, winter and spring 2014, and summer 2015 at all locations**

5.3.2 Light attenuation and irradiance

During August–September, K_d showed similar variation across the northern and southern areas – in the northern area, K_d ranged 0.04–0.17 per m while in the southern area, K_d ranged 0.06–0.19 per m (Figure 5-8). K_d measured over November-December showed similar variation across areas – in the northern area, K_d ranged 0.04–0.12 per m while in the southern area, K_d ranged 0.04–0.15 per m (Figure 5-9).

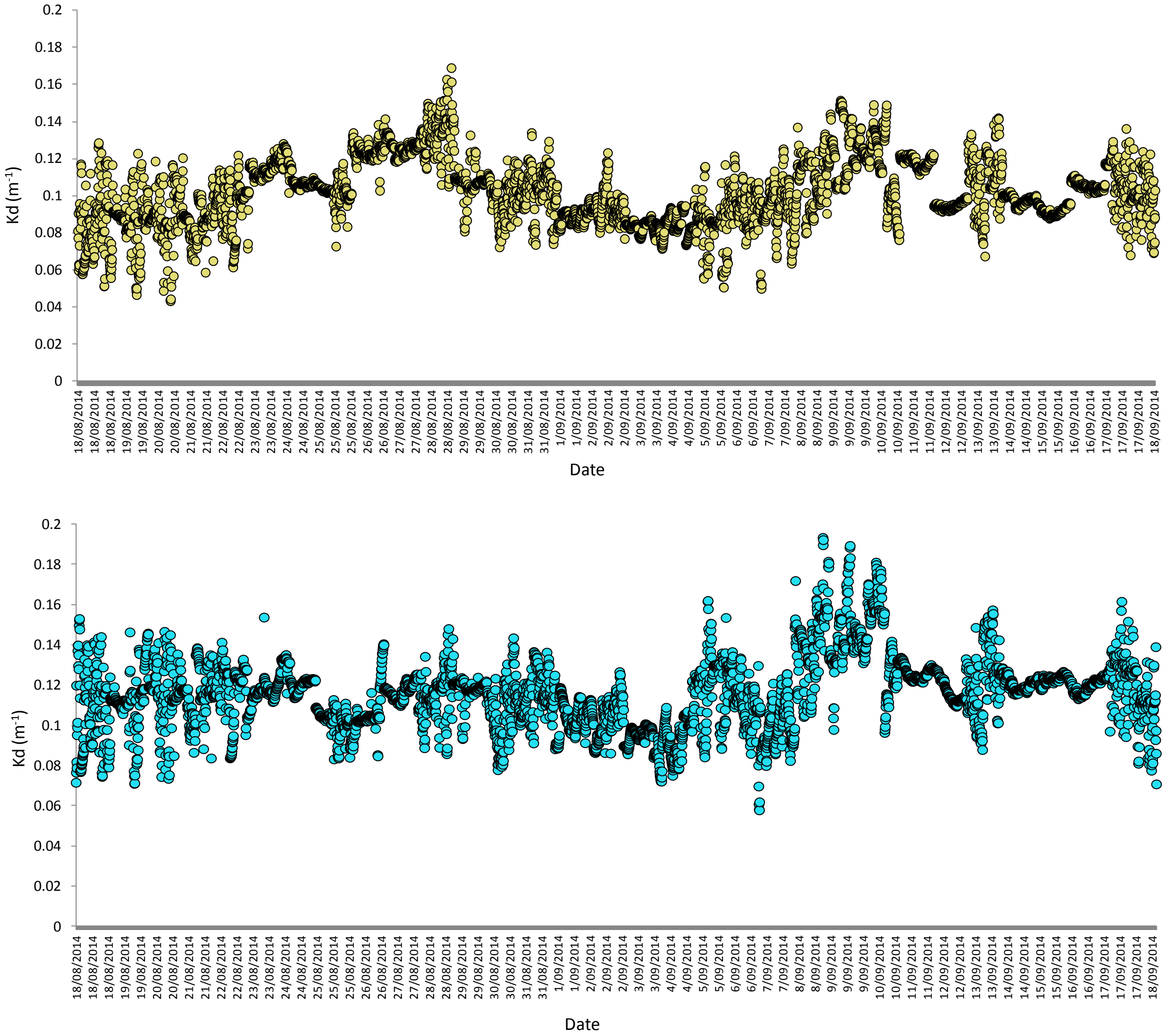


Figure 5-8 **Comparative light attenuation data between the northern (upper panel) and southern areas (lower panel) (August–September 2014)**

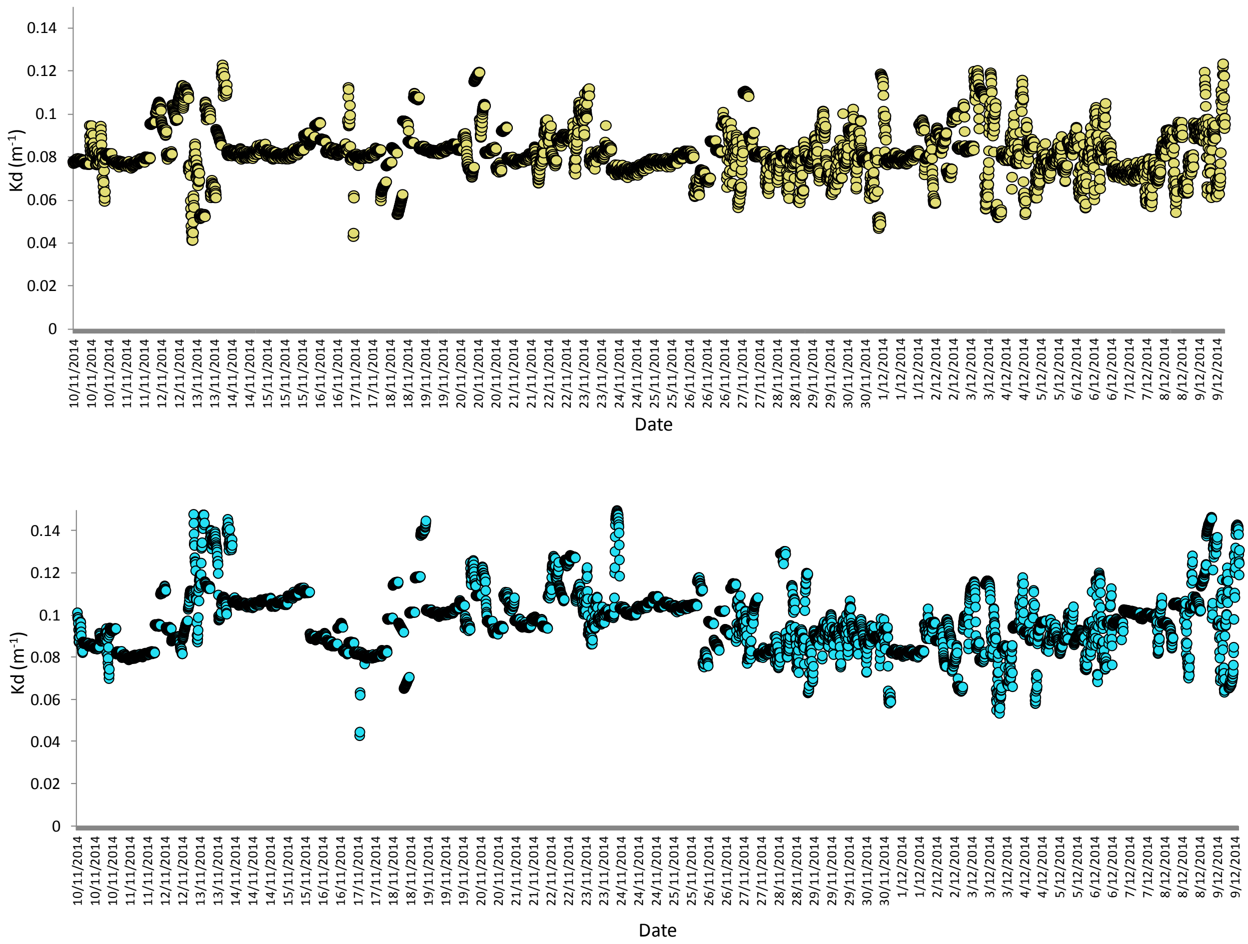


Figure 5-9 **Comparative light attenuation data between the northern (upper panel) and southern areas (lower panel) (November–December 2014)**

Light intensity for the 1st, 5th, 20th and 50th percentiles was calculated for each of the four sampling periods in the northern area and the southern area of the MWADZ (Table 5.2). Mean light intensity across the percentiles ranged 11.3–52.2 mol.photons/m^2/s in the northern are, while mean light intensity was lower in the southern area ranged 6.2–33.7 photons/m^2/s across percentiles. At both the northern and southern area, spring had the highest light intensity in each percentile, while autumn had the lowest light intensity in each percentile.

Table 5.2 **Light intensity statistics from the northern and southern areas**

Percentile	Autumn	Winter	Spring	Summer	Mean
Northern area					
1st	0.9	5.1	22.0	17.2	11.3
5th	1.5	8.4	31.3	21.0	15.5
20th	4.3	15.2	59.9	36.5	29.0
50th	9.0	27.6	108.3	64.1	52.2
Southern area					
1st	1.1	3.0	11.9	8.9	6.2
5th	2.6	5.1	17.5	12.0	9.3
20th	4.4	15.0	42.7	23.1	21.3
50th	6.3	22.5	62.7	43.3	33.7

Notes:

1. Northern MWADZ light intensity was measured at Rat Island
2. Autumn = May/June 2014, winter = August/September 2014, spring = November/December 2014 , summer = February/March 2015
3. Units are mol.photons/m^2/s.

5.3.3 Nutrients

Total nitrogen

Total nitrogen (TN) in both surface and bottom waters fluctuated in concentration across time (Table 5.3). June and November 2014 reported higher TN concentrations at the surface (0.151 ± 0.008 mg/L and 0.137 ± 0.004 mg/L, respectively) and bottom (0.16 ± 0.01 mg/L and 0.15 ± 0.01 mg/L, respectively) of the water column (Table 5.3). A significant Time x ZvR interaction in surface waters was detected, as the combined northern and southern areas (Zone) recorded higher TN concentrations than the reference locations, with the exception of May 2014 (Zone = 0.06 ± 0.01 mg/L, Reference = 0.09 ± 0.01 mg/L) and December 2014 (Zone = 0.07 ± 0.01 mg/L, reference = 0.084 ± 0.004 mg/L).

Table 5.3 Results of a three-factor PERMANOVA examining total nitrogen concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	2.49E-02	**0.0001*** **	2.14E-02	**0.0001*** **
ZvR	1	3.91E-03	0.2713	1.95E-05	0.9039
Location(ZvR)	4	6.37E-03	0.0913	8.81E-04	0.7361
TimexZvR	7	8.39E-03	**0.0044** **	1.32E-03	0.5612
TimexLocation(ZvR)	28	4.42E-03	0.1800	2.74E-03	0.0602
Res	168	2.99E-03		1.61E-03	
Total	215				

Notes:

1. Significant results shown in bold; **= highly significant (p<0.01), *** = very highly significant (p<0.001).

1. ZvR = Zone vs Reference

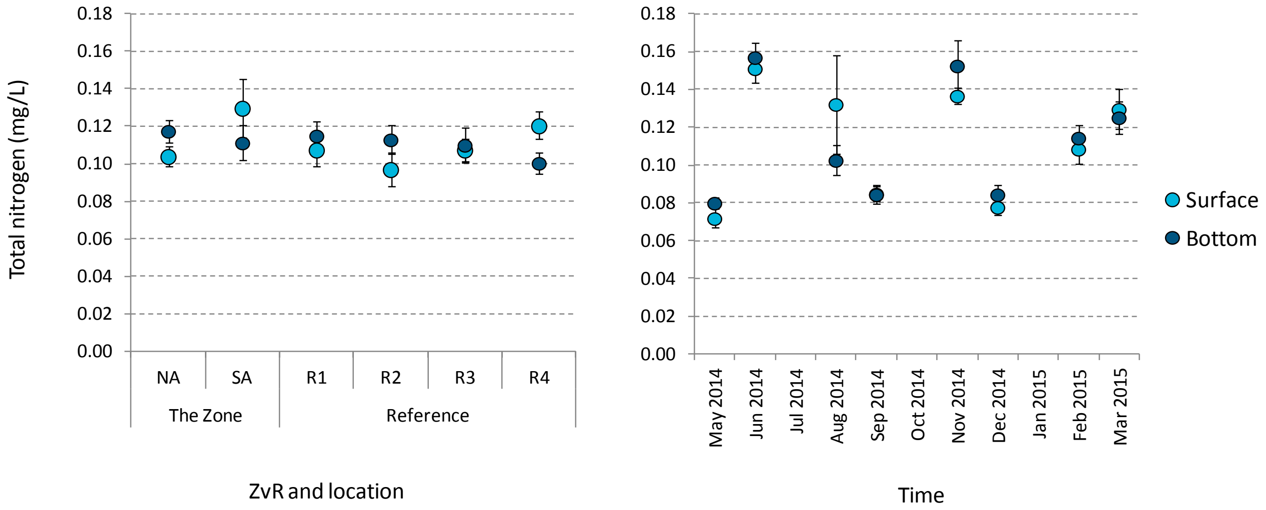


Figure 5-10 Total nitrogen (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR and time

Total phosphorus

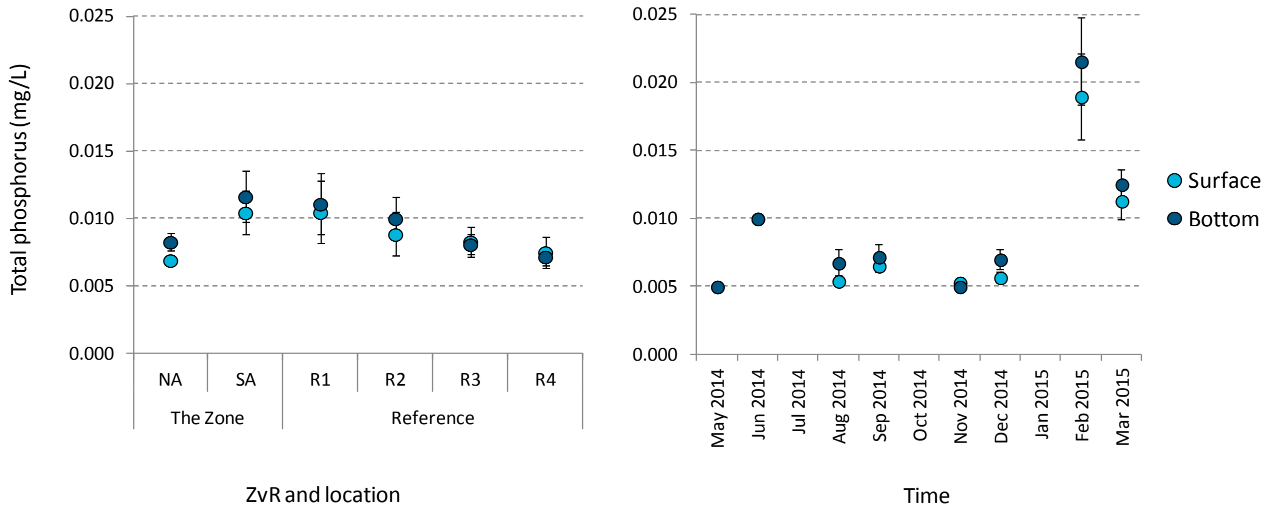
Results revealed distinct spatial and seasonal fluctuations in total phosphorus (TP) concentrations (Table 5.4). In general, both surface and bottom concentrations in TP remained relatively similar across Zone and reference locations (Figure 5-11). PERMANOVA results detected a significant Time x Location (ZvR) interaction in both surface and bottom waters (Table 5.4). The significant Time x Location (ZvR) interaction was primarily driven by time and location, with higher TP concentrations reported in February (surface = 0.019 ± 0.003 mg/L, bottom = 0.022 ± 0.003 mg/L) and March 2015 (surface = 0.011 ± 0.001 mg/L, bottom = 0.013 ± 0.001 mg/L) across all Zone and reference locations.

Table 5.4 Results of a three-factor PERMANOVA examining total phosphorus concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	9.56E-04	**0.0001*****	1.04E-03	**0.0001*****
ZvR	1	2.92E-07	0.9294	1.10E-04	0.0936
Location(ZvR)	4	1.34E-04	**0.0042****	1.22E-04	**0.0268***
TimexZvR	7	4.36E-06	0.9954	5.83E-05	0.1679
TimexLocation(ZvR)	28	1.30E-04	**0.0002*****	1.16E-04	**0.0015****
Res	168	3.23E-05		3.78E-05	
Total	215				

Notes:

1. Significant results shown in bold; **= highly significant (p<0.01), *** = very highly significant (p<0.001).
2. ZvR = Zone vs Reference



Note:

1. ZvR = Zone vs Reference

Figure 5-11 Total phosphorus (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR and time

Total organic carbon

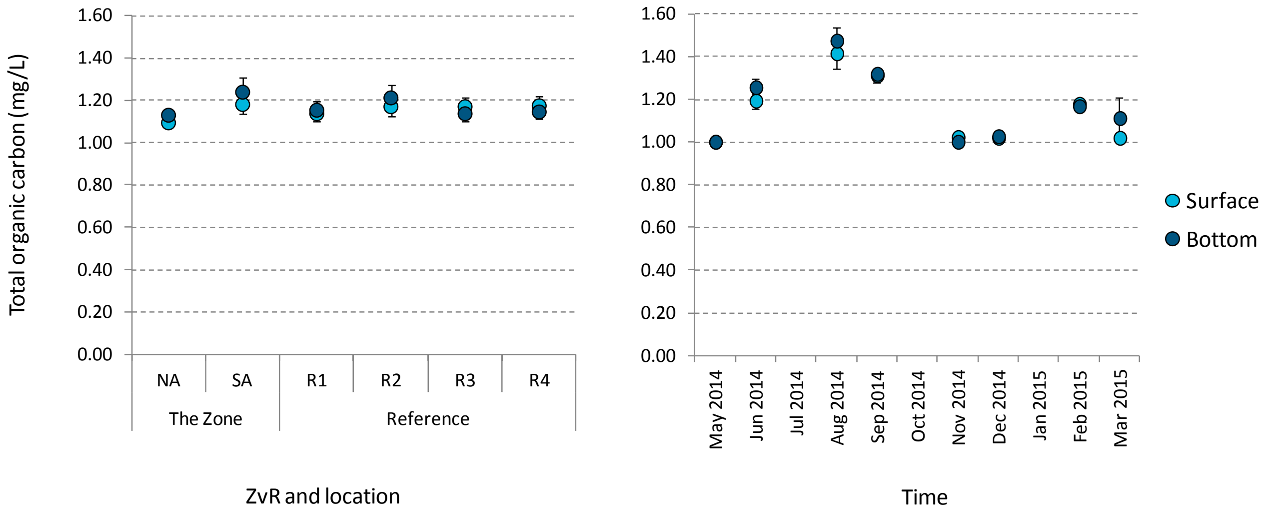
Concentrations of total organic carbon (TOC) varied significantly across time (Table 5.5). Sampling in August (surface = 1.40 ± 0.07 mg/L, bottom = 1.47 ± 0.06 mg/L) and September 2014 (surface = 1.31 ± 0.03 mg/L, bottom= 1.32 ± 0.03 mg/L) reported the greatest concentration of TOC in both surface and bottom waters (Figure 5-12). PERMANOVA also detected a significant Time x Location (ZvR) and Time x ZvR interaction (Table 5.5). Both interactions were driven by time, as TOC concentrations were below the detection limit across all Zone and reference locations during November and December 2014 and March and May 2015.

Table 5.5 **Results of a three-factor PERMANOVA examining total organic carbon concentrations at the surface and bottom of the water column**

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	3.7481	**0.0001*** **	3.6251	**0.0001*** **
ZvR	1	7.50E-02	0.2198	1.66E-02	0.6690
Location(ZvR)	4	0.10727	0.0814	8.17E-02	0.4925
TimexZvR	7	0.14088	**0.0097****	8.95E-02	0.4529
TimexLocation(ZvR)	28	8.62E-02	**0.0185***	0.1023	0.3389
Res	168	5.01E-02		9.31E-02	
Total	215				

Notes:

1. Significant results shown in bold; **= highly significant (p<0.01), *** = very highly significant (p<0.001).



Note:

1. ZvR = Zone vs Reference

Figure 5-12 **Total organic carbon (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR and time**

Total suspended solids

Concentrations of total suspended solids (TSS) remained relatively constant across locations, varying between 1.05 mg/L and 2.62 mg/L in surface and bottom waters (Figure 5-13). While no significant differences in TSS concentrations were detected in bottom waters, TSS concentrations were significantly different across time in surface waters (Table 5.6). Post-hoc tests revealed that TSS concentration measured during February 2015 was significantly different to other times[2].

---

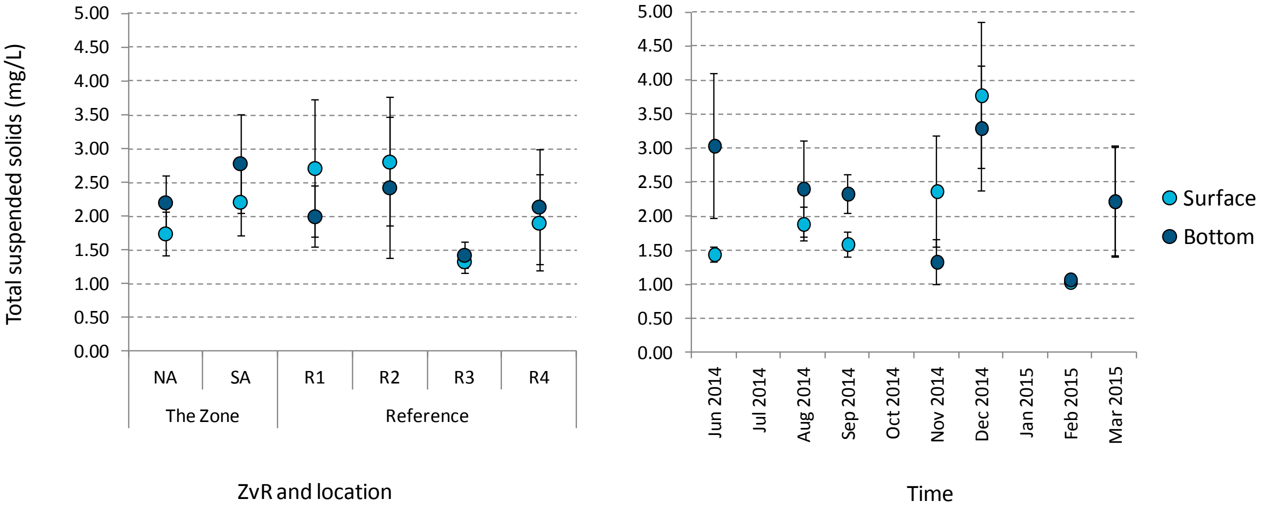
[2] *No TSS concentrations were measured during May 2014 due to inadequate flushing of salts with deionised water.*

Table 5.6 Results of a three-factor PERMANOVA examining total suspended solids concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	6	21.08	**0.0445***	22.775	0.1222
ZvR	1	2.51E+00	0.6174	0.47421	0.7579
Location(ZvR)	4	1.04E+01	0.3660	5.1543	0.8060
TimexZvR	6	5.76E+00	0.7372	7.4111	0.7869
TimexLocation(ZvR)	24	16.678	0.0510	14.388	0.3889
Res	147	9.59E+00		13.677	
Total	188				

Notes:

1. Significant results shown in bold; **= highly significant (p<0.01), *** = very highly significant (p<0.001).
2. ZvR = Zone vs Reference



Note:

1. ZvR = Zone vs Reference

Figure 5-13 Total suspended solids (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR and time

Volatile suspended solids

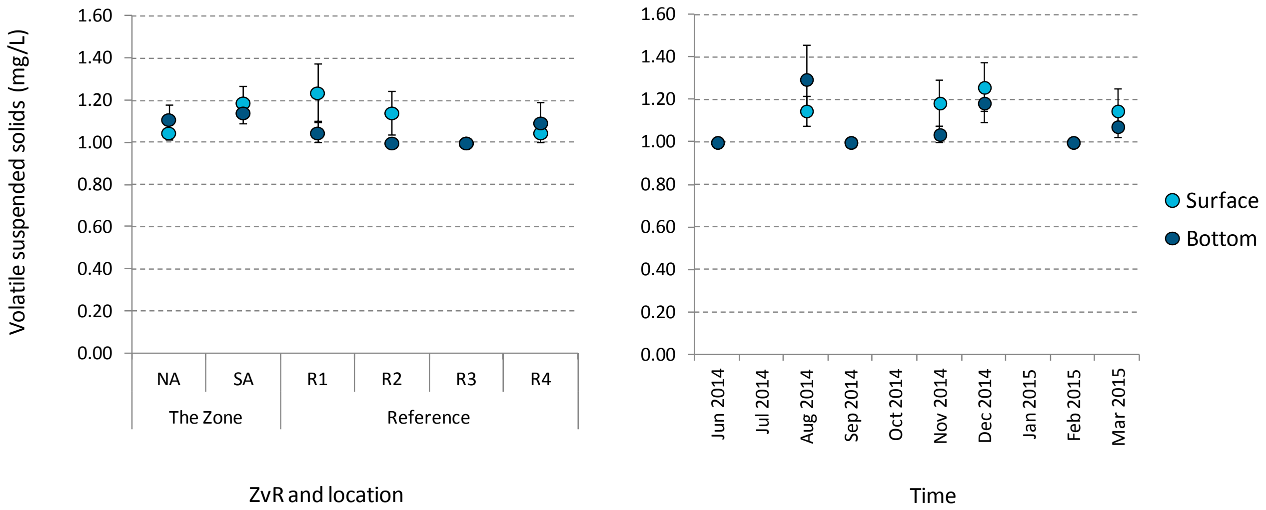
Concentrations of volatile suspended solids (VSS) varied in time and space (Table 5.7): for example, the highest concentrations in surface waters were detected in December 2014 (1.26 ± 0.11 mg/L), and the lowest concentrations in bottom waters were recorded in August 2014 (1.30 ± 0.16 mg/L). A significant Time x Location (ZvR) interaction was detected at the surface of the water column (Table 5.7). Post-hoc tests revealed that the driver of this interaction was time and location, which resulted from unusually high VSS concentrations at reference site R1 (2.33 ± 0.67 mg/L) during one of the months (November 2014).

Table 5.7 **Results of a three-factor PERMANOVA examining volatile suspended solids concentrations at the surface and bottom of the water column**

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	6	0.99892	**0.0010****	0.88938	**0.0037****
ZvR	1	1.46E-02	0.7981	5.89E-02	0.4324
Location(ZvR)	4	0.55818	**0.0476***	2.14E-02	0.9906
TimexZvR	6	0.2381	0.4003	0.30673	0.3069
TimexLocation(ZvR)	24	0.42068	**0.0295***	0.13743	0.8880
Res	147	0.22676		0.25181	
Total	188				

Notes:

1. Significant results shown in bold
2. *Significant = p<0.05; **Highly significant = p<0.01



Note:

1. ZvR = Zone vs Reference

Figure 5-14 **Volatile suspended solids sampled at the surface and bottom of the water column across locations within ZvR and time**

Ammonia

Ammonia concentrations at the surface of the water column were relatively consistent in space, though concentrations were marginally elevated at the northern and southern areas (Figure 5-15). Higher concentrations were also detected in June 2014 (5.56 ± 0.79 µg/L) and August 2014 (7.00 ± 2.43 µg/L) relative to other months, resulting in a significant Time x ZvR interaction (Table 5.8). Similar results were observed in the case of bottom waters, with significant Time x ZvR and Time x Location (ZvR) interactions (Table 5.8). These interactions were driven by both time and ZvR, but mainly due to the elevated concentrations at the northern area (SL1) in June 2014 (9.67 ± 1.60 µg/L).

Table 5.8 Results of a three-factor PERMANOVA examining ammonia concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	75.107	**0.0040****	28.562	**0.0001*****
ZvR	1	66.477	0.0824	1.0524	0.5984
Location(ZvR)	4	14.786	0.6476	6.0497	0.2120
TimexZvR	7	60.204	**0.0101***	14.604	**0.0040****
TimexLocation(ZvR)	28	37.274	0.1259	10.587	**0.0011****
Res	168	22.312		4.2707	
Total	215				

Notes:

1. Significant results shown in bold
2. *Significant = p<0.05; **Highly significant = p<0.01; ***Very highly significant = p<0.001

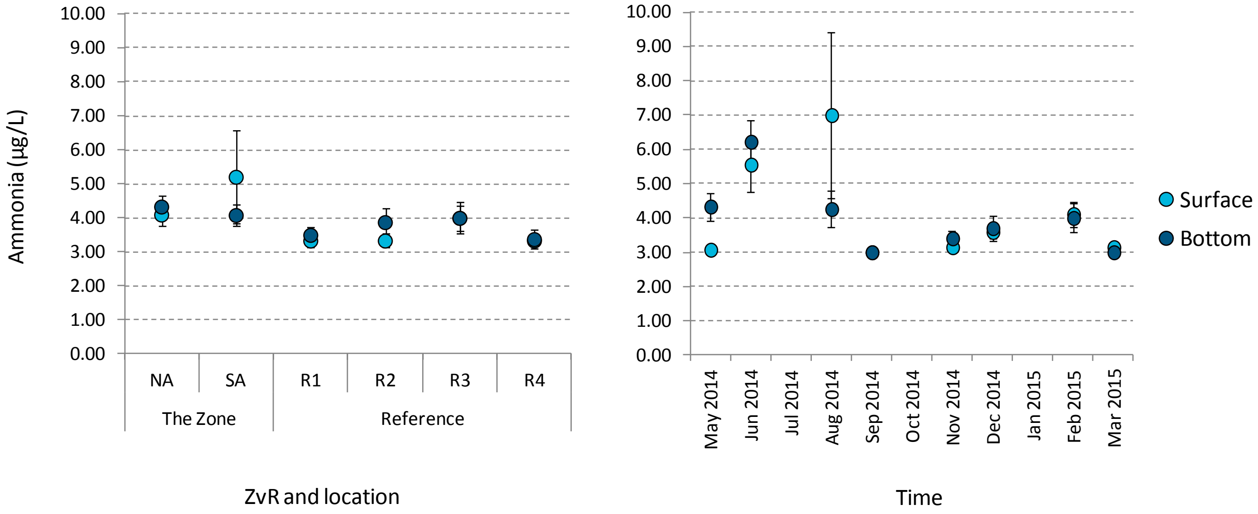


Figure 5-15 Ammonia (mean ± S.E.) (µg/L) sampled at the surface and bottom of the water column across locations within ZvR (left) and time (right)

Orthophosphate

Results revealed distinct spatial and seasonal fluctuations in orthophosphate concentrations. In general, similar surface concentrations were reported across the northern and southern areas and at the reference locations (Figure 5-16). A significant Time x Location(ZvR) interaction in surface waters was detected, primarily driven by time and location as higher orthophosphate concentrations were reported in June (3.04 ± 0.11 µg/L) and August 2014 (4.52 ± 0.50 µg/L) at the southern area (SL2) and reference location R3. For bottom waters, significant Time x Location (ZvR) and Time x ZvR interactions were reported (Table 5.9). These interactions were primarily driven by time, as post-hoc tests found that concentrations in bottom waters were greater at the northern area (SL1) and the reference locations R2, R3 and R4 during May, August and November 2014, and March 2015. Orthophosphate concentrations significantly differed between the northern and southern areas and the reference locations across time, with the exception of June 2014.

Table 5.9 Results of a three-factor PERMANOVA examining orthophosphate concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	38.681	**0.0001*** **	22.861	**0.0001*** **
ZvR	1	0.20455	0.6677	1.0681	0.1388
Location(ZvR)	4	3.3833	**0.0104***	1.0384	0.0583
TimexZvR	7	1.46	0.1772	1.8076	**0.0013** **
TimexLocation(ZvR)	28	2.3714	**0.0042** **	1.6314	**0.0001*** **
Res	168	0.98214		0.4988	
Total	215				

Notes:

1. *Significant = p<0.05; **Highly significant = p<0.01; ***Very highly significant = p<0.001
2. Significant results shown in bold

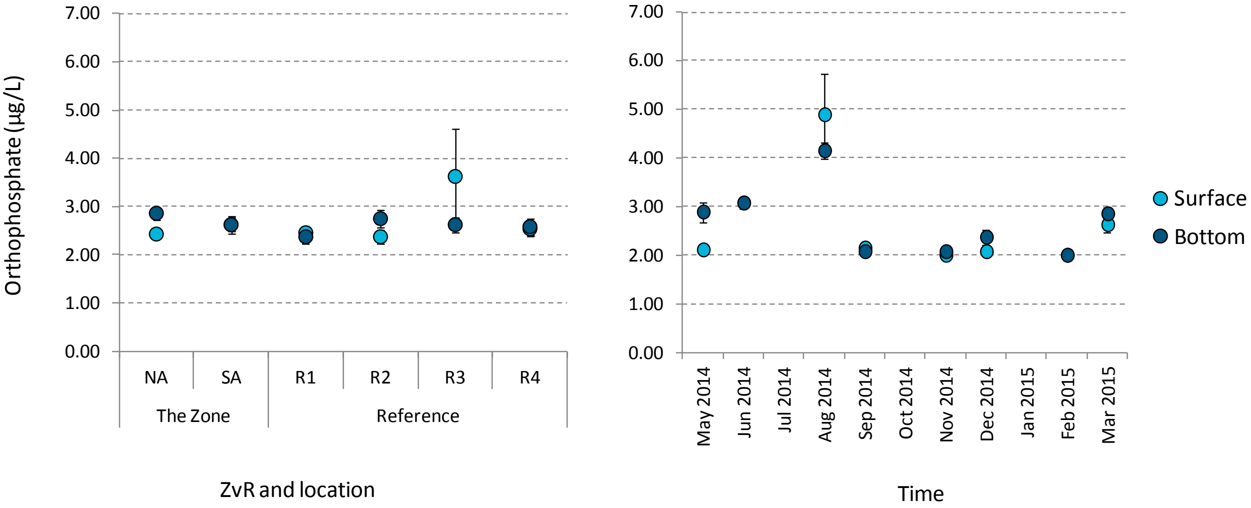


Figure 5-16 Orthophosphate (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR (left) and time (right)

Dissolved inorganic nitrogen

Concentrations of dissolved inorganic nitrogen (DIN) showed seasonal variations in surface and bottom waters (Figure 5-17; Table 5.10). Post-hoc test showed that concentrations at the surface were significantly higher during August 2014 (39.67 ± 10.60 µg/L), December 2014 (23.44 ± 1.83 µg/L) and February 2015 (21.96 ± 2.36 µg/L). For bottom waters, August 2014 reported greater DIN levels (30.59 ± 8.22 µg/L), while March 2015 (7.78 ± 0.86 µg/L) had the lowest concentration of DIN. Furthermore, higher concentrations of DIN were reported in the combined northern and southern areas (Zone = 22.58 ± 2.09 µg/L) compared to reference locations (17.60 ± 1.15 µg/L).

Table 5.10 Results of a two-factor PERMANOVA examining dissolved inorganic nitrogen concentrations at the surface and bottom of the water column

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	2083.2	**0.0001*** **	1231.7	**0.0004*** **
ZvR	1	41.698	0.8222	1644.6	**0.0144***
Location(ZvR)	4	1160.3	0.0690	475.14	0.2492
TimexZvR	7	561.33	0.3727	388.9	0.3337
TimexLocation(ZvR)	28	442.67	0.5065	213.04	0.6752
Res	168	497.28		330.44	
Total	215				

Notes:

1. *Significant = p<0.05; ***Very highly significant = p<0.001
2. Significant results shown in bold

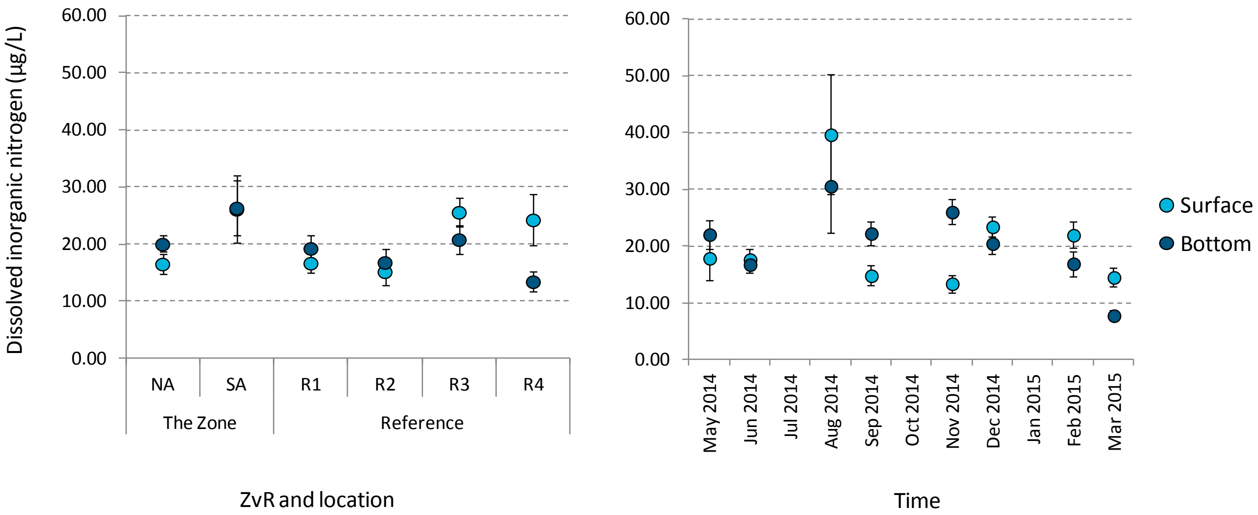


Figure 5-17 Dissolved inorganic nitrogen (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR (left) and time (right)

Nitrate and nitrite

Concentrations of nitrate and nitrite (NOx) were greatest in August 2014 irrespective of depth (surface 32.67 ± 8.62 µg/L and bottom 26.33 ± 7.78 µg/L). There was also a tendency toward spatial variation in concentrations (Figure 5-18). On average, reference locations R3 and R4 reported the greatest concentrations in surface waters (21.63 ± 2.50 µg/L and 20.96 ± 1.72 µg/L, respectively), followed closely by the southern area SL2 (20.94 ± 4.69 µg/L). PERMANOVA detected a significant seasonal decline in bottom water concentrations between November 2014 and March 2015 (Figure 5-18).

Table 5.11 **Results of a two-factor PERMANOVA examining nitrate and nitrite concentrations at the surface and bottom of the water column**

Source	df	Surface		Bottom	
		MS	P(perm)	MS	P(perm)
Time	7	1121.8	**0.0002*** **	738.59	**0.0020** **
ZvR	1	80.01	0.5147	323.65	0.0763
Location(ZvR)	4	515.8	**0.0239***	126.3	0.3199
TimexZvR	7	213.85	0.3040	145.87	0.2584
TimexLocation(ZvR)	28	269.79	0.0972	101.78	0.5667
Res	168	177.55		115.09	
Total	215				

Notes:

1. *Significant = p<0.05; **Highly significant = p<0.01; ***Very highly significant = p<0.001
2. Significant results shown in bold

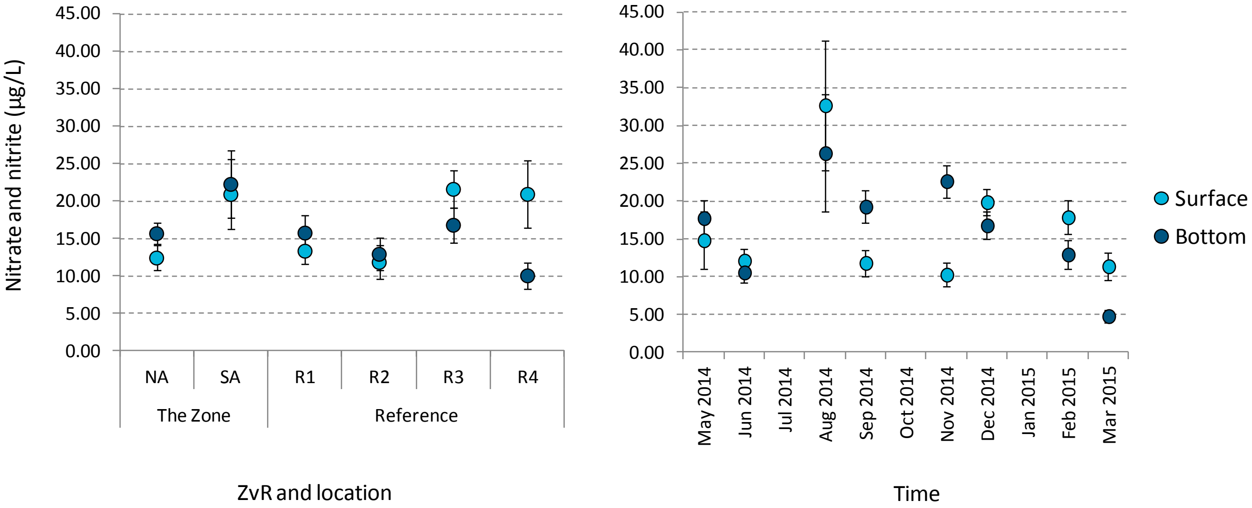


Figure 5-18 **Nitrate and nitrite (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR (left) and time (right)**

5.3.4 Hydrogen sulphide

Concentrations of hydrogen sulphide were below the limit of reporting (0.01 mg/L) in all samples.

5.3.5 Total Petroleum Hydrocarbons / Polycyclic Aromatic Hydrocarbons

Total petroleum hydrocarbons (TPH) and polycyclic aromatic hydrocarbons (PAH) were generally below the laboratory limit of reporting (LOR). Of the over 400 replicate water samples collected, less than 20 samples exceeded the LOR for PAHs (0.001 µg/L), 1 sample exceeded the LOR for TPH C6-C10, 5 samples exceeded the LOR for TPH C11-C16, 2 samples exceeded the LOR for TPH C17-C34 and 1 sample exceeded the LOR for total TPH (Table 5.24).

Table 5.12 Total Petroleum Hydrocarbons and Polycyclic Aromatic Hydrocarbons concentrations in the surface and bottom of the water column

Chemical	Species	LOR (µg/L)	Site and value (µg/L)
TPH	C6-C10	25	R1 bottom (120)
	C11-C16	25	NA bottom (89) SA surface (32) R1 bottom (41) R3 surface (34) R4 bottom (33 and 46)
	C17-34	100	NA bottom (160) R2 surface (120)
	Total	250	NA bottom (290)
PAH	Benzo(g,h,i)perylene	0.001	NA 3 reps, 1 rep SA (0.002 0.011)
	Phenanthrene		Numerous samples (0.001 – 0.017)
	Benzo(k)fluoranthene		R4 1 rep (0.024)
	Benzo(b)fluoranthene		R4 1 rep (0.038)
	Naphthalene		Numerous samples (0.001 – 0.88)

5.3.6 Chlorophyll-a

Univariate analyses applied to chlorophyll-a concentrations revealed a significant Time x Location interaction term (Table 5.13). This result indicates that there were differences among times, but that these were different for each location. Reference location R1 had greater concentrations of chlorophyll-a at the surface (0.27 ± 0.03 µg/L) and bottom (0.25 ± 0.04 µg/L) of the water column relative to other locations (Figure 5-19). A general increasing trend in chlorophyll-a was also observed at the surface and bottom of the water column from November 2014 to March 2015 (Figure 5-19).

Table 5.13 Results of a two-factor PERMANOVA examining chlorophyll-a concentrations at the surface and bottom of the water column

Source	df	Surface		df	Bottom	
		MS	P(perm)		MS	P(perm)
Time	6	0.24522	**0.0001*****	6	0.17707	**0.0001*****
Location	5	3.25E-02	**0.0005*****	5	3.03E-02	**0.0003*****
TimexLocation	30	2.05E-02	**0.0001*****	30	1.81E-02	**0.0001*****
Res	146	6.72E-03		147	5.48E-03	
Total	187			188		

Notes:

1. ***Very highly significant = p<0.001
2. Significant results shown in bold

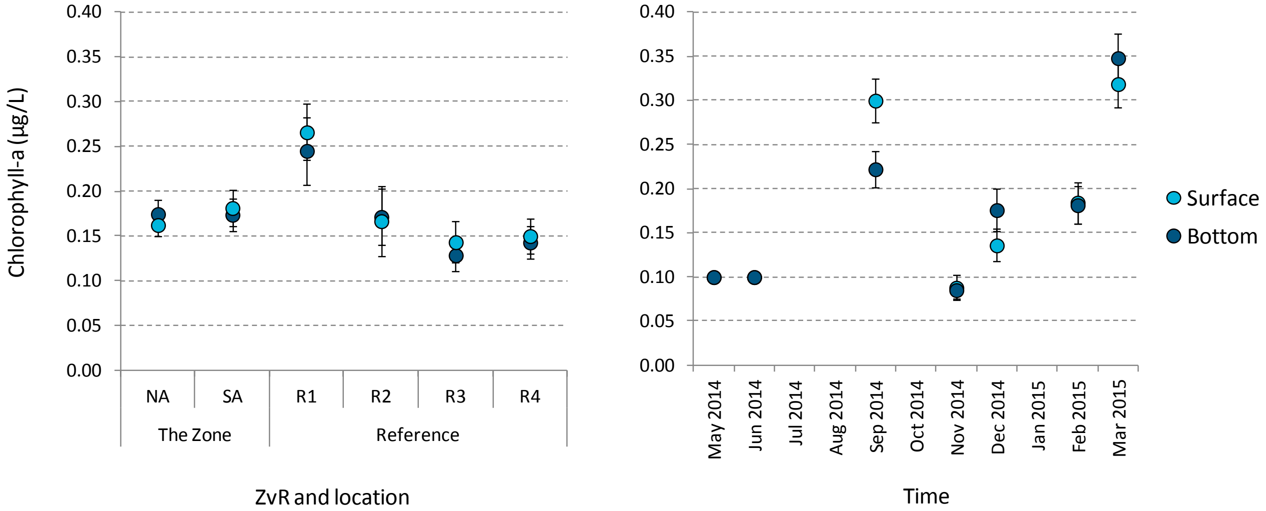


Figure 5-19 Chlorophyll-a (mean ± S.E.) sampled at the surface and bottom of the water column across locations within ZvR (left) and time (right)

5.3.7 Phytoplankton

Phytoplankton belonging to six divisions/phyla (Bacillariophyta, Chlorophyta, Chrysophyta, Cryptophyta, Cyanophyta, Dinophyta), plus unidentified others, were sampled across all locations. Counts were overwhelmingly dominated by the diatoms (Bacillariophyta represented ~90.8% of the total counts), followed by dinoflagellates (~3.5% of the total counts). Of the total counts, 12.4% were classified as potentially toxic algae and 1.6% were classified as potentially toxic blue green algae.

Results were characterised by very large scale fluctuations in community assemblage in time and space. This was reflected in the multivariate PERMANOVA routines which revealed significant differences in phytoplankton counts between months and locations (Table 5.14). Post-hoc pair wise comparisons found significant differences in phytoplankton counts across all times, except those between August 2014 and February 2015, and between December 2014 and February 2015. In addition, greater counts of Chlorophyta (green), Cryptophyta (monad), Cyanophyta (blue green) and Dinophyta (dinoflagellates) were reported during May 2014 (Figure 5-20), and greater counts of Bacillariophyta were recorded in December 2014 (92.93 ± 25.08 cells/ml; Figure 5-20). Post-hoc tests revealed that the northern and southern areas were significantly different to each other. This was particularly evident for Dinophyta, which was recorded in higher numbers at the southern areas relative to northern area (Figure 5-20). Phytoplankton counts at reference location R1 were also significantly different to counts at reference locations R2, R3 and R4. This was driven primarily by Bacillariophyta, which recorded very high numbers at location R1 relative to other locations (Figure 5-20).

Table 5.14 Results of a three-factor PERMANOVA examining phytoplankton counts

Source	df	MS	Pseudo-F	P(perm)
Time	3	2189.1	5.1900	**0.0002*** **
ZvR	1	624.22	1.4799	0.2343
Location(ZvR)	4	1310.7	3.1074	**0.0017** **
TimexZvR	3	539.33	1.2786	0.2668
TimexLocation(ZvR)	12	566.06	1.3420	0.1328
Res	84	421.8		
Total	107			

Notes:

1. **Highly significant = p<0.01; ***Very highly significant = p<0.001
2. Significant results shown in bold

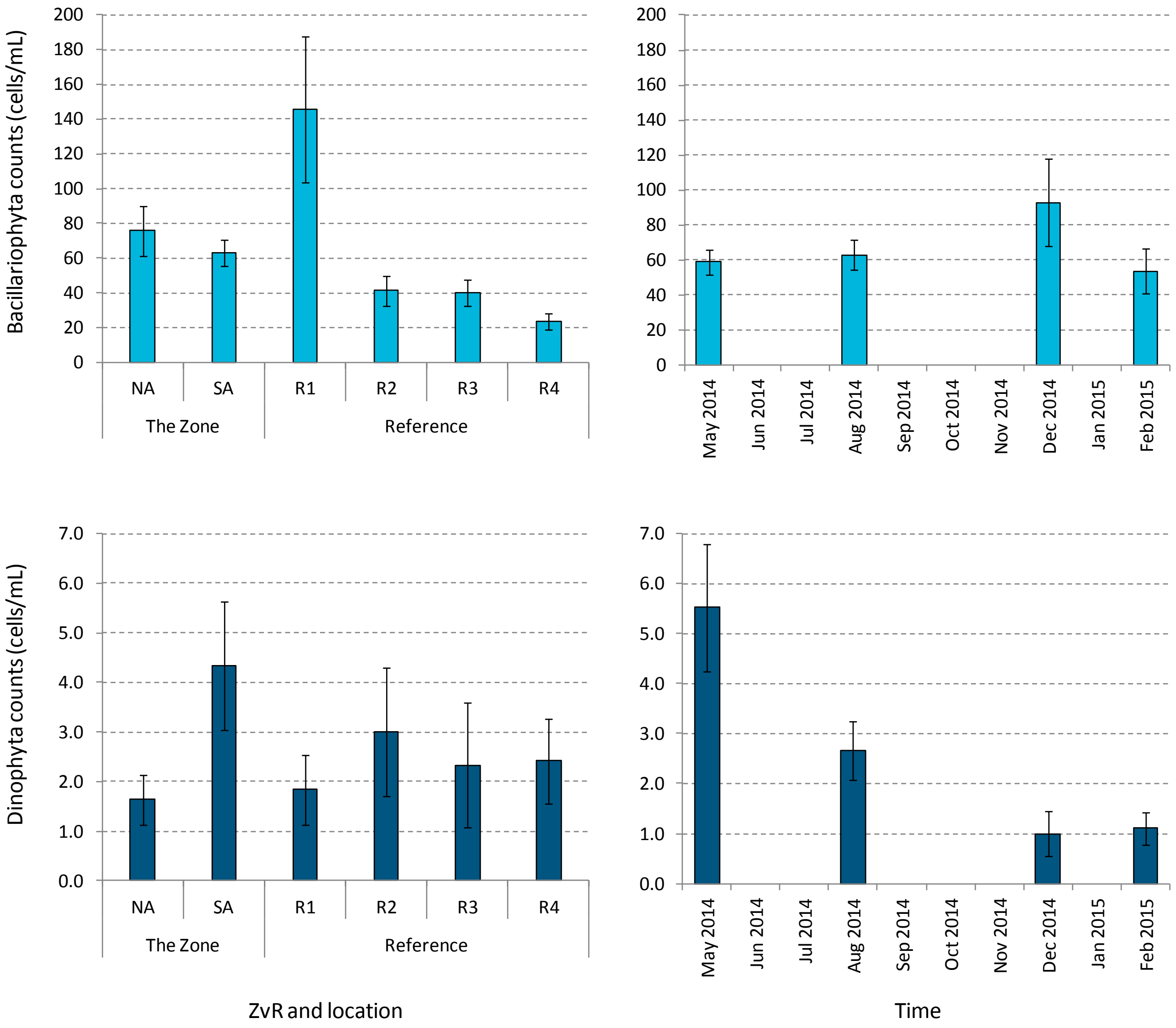


Figure 5-20 **Bacillariophyta (diatoms; top left and right) and Dinophyta (dinoflagellates; bottom left and right) counts (mean ± S.E.) across locations and time**

The multivariate analysis applied to phytoplankton biovolume revealed similar results as seen in the community data, however a significant Time x Location(ZvR) interaction was detected that was primarily driven by time and location (Table 5.15). Post-hoc test revealed significant differences across times (Figure 5-21) and R1 and R4. Higher biovolumes of Bacillariophyta and Dinophyta were recorded at R1 (Figure 5-21).

Table 5.15 **Results of a three-factor PERMANOVA examining biovolume of phytoplankton**

Source	df	MS	Pseudo-F	P(perm)
Time	3	303.11	4.1633	**0.0015****
ZvR	1	34.046	0.46762	0.6408
Location(ZvR)	4	248.4	3.4118	**0.0029****
TimexZvR	3	134.91	1.853	0.0983
TimexLocation(ZvR)	12	207.07	2.8441	**0.0002**
Res	84	72.807		
Total	107			

Notes:

1. **Highly significant = p<0.01
2. Significant results shown in bold

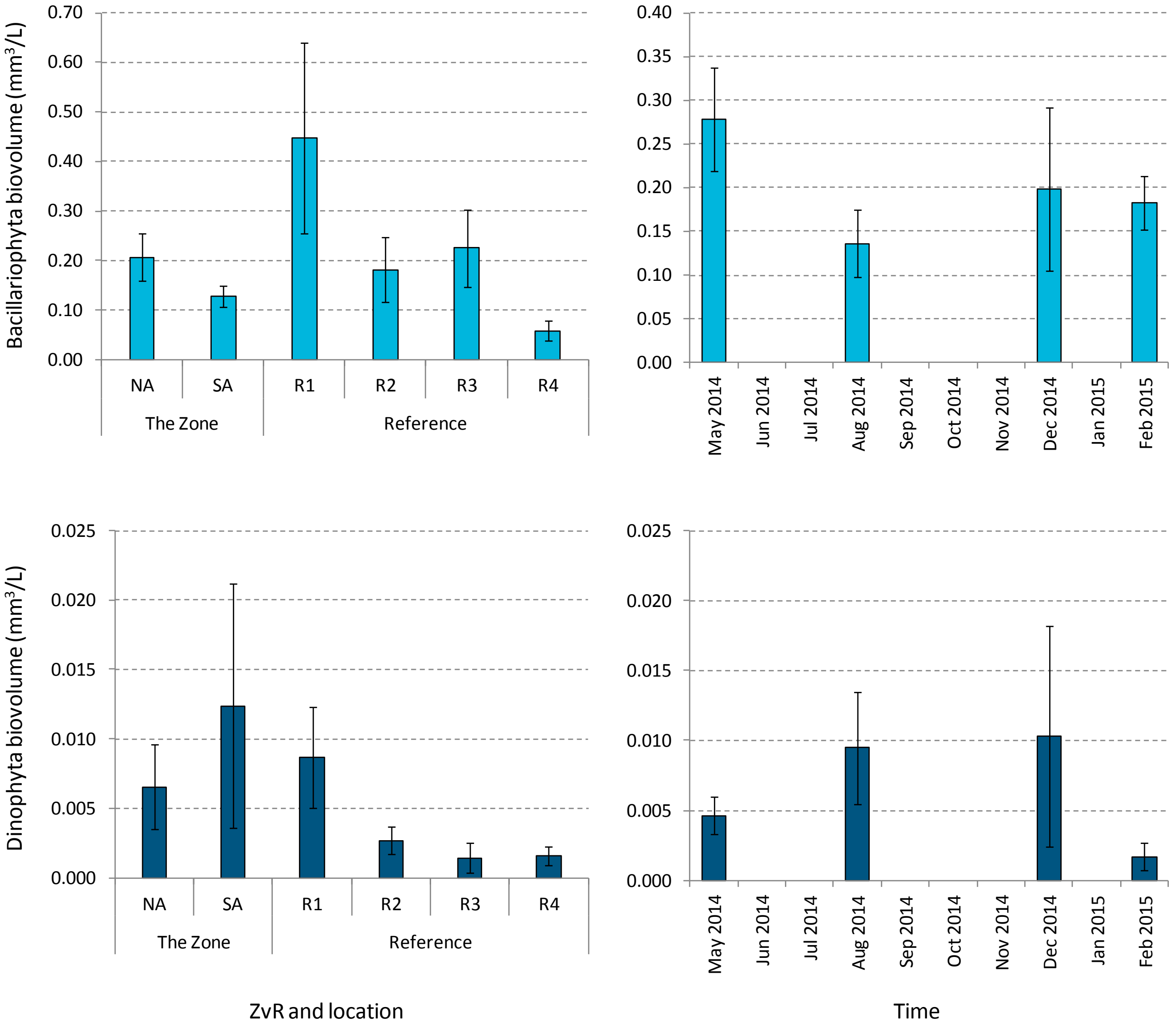


Figure 5-21 **Bacillariophyta (diatoms; top left and right) and Dinophyta (dinoflagellates; bottom left and right) biovolumes (mean ± S.E.) across locations**

Multivariate analysis of total algal and potential toxic algal counts revealed significant differences between time and locations (Table 5.16). Post-hoc tests for location showed significant differences in algal counts between reference R1 and all other three reference locations (R2, R3 and R4). Post-hoc tests for time only revealed a significant difference in total counts between May 2014 and December 2014. Total algal counts were greatest during December 2014 (99.56 ± 27.08 cells/ml) while May 2014 recorded the greatest counts of potentially toxic algae (11.81 ± 4.92 cells/ml; Figure 5-22).

Table 5.16 **Results of a three-factor PERMANOVA examining total algal and potentially toxic algal counts**

Source	df	MS	Pseudo-F	P(perm)
Time	3	1248.3	2.5149	**0.0229***
ZvR	1	843.5	1.6993	0.1669
Location(ZvR)	4	2169.8	4.3713	**0.0006*****
TimexZvR	3	999.79	2.0142	0.0686
TimexLocation(ZvR)	12	700.93	1.4121	0.1098
Res	84	496.38		
Total	107			

Notes:

1. *Significant = $p<0.05$; ***Very highly significant = $p<0.001$
2. Significant results shown in bold

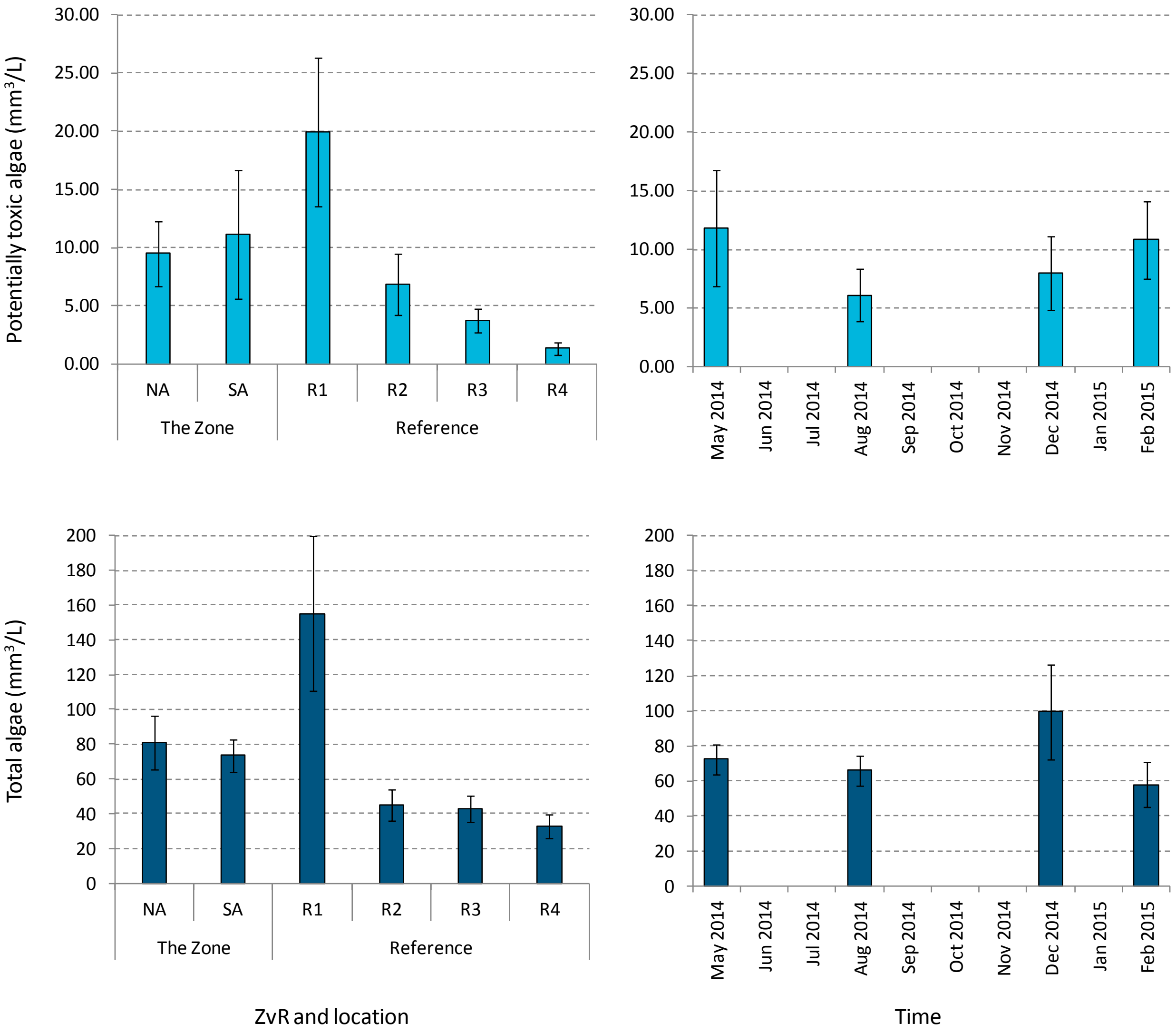


Figure 5-22 Biovolumes (mean ± S.E.) of potentially toxic algae (top left and right) and total algae (bottom left and right) across locations and time

5.4 Sediment quality

5.4.1 Particle size analysis

In general, there were no major differences in sediment particle sizes between the MWADZ and reference locations (Figure 5-23), with sediments in all areas composed of varying proportions of different particle size fractions (Figure 5-23). Some differences in time were detected – fine to coarse sand dominated in the winter season, while fine clays and silts dominated in the summer season. This was reflected in the multivariate analyses applied to sediment particle size data, which revealed significant interaction terms for Season x Location(ZvR) and Season x ZvR (Table 5.17). Post-hoc tests revealed that sediment particle sizes differed across all locations and across the winter and the summer season, again reflecting the general high level of variability.

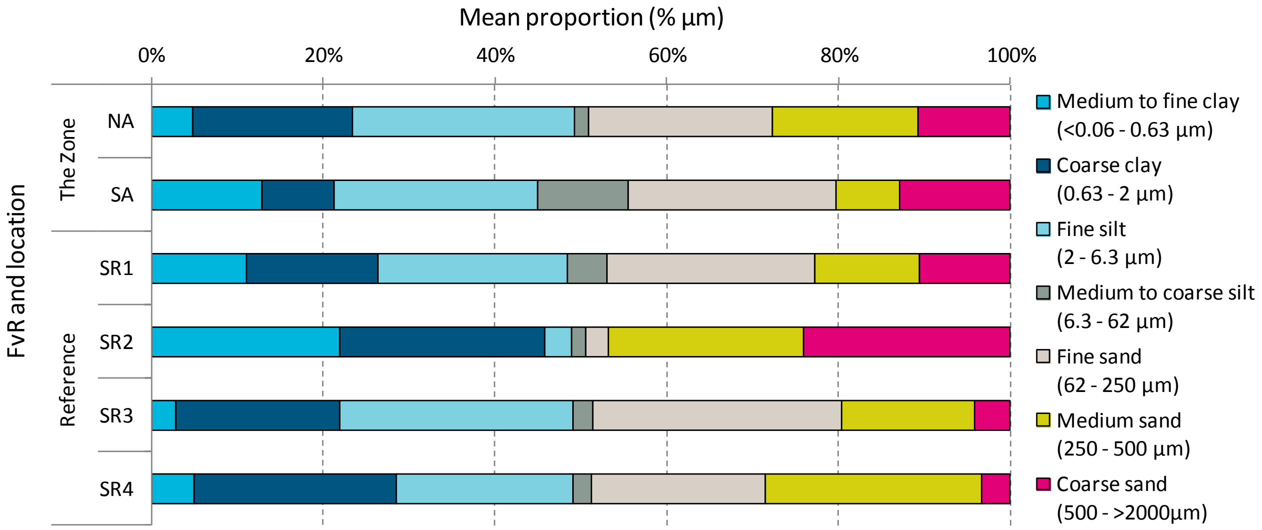


Figure 5-23 Mean proportion (% µm) of seven sediment grain size fractions across locations within ZvR

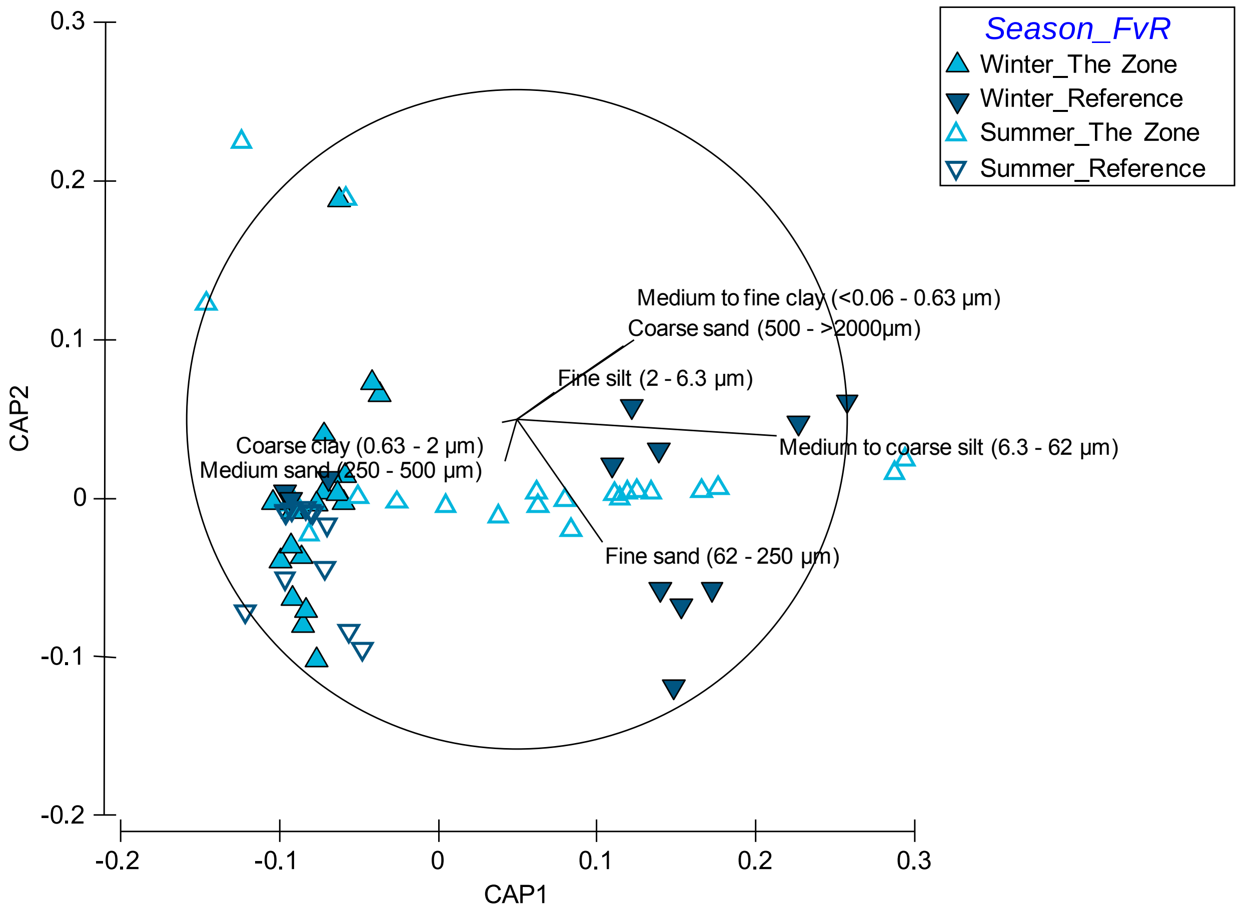
Table 5.17 Results of a four-factor PERMANOVA examining particle size distribution

Source	df	MS	Pseudo-F	P(perm)
Season	1	90802	2098.4	**0.0001*** **
ZvR	1	1357.7	15.575	**0.0042** **
Location(ZvR)	4	1694.3	19.436	**0.0001*** **
SeasonxZvR	1	556.03	12.849	**0.0144* **
Site(Location(ZvR))	5	87.172	0.94594	0.5225
SeasonxLocation(ZvR)	4	548.96	12.686	**0.0162* **
SeasonxSite(Location(ZvR))	5	43.273	0.46957	0.9274
Res	44	92.154		
Total	65			

Notes:

1. *Significant = p<0.05; **Highly significant = p<0.01; ***Very highly significant =p<0.001
2. Significant results shown in bold

The CAP ordination plot for Season x ZvR showed a separation of the combined northern and southern areas (represented by Zone) and the reference locations in the winter period. Clays (<0.06–0.63 µm) to coarse sands (500>2000 µm) tended to dominate at the reference sites in the winter months whereas coarse clay (0.63–2 µm) and medium-sized sand (250–500 µm) dominated in the summer months (Figure 5-24).

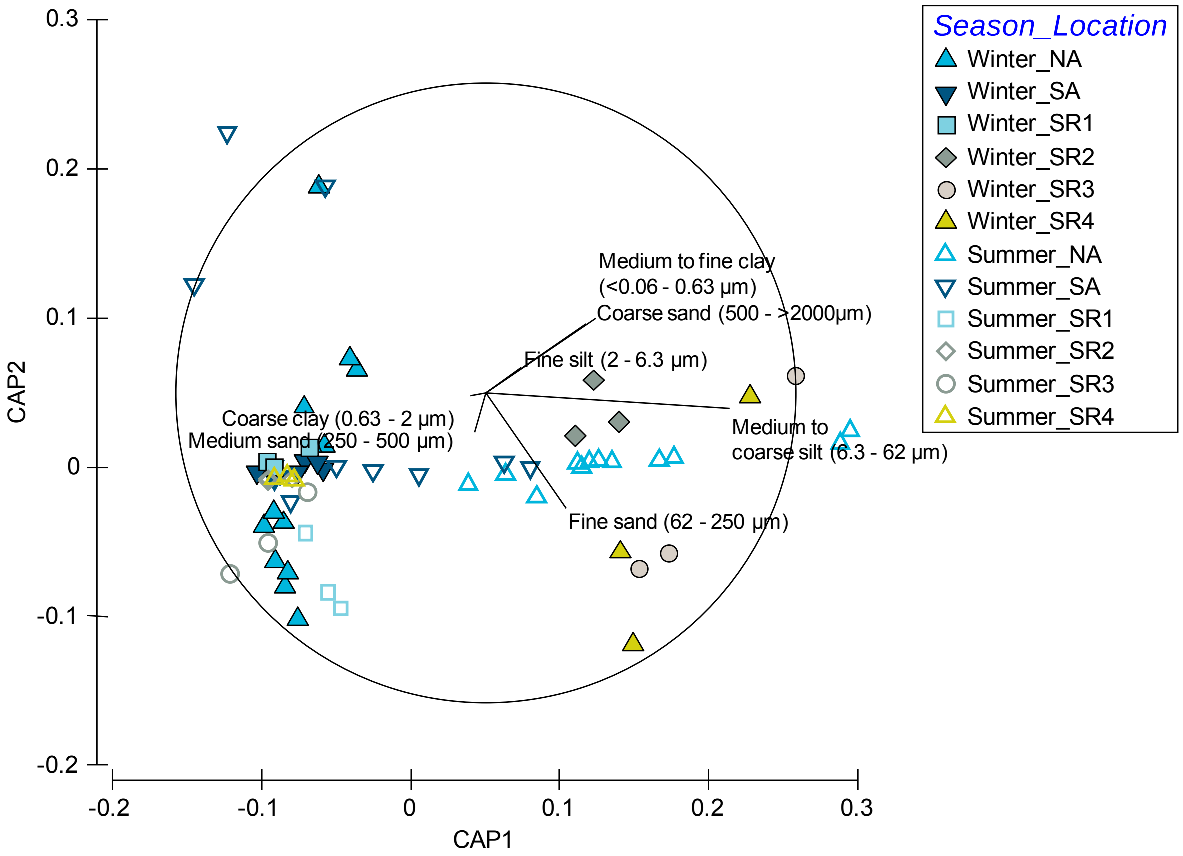


Notes:

1. Zone (combined northern and southern areas); Reference (combined R1-R4 locations)
2. CAP (Canonical analysis of principal coordinates)

Figure 5-24 CAP ordination plot of the particle size distribution among the winter and summer seasons and future lease and reference locations (ZvR) with vector overlays

The CAP ordination plot for Season x Location (ZvR) showed a separation across seasons for locations. Reference locations SR2, SR3 and SR4 were characterised by fine clays (<0.06–0.63 µm) to coarse sand (500–>2000 µm) during the winter months. Both the combined northern and southern areas (represented by the Zone) and the reference locations were characterised by coarse clay (0.63–2 µm) and medium-sized sand (250–500 µm) sampled in the summer months (Figure 5-25).



Notes:

1. NA (northern area); SA (southern area); SR (sediment reference)
2. CAP (Canonical analysis of principal coordinates)

Figure 5-25 CAP ordination plot of the particle size distribution among seasons and locations with vector overlays

5.4.2 Nutrients

Individual PERMANOVA routines revealed highly significant differences between seasons for ammonium, nitrogen and TOC concentrations (Table 5.18, Table 5.19), and a significant difference between locations for both phosphorus and TOC (Table 5.19). Post-hoc pair wise comparisons reported higher TOC concentrations in the southern area in both seasons compared to the northern area.

Table 5.18 Results of a four-factor PERMANOVA examining ammonium and nitrogen concentrations

Source	df	Ammonium		Nitrogen	
		MS	P(perm)	MS	P(perm)
Season	1	5.1822	**0.0011****	2.62E-04	**0.0004*****
ZvR	1	2.30E-02	0.8176	3.59E-06	0.5749
Location(ZvR)	4	1.2483	0.0955	3.19E-05	0.0939
SeasonxZvR	1	6.62E-02	0.4614	4.85E-06	0.3823
Site(Location(ZvR))	5	0.37727	0.1290	1.07E-05	0.1626
SeasonxLocation(ZvR)	4	0.44344	0.0653	4.74E-06	0.5263
SeasonxSite(Location(ZvR))	5	0.10241	0.7768	5.30E-06	0.5447
Res	42	0.20536		6.49E-06	
Total	63				

Notes:

1. **Highly significant = p<0.01***; Very highly significant = p<0.001
2. Significant results shown in bold

Table 5.19 Results of a four-factor PERMANOVA examining phosphorus and total organic carbon concentrations

Source	df	Phosphorus		Total organic carbon	
		MS	P(perm)	MS	P(perm)
Season	1	50.784	0.5766	1.05E-04	**0.0117***
ZvR	1	12583	0.1452	3.59E-05	0.3504
Location(ZvR)	4	32605	**0.0397***	2.77E-04	**0.0073****
SeasonxZvR	1	3341.1	**0.0047****	5.30E-07	0.8008
Site(Location(ZvR))	5	4948.2	**0.0012****	3.37E-05	0.5256
SeasonxLocation(ZvR)	4	2015.6	**0.0067****	1.86E-05	0.1774
SeasonxSite(Location(ZvR))	5	121.75	0.9884	7.27E-06	0.9639
Res	42	1021.4		4.02E-05	
Total	63				

Notes:

1. *Significant = p<0.05; **Highly significant = p<0.01
2. Significant results shown in bold

A seasonal effect was evident for ammonium and nitrogen concentrations (Figure 5-26). On average, higher concentrations of ammonium were reported in winter (1.61 ± 0.12 mg/kg) relative to summer (1.06 ± 0.05 mg/kg). In contrast, a higher percentage of nitrogen was observed in sediments during summer (0.022 ± 0.001 %) than winter (0.018 ± 0.001 %; Figure 5-26). While no seasonal variations were detected for phosphorus concentrations, phosphorus varied among locations – lower concentrations were reported at reference location SR1 (272.50 ± 4.43 mg/kg) and higher concentrations were reported at reference location SR3 (472.00 ± 13.19 mg/kg).

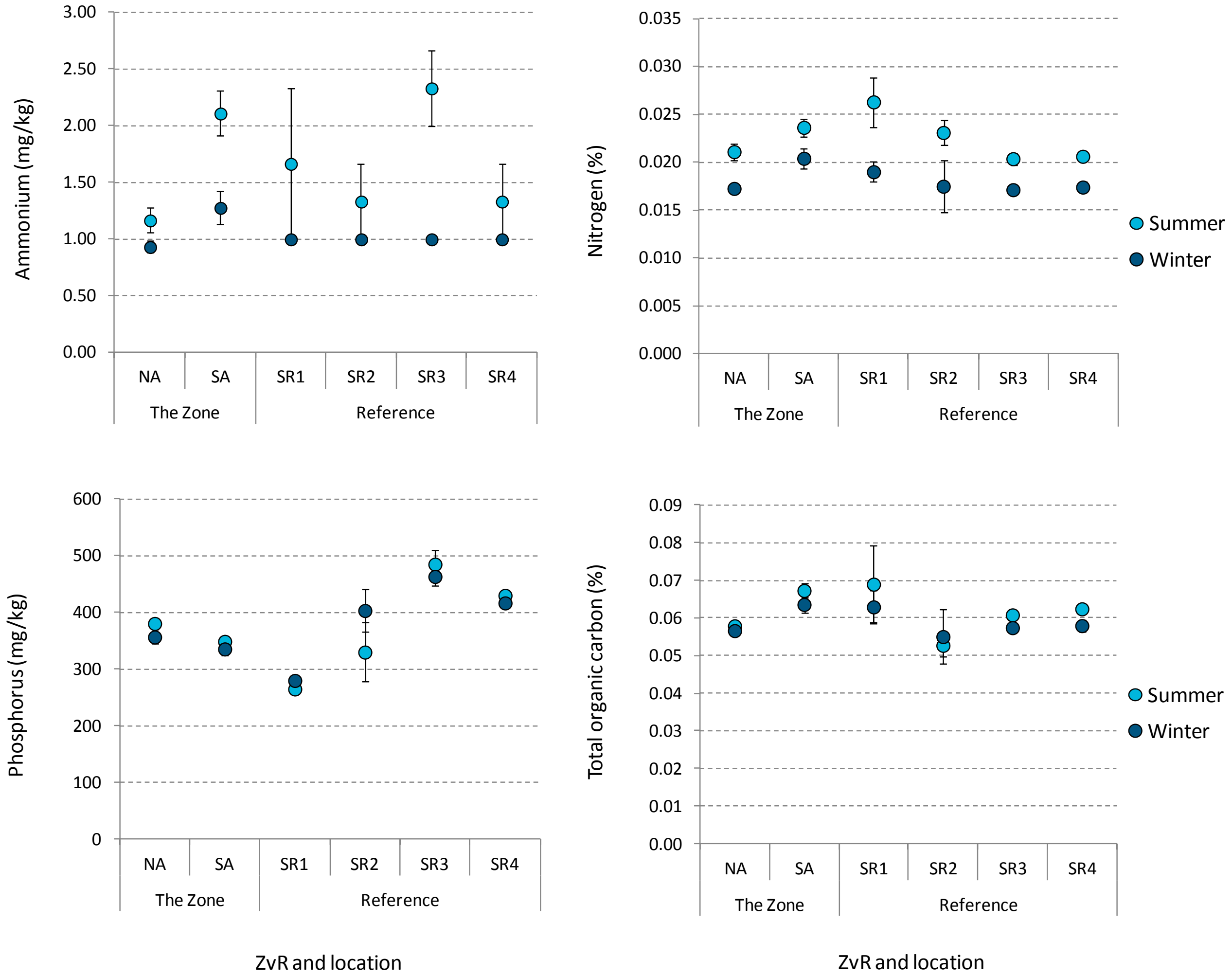


Figure 5-26 Ammonium (mg/kg; top left), nitrogen (%; top right), phosphorus (mg/kg; bottom left) and total organic carbon (%; bottom right) concentrations (mean ± S.E.) across seasons and locations

5.4.3 Metals

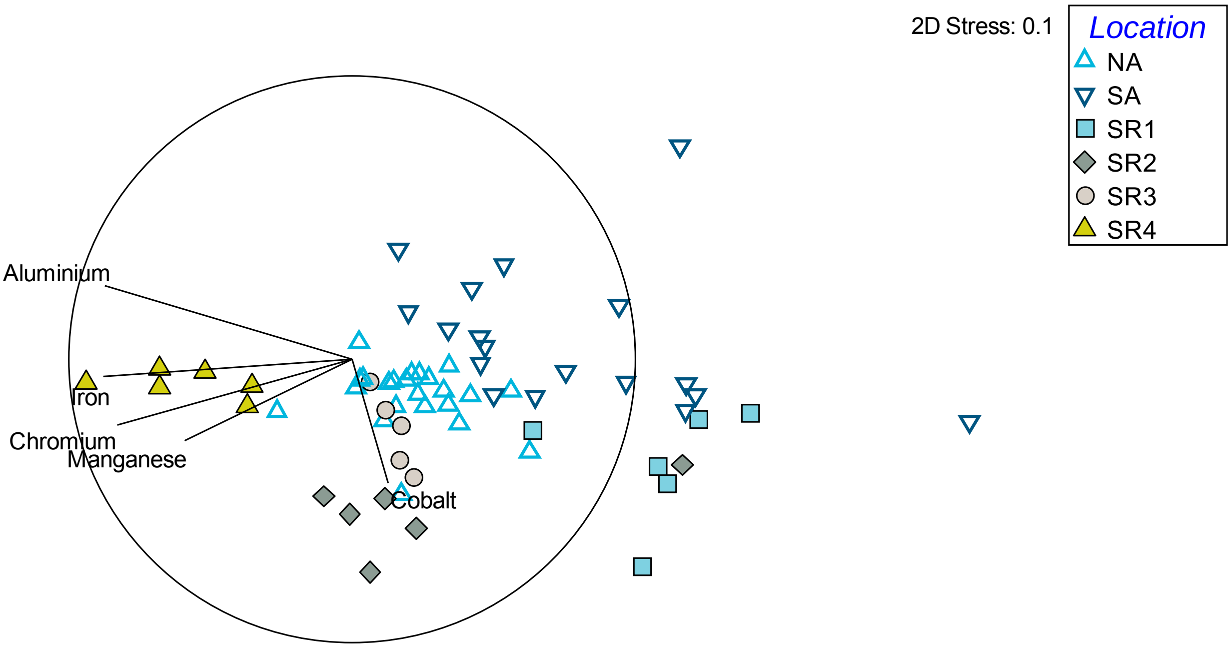
Trace metals in the MWADZ sediments were variable in space in time, but were otherwise low in concentration. Multivariate analysis revealed a significant Season x ZvR interaction term (Table 5.19), indicating there were differences between the zone and the reference locations, but only at certain times. Post-hoc tests on the interaction term revealed that the differences were restricted to the summer sampling period only. On a finer scale, differences were also detected between the northern and the southern area, and among the reference locations SR1 and SR4. SR2 and SR3 displayed similar characteristics to one another. The tendency toward inter-locational variability was reflected in the MDS plot which showed separations in trace metal concentrations across locations (Figure 5-27). The top five trace metals were aluminium (Al), iron (Fe), chromium (Cr), manganese (Mn) and Cobalt (Co). The vector overlay on the MDS plot show that the reference location SR4 had greater concentrations of Mn, Cr, Fe and Al compared to other locations, while the southern area recorded greater Co concentrations relative to other locations (Figure 5-27).

Table 5.20 Results of a four-factor multivariate PERMANOVA examining concentrations of trace metals

Source	df	SS	MS	P(perm)
Season	1	103.51	103.51	**0.0199***
ZvR	1	246.01	246.01	0.1446
Location(ZvR)	4	1821.7	455.42	**0.0222***
SeasonxZvR	1	60.896	60.896	**0.0366***
Site(Location(ZvR))	5	463.01	92.603	**0.0001*****
SeasonxLocation(ZvR)	4	71.753	17.938	0.5201
SeasonxSite(Location(ZvR))	5	95.451	19.09	0.4653
Res	42	810.8	19.305	
Total	63	3645.7		

Notes:

1. *Significant = p<0.05; ***Very highly significant = p<0.001
2. Significant results shown in bold



Note:

1. NA (northern area); SA (southern area); SR (sediment reference)
2. MDS (multi-dimensional scaling ordination)

Figure 5-27 MDS ordination of trace metal concentrations among locations with vector overlays

5.4.4 Infauna

Community assemblage

Analysis of infauna samples revealed a diverse community, comprising 10 phyla (Arthopoda, Chordata, Echinodermata, Mollusca, Nematoda, Nemertea, Phoronida, Platyhelminthes, Polychaeta and Sipuncula) and 129 families. Sampling recorded 36 families of polychaetes (accounting for 45% of the infauna sampled), 33 families of molluscs (25% of the infauna sampled), 41 families of Arthropods (18% of the infauna sampled) and 10 families of echinoderms (7% of the infauna sampled). The PERMANOVA analysis revealed high levels of variability. This was reflected in significant results for the factors Season and Location (IvR), indicating that both were important in driving the observed community structure (Table 5.21).

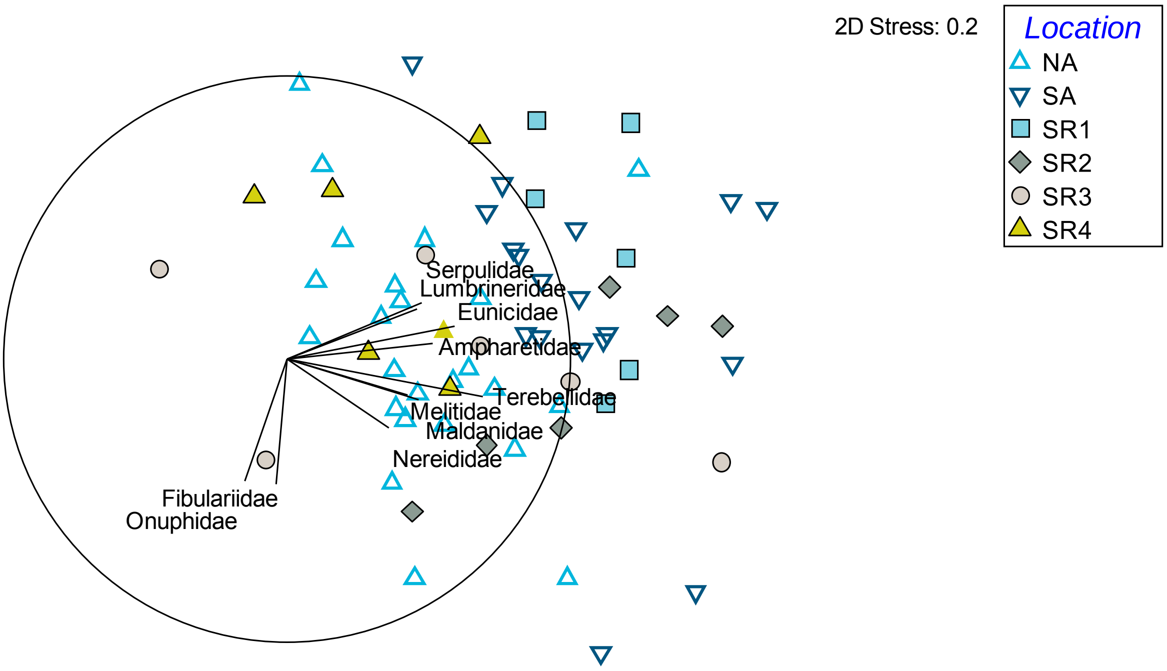
The general variability in the community is also mirrored in the MDS ordination (Figure 5-28). The MDS shows differences at the site level, but no clear separation at the location level. In general, higher counts of polychaete fauna were reported in summer than winter (Figure 5-29). The southern area contained higher numbers of polychaetes and amphipods in both seasons compared to the northern area; however, the northern area reported higher counts of echinoids, Nereididae and Onuphidae than the southern area (Figure 5-29). Reference location SR2 had the greatest counts of polychaete fauna and amphipods, followed by reference locations SR1 and SR3, however neither reference location contained echinoids (Figure 5-28).

Table 5.21 Results of a four-factor PERMANOVA on community assemblage

Source	df	SS	MS	Pseudo-F	P(perm)
Season	1	13580	13580	4.8147	**0.0089****
IvR	1	4396.1	4396.1	1.2607	0.2721
Location(IvR)	4	24859	6214.6	1.7822	**0.0197***
SeasonxIvR	1	2954.9	2954.9	1.0477	0.4076
Site(Location(IvR))	5	17436	3487.1	1.3505	**0.0148***
SeasonxLocation(IvR)	4	17935	4483.8	1.5897	0.0557
SeasonxSite(Location(IvR))	5	14103	2820.5	1.0923	0.2672
Res	44	1.14E+05	2582.1		
Total	65	2.09E+05			

Notes:

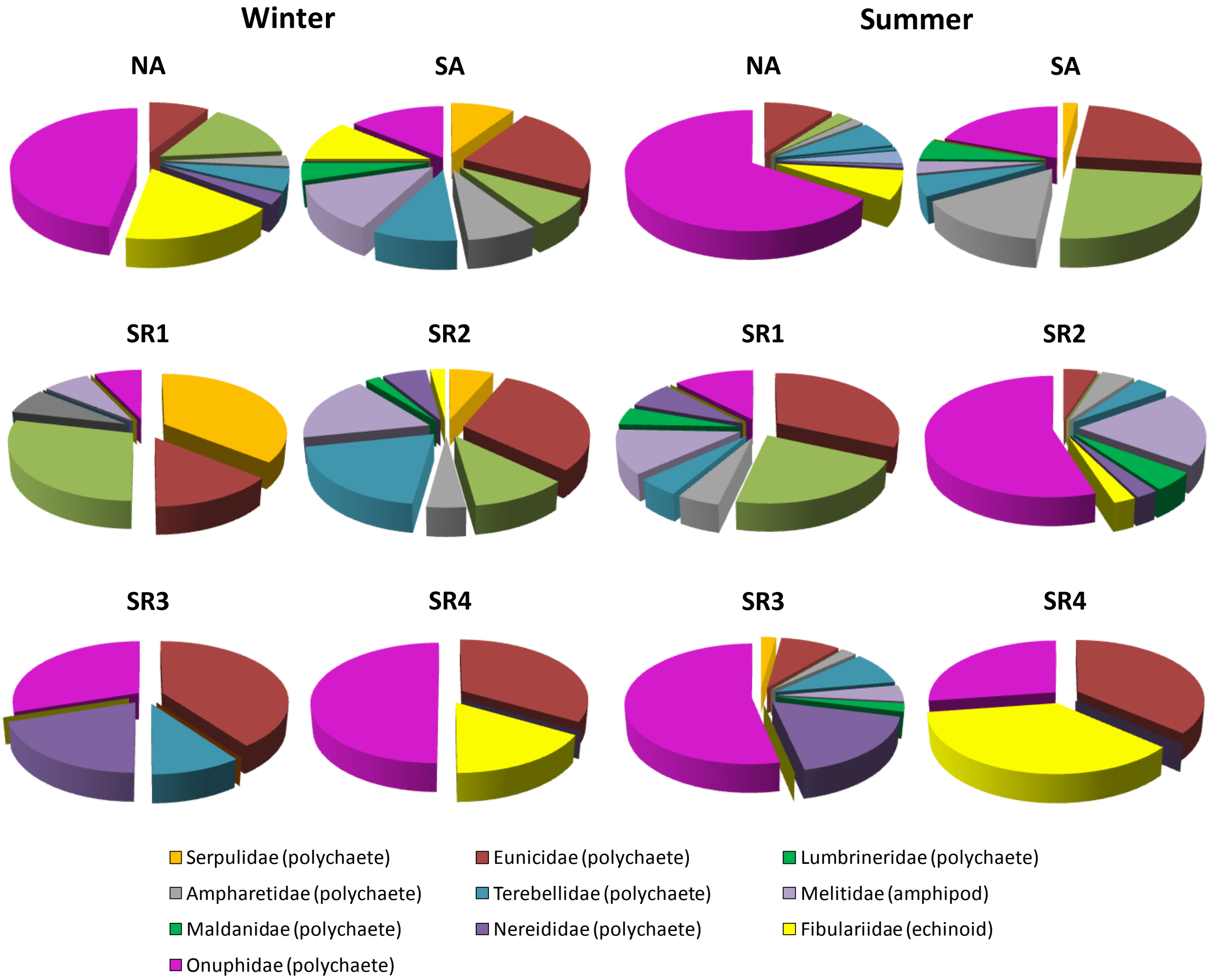
1. *Significant = p<0.05; **Highly significant - p<0.01
2. Significant results shown in bold



Note:

1. NA (northern area); SA (southern area); SR (sediment reference)
2. MDS (multi-dimensional scaling ordination)

Figure 5-28 MDS ordination of community assemblage among locations with vector overlays



Note:

1. NA (northern area); SA (southern area); SR (sediment reference)

Figure 5-29 Percentage representation of the top ten most abundant infauna families

Family richness

Univariate tests revealed significant differences in family richness among Locations(ZvR) and seasons (Table 5.22). In general, higher family richness was observed in summer (17.9 ± 1.3 richness) than in winter (10.1 ± 1.0 richness; Figure 5-30). The southern area reported higher family richness (15.9 ± 2.1 richness) relative to the northern area (11.5 ± 1.2 richness).

Table 5.22 Results of a four-factor PERMANOVA on family richness

Source	df	SS	MS	Pseudo-F	P(perm)
Season	1	913.69	913.69	16.8920	**0.0081****
IvR	1	28.082	28.082	1.8570	0.2116
Location(IvR)	4	458.09	114.52	7.5730	**0.0160***
SeasonxIvR	1	4.0029	4.0029	7.40E-02	0.7919
Site(Location(IvR))	5	75.611	15.122	0.31209	0.9072
SeasonxLocation(IvR)	4	261.3	65.325	1.2077	0.4033
SeasonxSite(Location(IvR))	5	270.44	54.089	1.1163	0.3594
Res	44	2132	48.455		
Total	65	4251.8			

Notes:

1. *Significant = $p<0.05$; **Highly significant = $p<0.01$
2. Significant results shown in bold

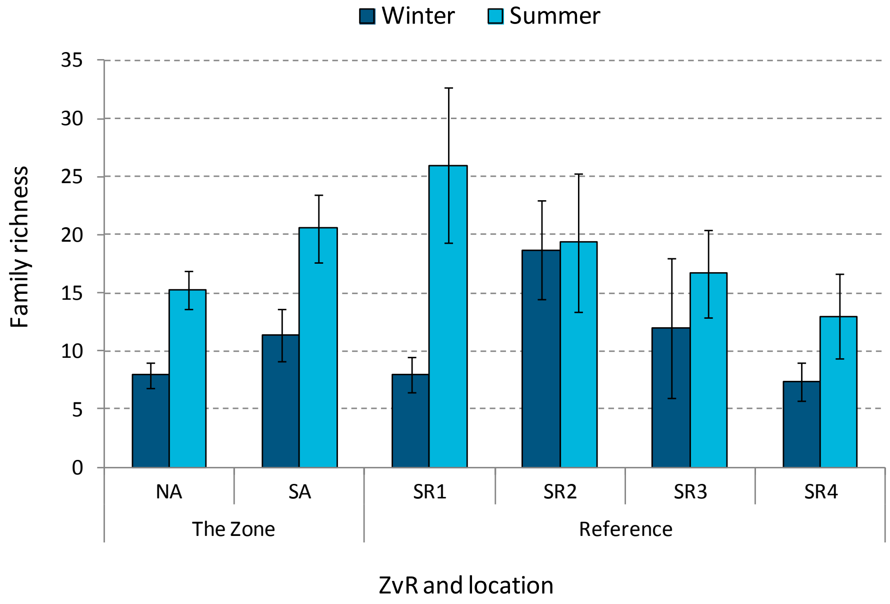


Figure 5-30 **Family richness (mean ± SE) of benthic infauna across seasons and locations within ZvR**

Family abundance

The four-factor design revealed a significant seasonal effect for family abundance (Table 5.23). Family abundance was greater in summer across all locations (35.39 ± 3.27 individual animals) compared to winter (16.09 ± 2.33 individual animals; Figure 5-31).

Table 5.23 **Results of a four-factor PERMANOVA on family abundance**

Source	df	SS	MS	Pseudo-F	P(perm)
Season	1	5156.7	5156.7	24.833	**0.0046****
IvR	1	65.789	65.789	0.3290	0.5794
Location(IvR)	4	2067.8	516.94	2.5851	0.1451
SeasonxIvR	1	138.96	138.96	0.6692	0.4514
Site(Location(IvR))	5	999.83	199.97	0.73962	0.5970
SeasonxLocation(IvR)	4	751.8	187.95	0.9051	0.5217
SeasonxSite(Location(IvR))	5	1038.3	207.66	0.7681	0.5735
Res	44	11896	270.36		
Total	65	23145			

Notes:

1. **Highly significant = p<0.01
2. Significant results shown in bold

Relationship between benthic assemblage and sediment parameters

Vector overlays of the sediment parameters onto the infauna CAP ordination plot showed that the infauna assemblage at the northern lease area (SL1) and reference location SR4, which include higher counts of polychaetes, amphipods, echinoids, Nereididae and Onuphidae (see text on 'Community assemblage', above), reside in fine to coarse sediments (62–>2000 μm) (Figure 5-28). Polychaetes and amphipods, which were found in greater abundance at the southern lease area (SL2) and reference location SR1 (see text on 'Community assemblage', above), inhabited sediments containing higher TOC content, phosphorus, aluminium and chromium levels (Figure 5-32).

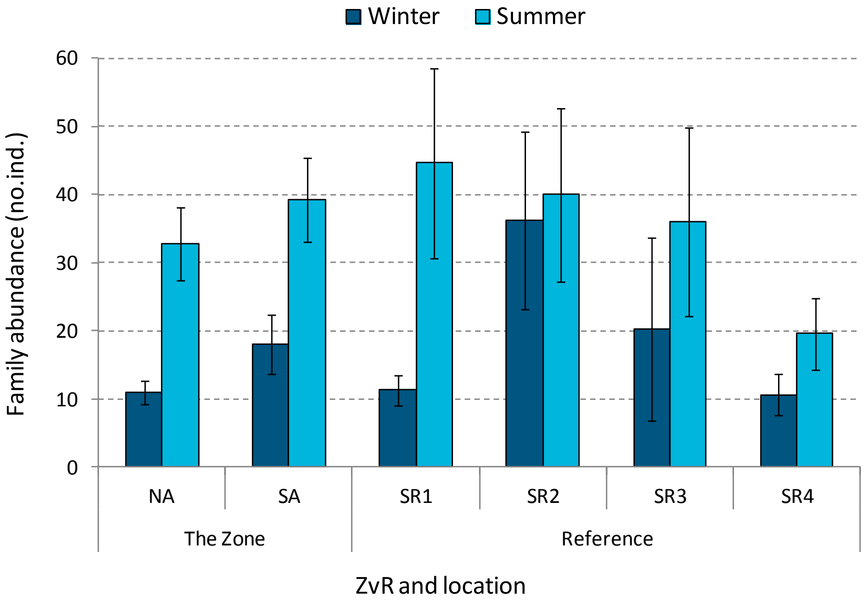
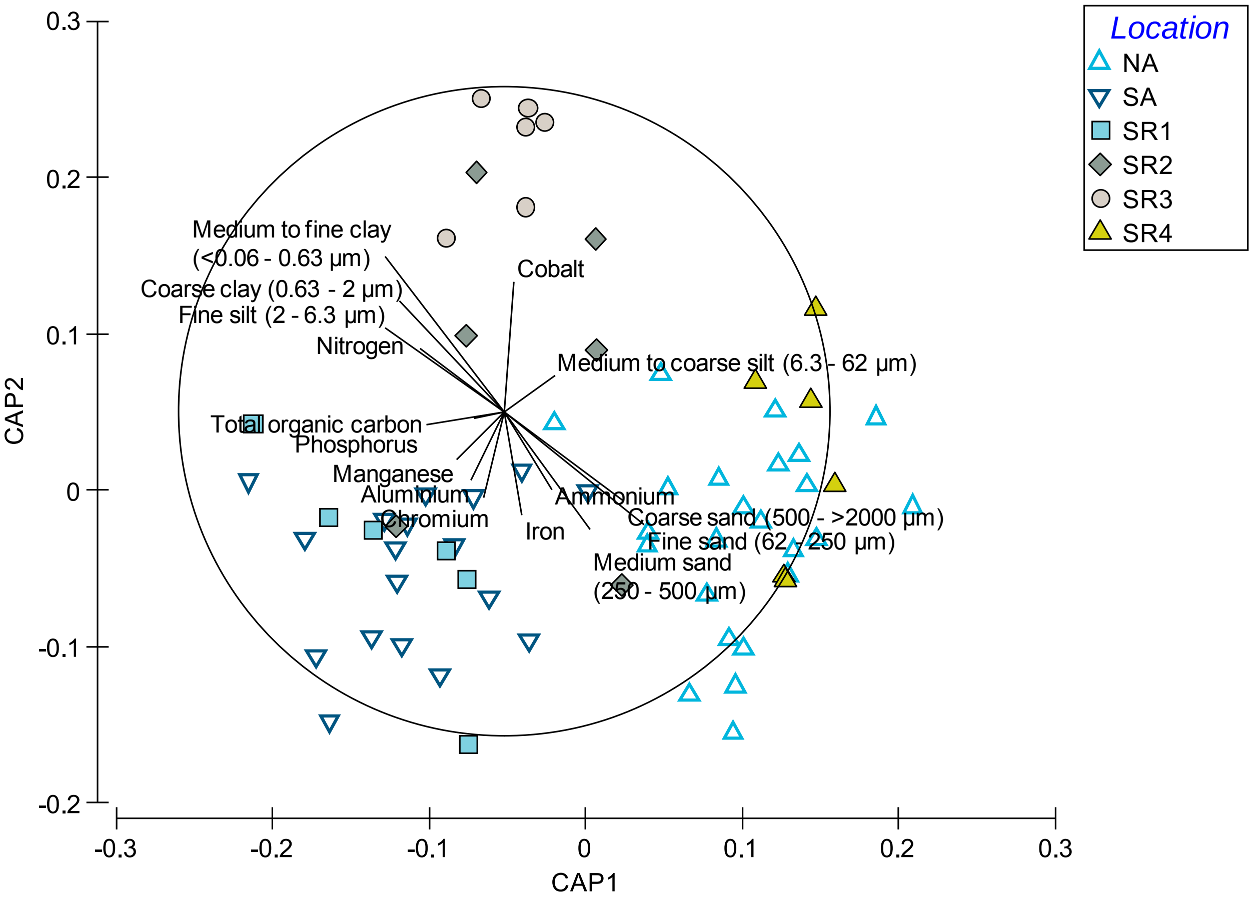


Figure 5-31 **Family abundance (mean ± SE) of benthic infauna across seasons and locations**



Note:

1. NA (northern area); SA (southern area); SR (sediment reference)

Figure 5-32 **CAP ordination plot of the benthic assemblage among locations with vector overlays of sediment parameters**

5.4.5 Total Petroleum Hydrocarbons / Polycyclic Aromatic Hydrocarbons

Total petroleum hydrocarbons (TPH) and polycyclic aromatic hydrocarbons (PAH) in marine sediments were generally below the laboratory limit of reporting (LOR). Of the 188 replicate sediment samples collected, 16 samples exceeded the LOR for PAHs (0.001 µg/L), 1 sample exceeded the LOR (100 mg/kg) for TPH C16-C34 and 1 sample exceeded the LOR (200 mg/kg) for total TPH (Table 5.24).

Table 5.24 Total Petroleum Hydrocarbons and Polycyclic Aromatic Hydrocarbons concentrations in sediments

Chemical	Species	LOR (mg/kg)	Site and value (mg/kg)
TPH	C16-34	100	SA winter (110)
	Total	200	SA winter (200)
PAH	Anthracene	0.001	SR4 summer (0.002)
	Fluoranthene		SA summer (0.002) SR4 summer (0.044)
	Fluorene		SR4 summer (0.006)
	Naphthalene		NA summer (0.002) SA summer 2 reps (0.001) SR1 summer (0.001) SR2 summer 2 reps (0.001 and 0.002) SR3 summer 2 reps (0.002) SR4 summer 3 reps (0.001 and 0.006)
	Phenanthrene		SA summer (0.007) SA winter (0.001) SR4 summer (0.078)
	Pyrene		SA winter 3 reps (0.001 – 0.002) Sa summer (0.002) SR4 summer (0.033)

5.5 Benthic habitats

5.5.1 Northern area

Surveys of the MWADZ study area indicated that much of the seafloor consisted of rocky pavement overlain with sand, with sparsely distributed biological assemblages. This contributed to a mosaic of habitats consisting of sandy meadows and areas of mixed assemblages, comprising filter feeders (sponges, and bryozoans), macroalgae, rhodoliths and hard corals (though the latter was observed infrequently). Because interpolation was used to spatially determine the major habitat categories, some parts of the study area could not be mapped with adequate certainly. These are shown in Figure 5-33 as white coloured pixels.

Habitats in the northern area consisted mainly of bare sand (59%) and mixed assemblages (34%; Figure 5-33). Small patches of reef were present near the north-east boundary but made up only 8% of the identified habitats within the area. The mixed assemblage habitats were mainly composed of macroalgae, rhodolith and sponges with a distribution of 3.7%, 3.3% and 2.3% of the total northern lease area respectively, with the remainder consisting of sand. Examples of the most commonly observed habitats are presented in Figure 5-34.

5.5.2 Southern area

Habitats in the southern area were predominantly bare sand (96%; Figure 5-33) with sparse mixed assemblages (5%) close to the Island. Of the mixed assemblages, rhodoliths and unknown organisms comprised 0.3% and 0.1% of the total southern lease area, respectively, with the remainder consisting of sand. Reef areas in the southern lease were dominated by rohodolith communities, with no evidence of significant hard coral cover.

5.5.3 Reference sites

The habitats of the three reference sites (with the exception of the northern-most reference site) were dominated by bare sand (42.5%) followed by mixed assemblage categories on sand and reef (total 17.7%; Figure 6.24). The northern reference site had a more diverse distribution of habitats throughout the area with reef and mixed assemblages/reef habitats present (12.4%; Figure 5-33). The main biotic constituents of the mixed assemblage habitats were macroalgae, sponges and hard coral with a distribution of 2.1%, 1.3% and 0.1% of the total reference site area, respectively.

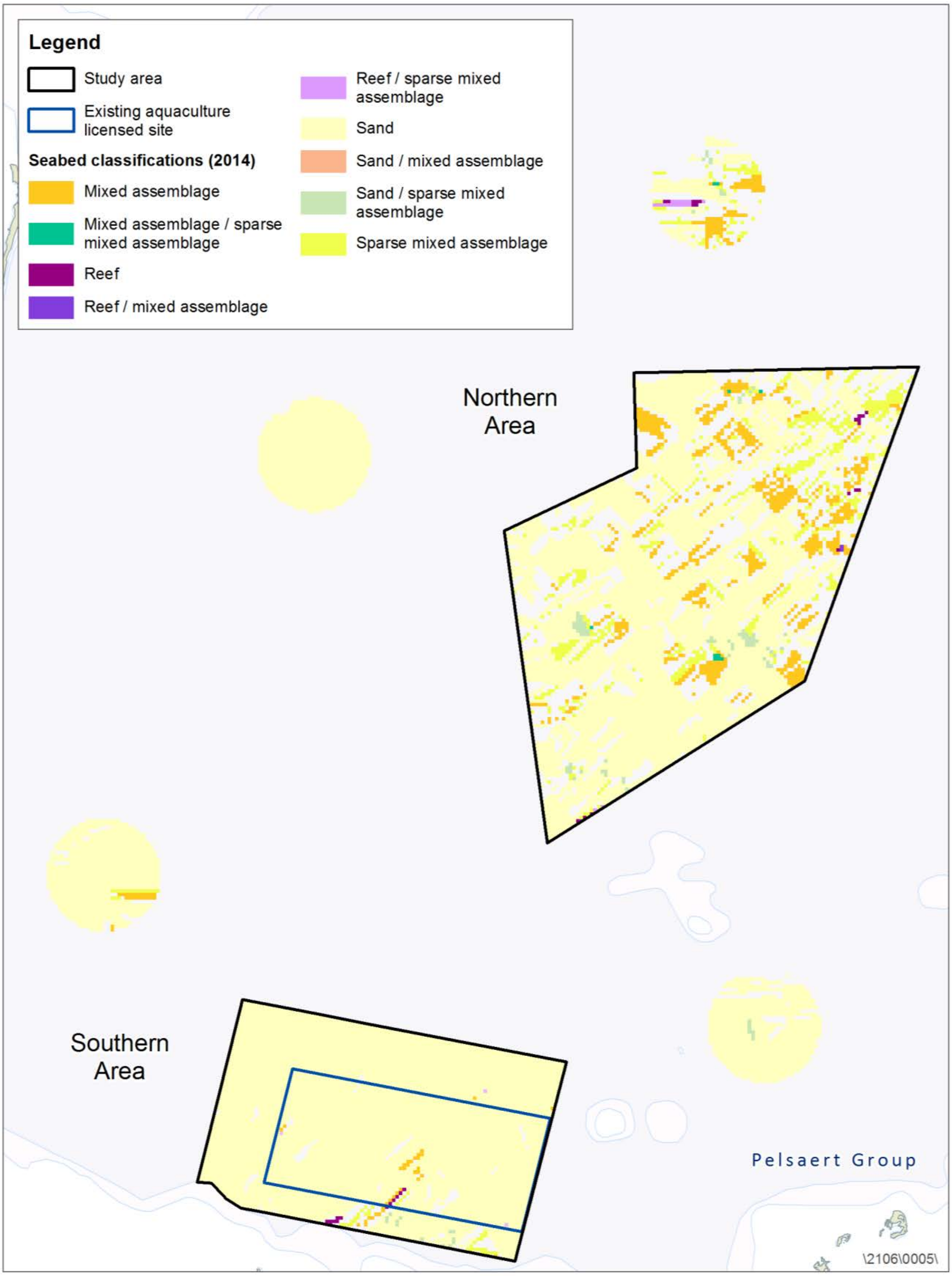
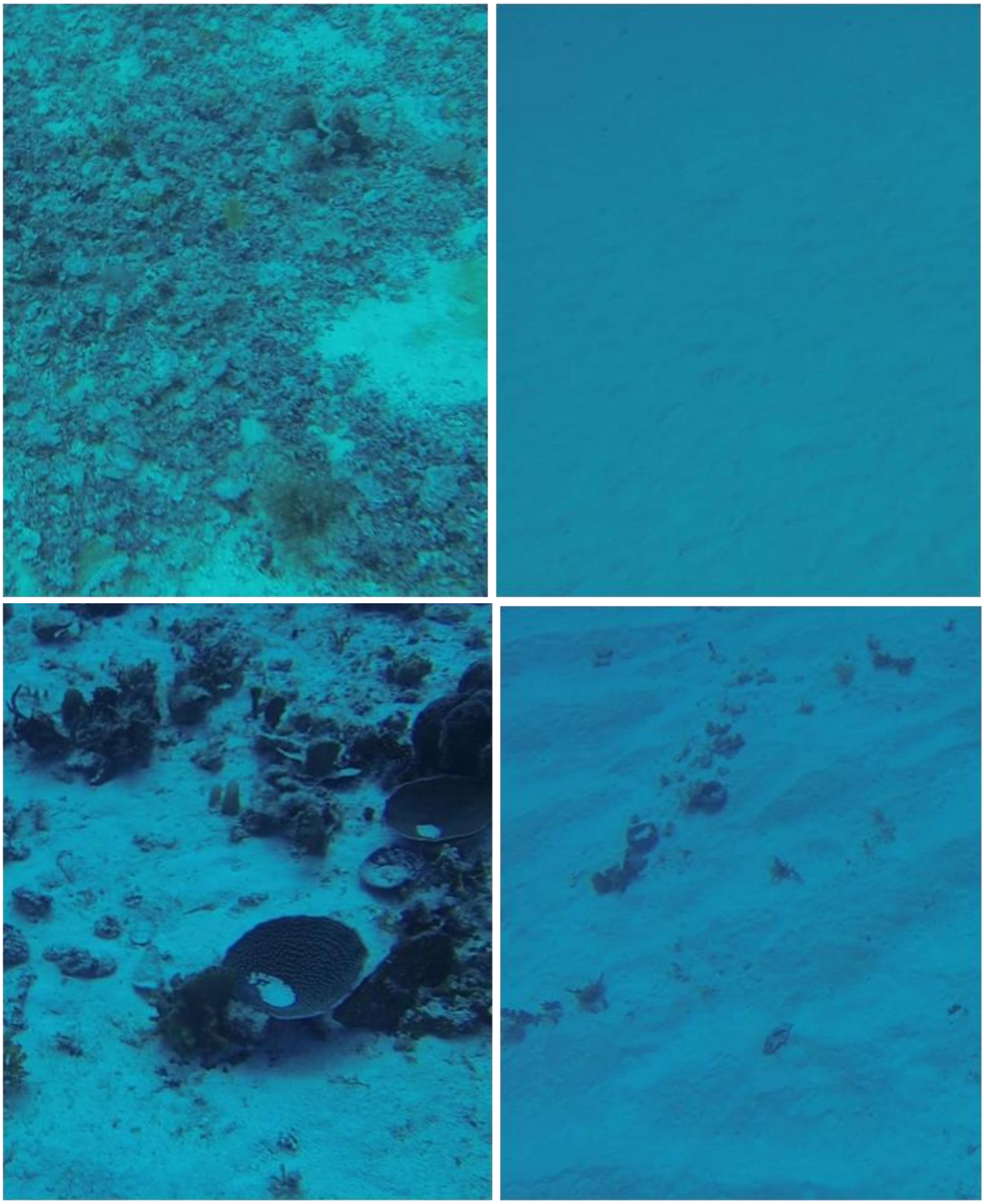


Figure 5-33 Major habitat assemblages observed in the study area in 2014



Notes:

1. Mixed assemblages with hydroids and macroalgae (top left); Mixed assemblages with rhodoliths (top right); mixed assemblages with sponges and macroalgae (lower left) and sparse mixed assemblages (lower right)

Figure 5-34 Examples of the common habitats observed during benthic habitat surveys

5.5.4 Agreement with previous surveys

Comparisons between the surveys are made at a high level, and results are provided here for contextual purposes only. The historical 2003, 2006/2008 and 2014 surveys differed significantly in their approaches, in terms of equipment and the classification schemes used. Changes may have occurred between surveys as a result of the dynamic nature of the seabed within the project area and is indicative the effects of sand sheet movement and variability over time.

Historical surveys (Section 4.3.1) identified a range of habitats present in the northern and southern lease areas (Figure 5-35, Figure 5-36) that were not consistently identified in 2014. For example, although the 2006 survey only captured a fraction of the proposed northern MWADZ, it identified larger proportions of mixed assemblage than the 2014 survey. The 2014 survey indicated a change to a sand dominated habitat with a noticeable reduction of mixed assemblages and reef habitats.

Similarly, previous surveys of the southern MWADZ identified significant areas of rhodolith, reef and sand with areas of *Halophila* spp., algae and mixed assemblages. A shift to a sand dominated habitat with a reduction of biotic organisms (<1%; Figure 5-36) was observed in 2014. No seagrass was observed within the southern lease area in 2014.

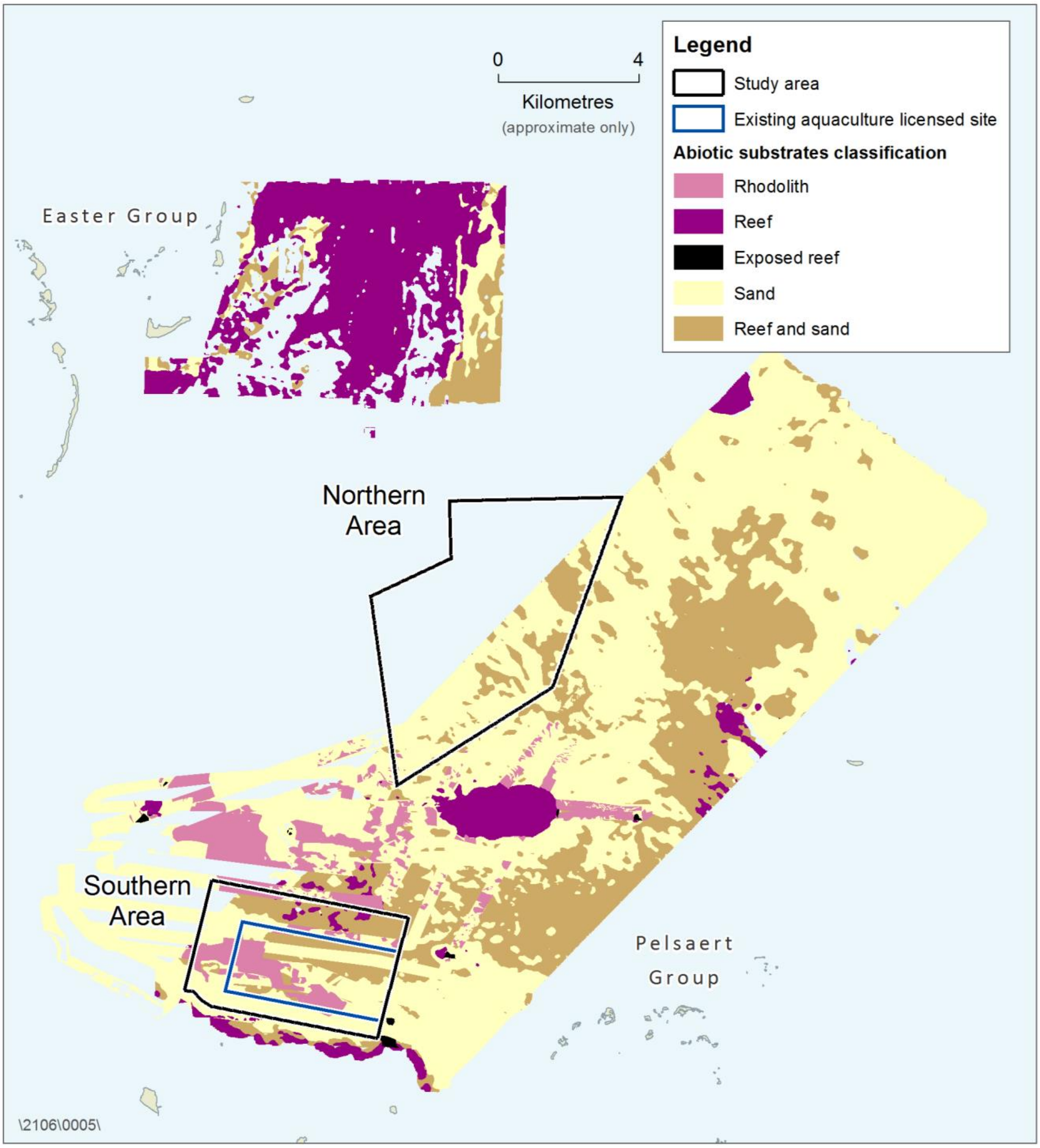


Figure 5-35 Major abiotic habitat assemblages observed in 2003, 2006 and 2008

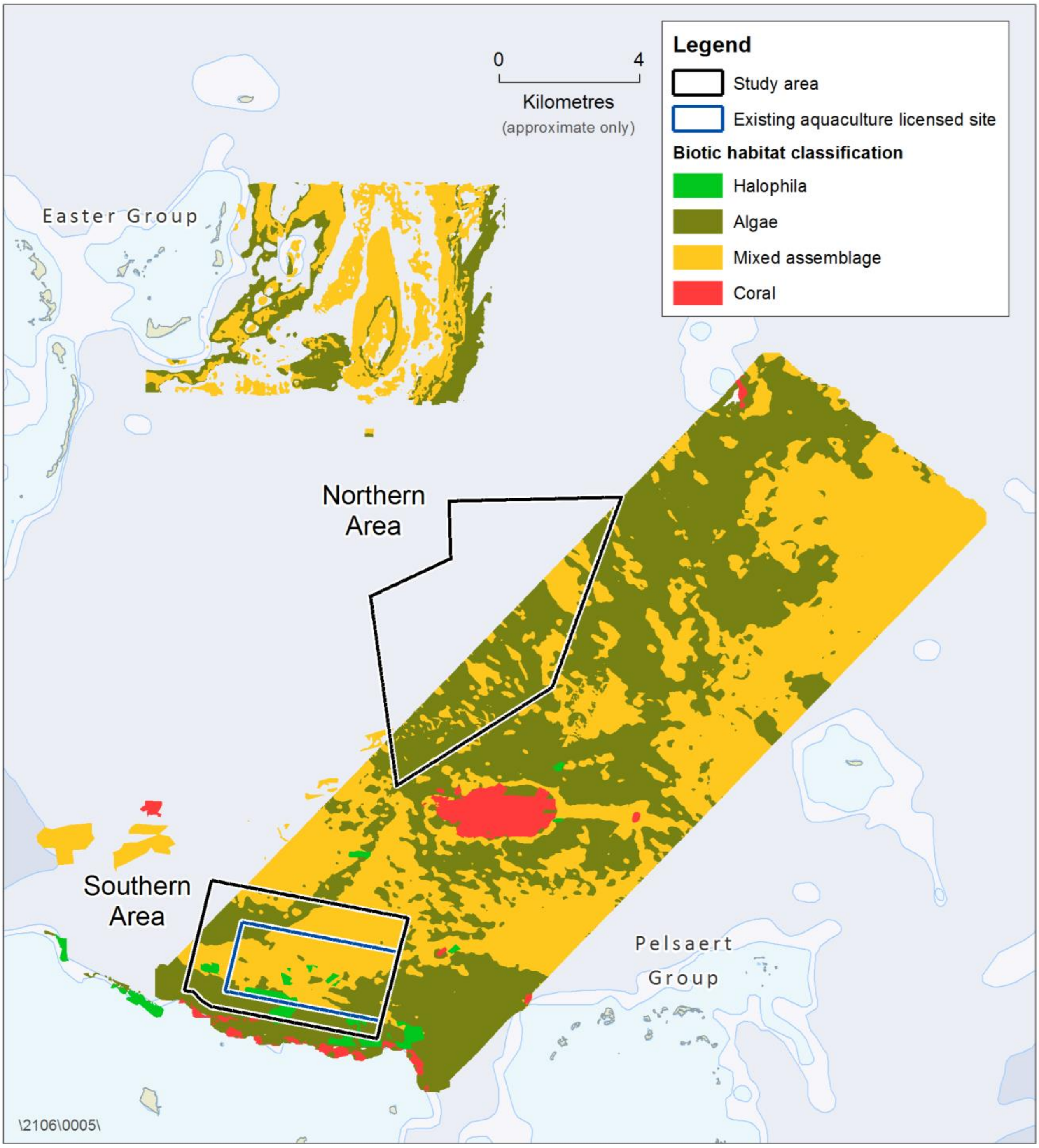


Figure 5-36 Major biotic habitat assemblages observed in 2003, 2006 and 2008

6. Impact Assessment - Cumulative loss of BPPH

6.1 Development of the local assessment unit

The LAU for this assessment was developed by DoF in consultation with the OEPA. The first point was to consider the extent of previous losses of BPPH, such as those which may have been lost due to historical anthropogenic activities. It was considered that benthic habitats in the MWADZ are relatively pristine, and that any effects of historical anthropogenic activities were transient, and now fully recovered.

EAG 3 requires that the expected cumulative losses of BPPHs are assessed as a proportion against those in an agreed Local Assessment Unit (LAU). In consultation with the EPA, DoF used relevant data to define two local assessment units (LAU) within a one kilometre buffer around the Northern and Southern Areas of the proposed zone (Figure 6-1). In relation to benthic habitat, most (71%) of the Northern LAU (44.2 km^2) and nearly all (96%) of the Southern LAU (23.2 kilometre squared) has been surveyed. The benthic layers in the attached map are primarily based on a hydro-acoustic survey of the study site for the MWADZ proposal undertaken by the Department of Fisheries Marine Ecosystem Monitoring Section. This survey was conducted in 2014, using a single beam echo sounder and a drop video for ground-truthing (here on referred to as the DoF 2014 survey).

To gain an understanding of the dynamics of the BPPH in and around the strategic proposal areas, and interpolate/extrapolate the coverage of BPPH to include a 1 km strip outside the proposed MWADZ, two other habitat surveys were taken into account:

1. The University of Western Australia Marine Futures Project - hydro-acoustic mapping, towed video and biodiversity sampling in and around the Southern Group of Abrolhos Islands, 2006 and 2008 (here on referred to as Marine Futures 2006 survey).
2. The University of Western Australia and Undersea Community Pty Ltd Habitat Survey North of the Pelsaert Group of the Abrolhos Islands, by Andy Bickers in 2003. This survey (here on referred to as Bickers 2003 survey) used side-scan sonar.

Each of the three surveys provided discrete, low-resolution assessments and used different technical approaches. The surveys served to provide an indicative description of the benthic substrates in the vicinity of the MWADZ at the times they were conducted. Interpolation of the one kilometre strips surrounding the proposed MWADZ is primarily based on the Marine Futures 2006 survey. The Bickers 2003 survey data was used to describe the small portions of the LAUs that were not covered by the other surveys. The data used to describe both the Northern and Southern LAUs consists of 67% DoF 2014 survey data, 31% Marine Futures 2006 survey data, and two percent Bickers 2003 survey data.

Collectively, all of the available data from the three surveys suggest that the benthic environment within the Northern and Southern LAUs are continually changing due to sand sheet movement and corresponding natural variability of the benthic habitat coverage. The data was used to estimate the most likely coverage of Mixed Assemblages, Reef and Bare Sand in the LAUs. For the purposes of this assessment, Mixed Assemblages and Reef have been conservatively assumed to correspond to habitats capable of supporting BPPH.

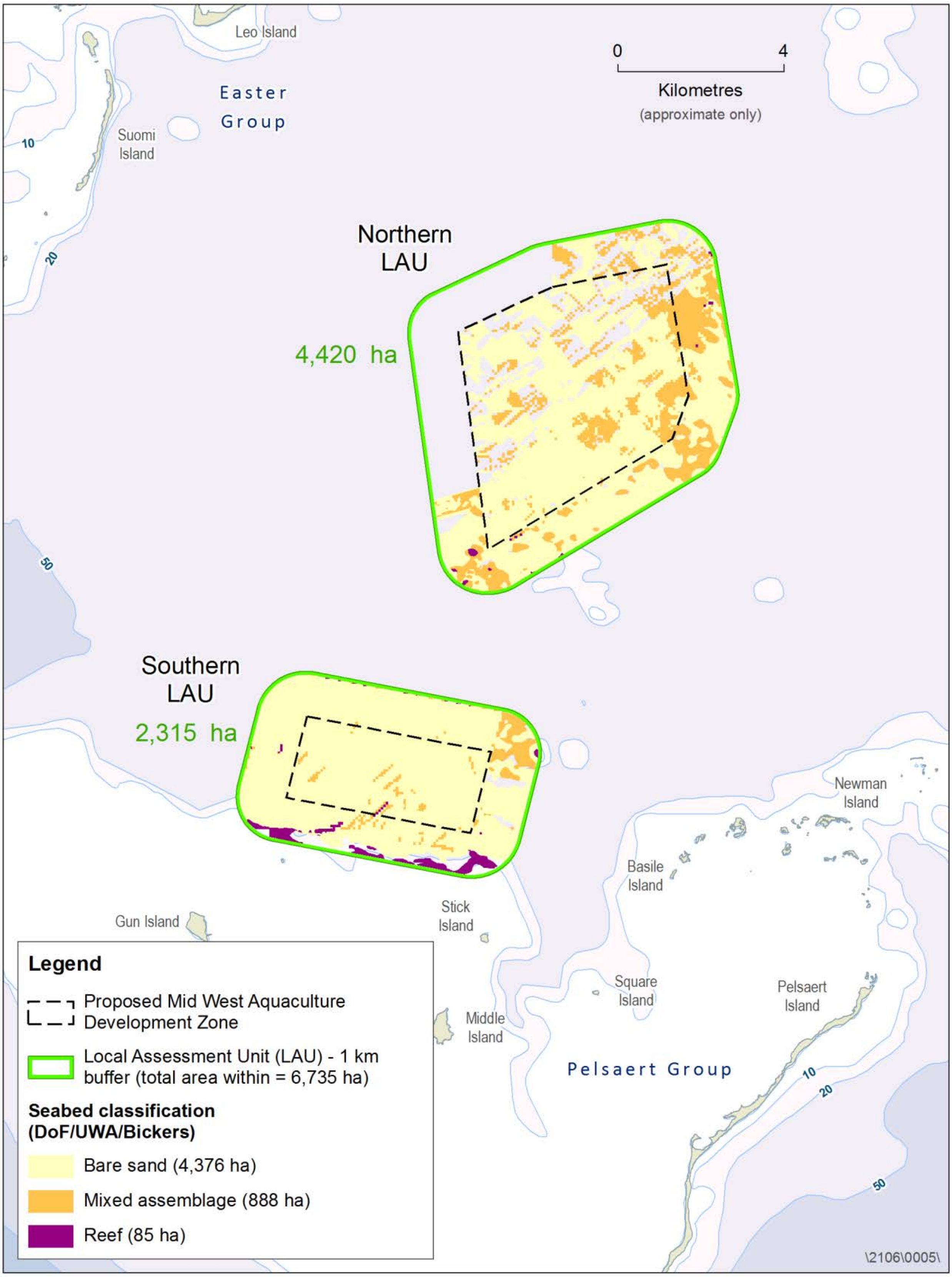


Figure 6-1 The Northern and Southern Local Assessment Units and the indicative benthic substrates in the vicinity of the MWADZ

6.2 Estimating the benthic cover of BPPHs

6.2.1 Northern LAU

Habitat surveys in Northern LAU adequately captured the diversity and natural variability of the environment (i.e. bathymetry and proximity to islands) within a one kilometre buffer around the Northern Area of the MWADZ. Although approximately 29% of the Northern LAU has not been surveyed in relation to benthic habitat, this portion was extrapolated for the purposes of this assessment.

The existing data suggests at least 24% of the Northern LAU supports mixed assemblages consisting of algae and sessile invertebrates (Table 6.1). The benthic substrate classified as reef (medium relief) is the only substrate capable of sustaining coral reef habitat and makes up less than one percent of the Northern LAU. The benthic substrate classified as bare sand makes up approximately 75% of the Northern LAU. The DoF ground-truthing studies indicate that this substrate is predominantly bare sand overlying platform limestone reef (to a depth ~15 cm).

Of the 4420 hectares in the Northern LAU, approximately 25% of this area (1091 hectare) comprises habitats capable of supporting BPPH (i.e. around 0.29% reef and 24% mixed assemblages, while approximately 75% is bare sand) (Table 6.1).

Table 6.1 Calculation used to estimate and extrapolate BPPH cover within the Northern LAU

Habitat Type	Relative contributions (ha)			Calculations to estimate coverage based on: Total area surveyed in Northern LAU (3133 ha) Area of Northern LAU (4420 ha)
	DoF survey 2014	Marine Futures 2006	Bickers survey 2003	
Reef	3	6	0	Sum (9 ha) div. by 3133 x 100 = 0.29% 0.29% div. by 100 x area of Northern LAU = 12.7 ha
Mixed Assemblage	427	312	25	Sum (764 ha) div. by 3133 x 100 = 24.4% 24.4% div.by 100 x area of Northern LAU = 1078 ha
Bare Sand	1476	837	47	Sum (2360 ha) div. by 3133 x 100 = 75.3% 75.3% div. by 100 x area of Northern LAU = 3329 ha

6.2.2 Southern LAU

Data compiled from both recent and historical habitat surveys were used to determine the diversity and variability of the benthic environment in the Southern Area of the MWADZ. Surveys covered habitats out to a distance of 1 km from the zone boundaries. Although 4% of the Southern LAU has not been mapped, the remaining habitats were extrapolated for the purposes of this assessment.

The existing data suggests approximately 6% of the Southern LAU supports mixed assemblages consisting of algae, rhodolith and sessile invertebrates. The benthic substrate classified as reef (medium relief) is the only substrate capable of sustaining coral reef habitat and makes up less than four percent of the Southern LAU. The benthic substrate classified as bare sand makes up approximately 91% of the Southern LAU. The DoF ground truthing studies indicate that this substrate is predominantly bare sand overlying limestone platform reef (to ~15 cm depth).

Of the 2315 hectares in the Southern LAU, approximately 9% (208 hectares) of the Southern LAU comprises habitats capable of supporting BPPH (3.4% Reef and 5.6% mixed assemblages, while approximately 91% is bare sand).

Table 6.2 Calculation used to estimate and extrapolate BPPH cover within the Southern LAU

Habitat Type	Relative contributions (ha)			Calculations to estimate coverage based on: Total area surveyed in the Southern LAU (2217 ha) Area of Southern LAU (2315 ha)
	DoF survey 2014	Marine Futures 2006	Bickers survey 2003	
Reef	4	62	10	Sum (76 ha) div. by 2217 x 100 = 3.4% 3.4 div. by 100 x area of Southern LAU = 79.4 ha
Mixed Assemblage	29	95	1	Sum (125 ha) div. by 2217 x 100 = 5.6% 5.6 div. by 100 x area of Southern LAU = 130.5 ha
Bare Sand	1621	354	41	Sum (2016 ha) div. by 2217 x 100 = 90.9% 90.9 div. by 100 x area of Southern LAU = 2105.1 ha

6.3 Estimated losses of BPPH

6.3.1 Northern LAU

Approximately 25% of the Northern LAU (1091 hectares) comprises habitats capable of supporting BPPH. Under S4 (24 000 t), modelling predicted that the ZoHI in the Northern LAU would occupy 41 ha after three years production[3] (Section 7.3.2). This figure was doubled to allow for recovery sites generated by fallowing the aquaculture sites.

Table 6.3 Calculation used to estimate the loss of BPPH within the Northern LAU

Average area of BPPH (ha) within the Northern LAU under ZoHI	Estimated % loss of BPPH within the Northern LAU
Area of BPPH inside the Northern Area of the Zone 269 ha *Percentage of BPPH within the Zone* 269 ha divided by the Northern Area of the Zone (2200 ha) x 100 = 12.3% *ZoHI within the Zone* ZoHI (41 ha) x 2 (recovery sites) = 82 ha *Area of BPPH effected by the ZoHI* (12.3 % divided by 100) x 82 ha = 10.1 ha	Estimated % loss of BPPH 10.1 ha divided by area of BPPH in the Northern LAU (1091 ha) x 100 = 0.93%

6.3.2 Southern LAU

Approximately nine percent (209.9 hectares) of the Southern LAU comprises habitats capable of supporting BPPH. Under S4 (24 000 t), modelling predicted that the ZoHI in the Southern LAU would occupy 21 ha after three years production. This figure was doubled to allow for recovery sites generated by fallowing the aquaculture sites.

---

[3] *Note that the figures shown for the area occupied by the ZoHI in Section 7.3.2 are for the combined northern and southern areas.*

Table 6.4 Calculation used to estimate the loss of BPPH within the Southern LAU

Average area of BPPH (ha) within the Southern LAU under ZoHI	Estimated % loss of BPPH within the Southern LAU
Area of BPPH inside the Southern Area of the Zone 279.1 ha *Percentage of BPPH within the Zone* 10.6 ha divided by the Southern Area of the Zone (800 ha) x 100 = 1.33% *ZoHI within the Zone* ZoHI (21 ha) x 2 (recovery sites) = 42 ha *Area of BPPH effected by the ZoHI* (1.33% divided by 100) x 42 ha = 0.56 ha	*Estimated % loss of BPPH* 0.56 ha divided by area of BPPH in the Southern LAU (209.1 ha) x 100 = 0.27%

6.4 Conclusion

The proposed MWADZ is within the FHPA. The Management Plan for the FHPA does not identify any areas of high conservation value that would be category A, and there have been no historical irreversible losses of BPPH in the LAU. Based on this, the assessment against EAG 3 was undertaken using the Category C cumulative loss guidelines (Table 4.8).

The Cumulative Loss Guidelines (EAG 3) recommend that cumulative losses of BPPH within Category C areas should not exceed 2% of the BPPH within the LAU. The cumulative loss of BPPH likely to result from the proposed aquaculture in the Northern LAU and Southern LAU was estimated at <1%, which is below the 2% benchmark.

7. Impact Assessment – Modelled

7.1 Overview

An integrated hydrodynamic, particle transport, water quality and sediment diagenesis model was used to simulate a total of six scenarios (S1–S6) as per the criteria detailed in Section 4.5.4 and Table 4.16. Sections 7.2 to 7.4 describe the predicted impacts of each of these scenarios on the marine environment, in terms of hydrology, sediments, benthic primary producing habitats and regional water quality. Results are described in the context of EAG 3 (EPA 2009) and EAG 7 (EPA 2011), which respectively describe the area of acceptable loss of BPPHs and the zones of impact, based on the criteria outlined in Table 4.9, Section 4.5.

7.2 Hydrodynamics

Sea-cages, or any other floating structures at sea, invariably impart some resistance to flows acting to slow or deflect waters in the vicinity of the cages. The potential for changes to the hydrodynamic regime in and around the proposed MWADZ sea-cages was investigated using the findings of Wu et al. (2014) and Cornejo et al. (2014).

Both Wu et al. (2014) and Cornejo et al. (2014) used numerical models and appropriate assumptions to determine the impact of cage clusters on the local current field. Cornejo et al. (2014) used a numerical model of an idealized environment to describe the changes to current dynamics and the formation of a wake arising from the introduction of sea-cages. They examined the impacts for various choices of mesh type for each cage, from high-drag materials (Cd=1.7) to low-drag materials (Cd=0.7).

Wu et al. (2014) derived a relationship between cage height, depth and an assumed friction parameter (Hasegawa et al. 2011) which can be used described impacts on the current field: $H=0.5H_0$, where H is the cage height and H_0 is depth. The assumed friction parameter used to derive this relationship was $\lambda=0.6$ per/m. The effect of MWADZ sea-cages on the surrounding hydrodynamic regime was extrapolated using the findings of Wu et al. (2014) together with the known characteristics of the MWADZ environment (12–50 m depth) and the proposed infrastructure (18 m depth cages).

Under high-drag scenarios and the ambient velocities observed in the proposed MWADZ (~0.1 m/s), bottom velocity is expected to increase by approximately 20% and surface velocity within the cages is expected to reduce by approximately 80%. Natural surface current velocities through the proposed MWADZs 8.7–14.1 cm/s in the summer months, and 10.5–14.5 cm/s in the winter months. Current velocities recorded at depth were somewhat lower than this at 5.8–11 cm/s and 6.1–11.5 cm/s in the summer and winter months, respectively (Table 4.5). Based on the findings of Wu et al. (2014) surface current speeds inside the sea-cages are expected to reduce to between 1.8–3.0 cm/s and currents speeds under the cages, to increase to between 6.9–13.8 cm/s.

While this analysis indicates a potential increase in velocity near the seabed of 20%, it is not expected that this will substantially affect the erosion of sediments under the aquaculture cages. Sediment erosion and deposition is driven by bottom shear stress, and the hydrodynamic model indicates that bottom shear stress is dominated by wave action rather than current velocities within the proposed lease areas.

Table 7.1 Current speeds through the MWADZ before and after the introduction of sea-cage infrastructure

	Summer		Winter	
	Surface	**Bottom**	**Surface**	**Bottom**
Before the introduction of sea-cages	8.7–14.1 cm/s	5.8–11.0 cm/s	10.5–14.5 cm/s	6.1–11.5 cm/s
After the introduction of sea-cages	1.8–2.8 cm/s	6.9–13.2 cm/s	2.1–3.0 cm/s	7.3–13.8 cm/s

7.3 Soft sediments

7.3.1 Inputs of organic waste (carbon)

An integrated hydrodynamic, particle transport, water quality and sediment diagenesis model was used to determine the trajectory, settlement and impacts of organic wastes leaving the sea-cages. For modelling purposes, inputs of organic waste to the seafloor were termed 'flux of organic matter', or rate of FOM mmol.C/m^2/yr. FOM was used as a proxy for organic enrichment, and as an indicator of potential secondary effects, including deoxygenation and accumulation of sulphides. FOM data are reported here for contextual purposes only. EAG 7 was applied with consideration to the potential secondary effects described in Section 7.3.2.

Figure 7-1–Figure 7-4 show the predicted rate of FOM to the seafloor under a range of scenarios (S1,S2, S5 and S6), and after twelve months of continuous finfish production. FOM increased with increasing standing biomass (FOM S5-S6 > FOM S1-S2) (Figure 7-1, Figure 7-2) and increasing stocking density (FOM S6>S5 and S2>S1) (Figure 7-3, Figure 7-4). FOM levels greater than background were detectable beneath and near to the sea-cages in each of the modelled scenarios–the highest FOM values beneath the sea-cages corresponded with the highest levels of standing biomass (FOM S5>S1 and FOM S6>S2). Accumulation of organic material occurred under each of the scenarios, and commenced rapidly following beginning of production; FOM beneath sea-cages was observed to build rapidly, even under biomasses much lower than those modelled here (<1000 t finfish per cluster) (Appendix G).

The highest FOM was concentrated immediately below the sea-cage clusters. The confinement of the majority of FOM to the area immediately beneath the sea-cages is indicated in the colour change from light blue to red between scenarios S2 (15 000 t) and S6 (30 000 t), representing a change in FOM from ~2 x 10^5 to 15 x 10^5 mmol.C/m^2/yr (Figure 7-4, Figure 7-3). Areas beyond the sea-cage clusters, by contrast, maintained similar levels of FOM, despite the modelled increases in standing biomass. These data are indicative of a highly concentrated effect, whereby the deposition of organic waste is centred on the area of seafloor immediately under the sea-cages.

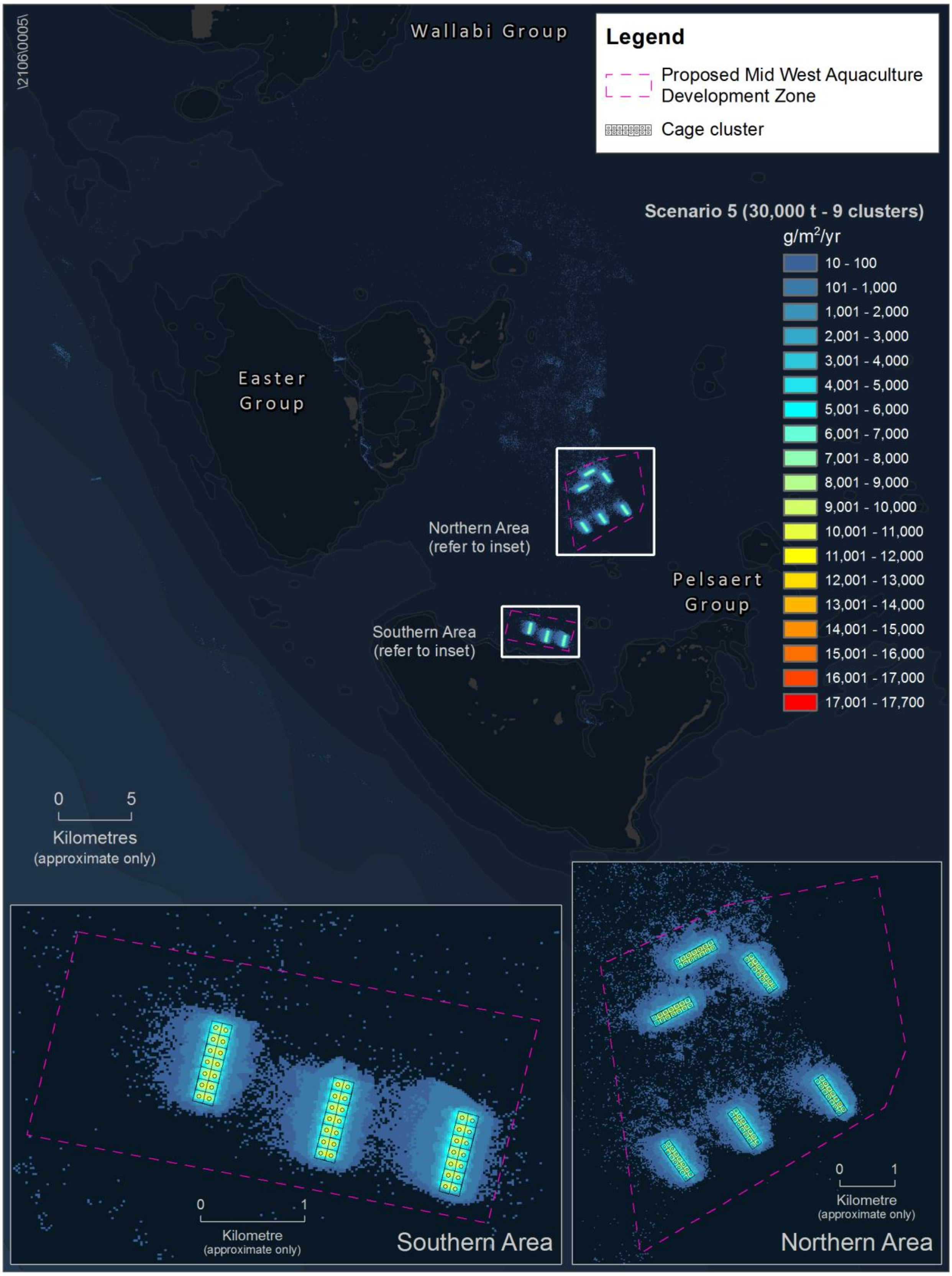


Figure 7-1 Inputs of organic carbon (FOM) under scenario 5 (30 000 t; 9 clusters)

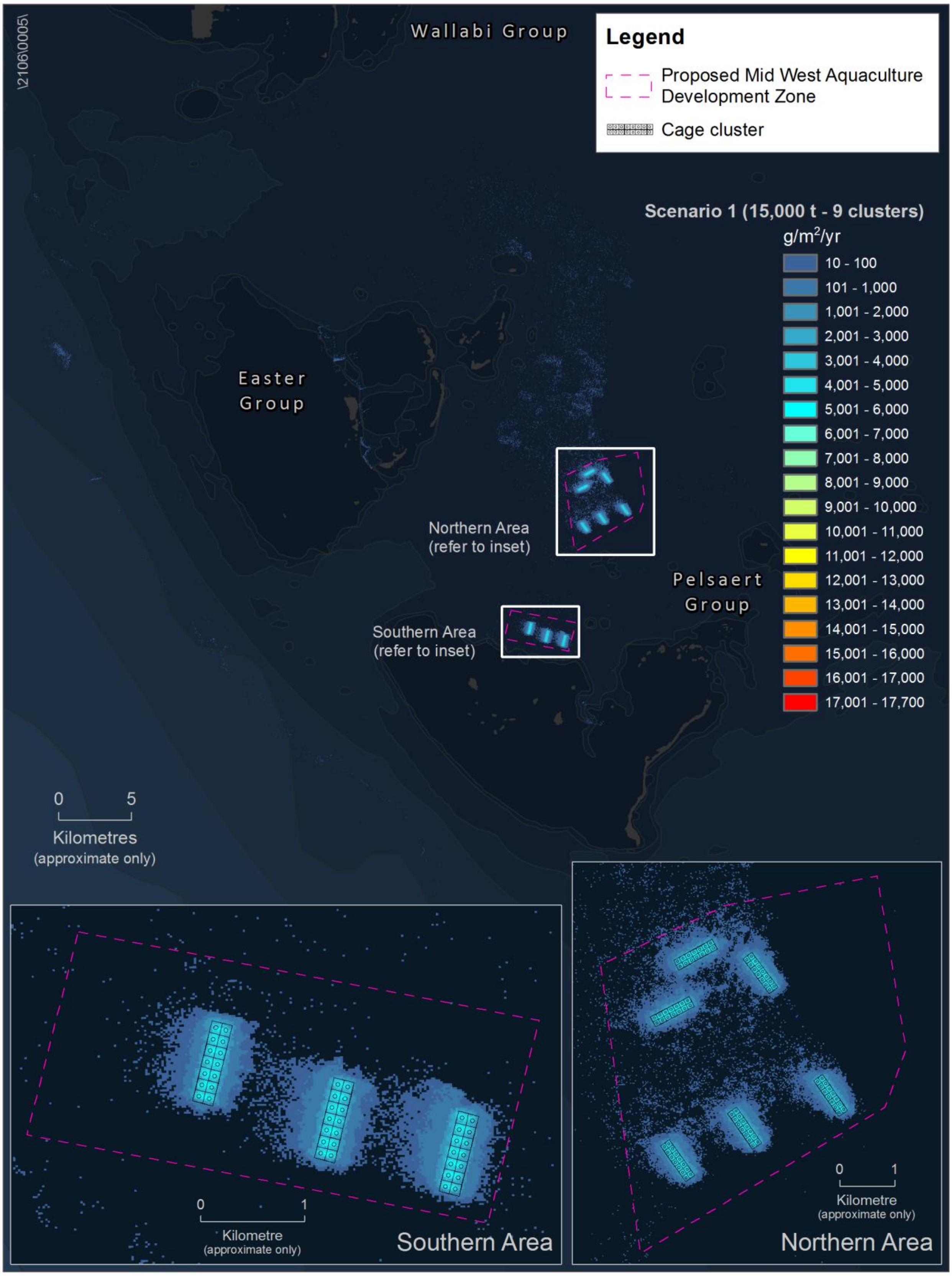


Figure 7-2 Inputs of organic carbon (FOM) under scenario 1 (15 000 t; 9 clusters)

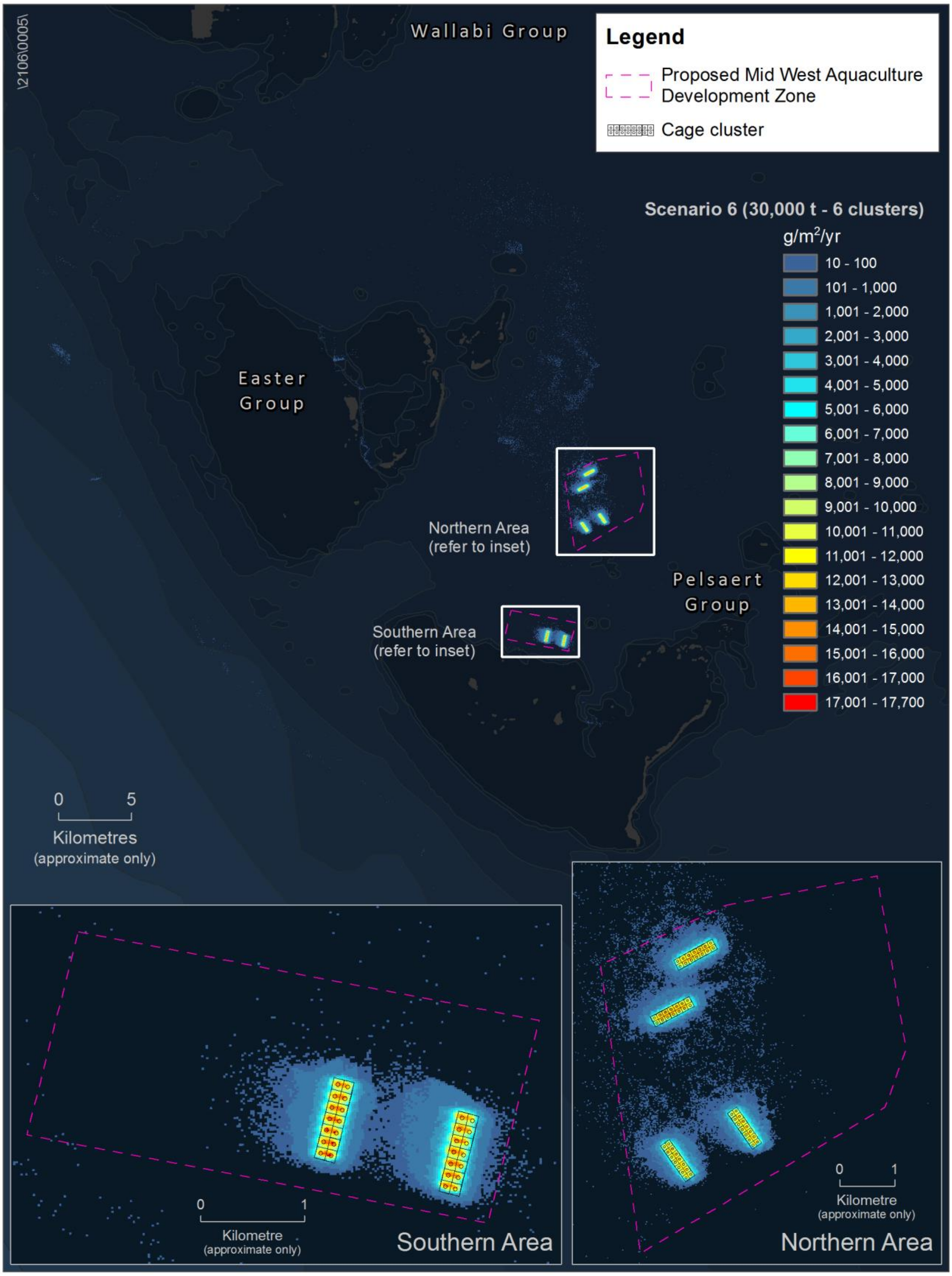


Figure 7-3 Inputs of organic carbon (FOM) under scenario 6 (30 000 t; 6 clusters)

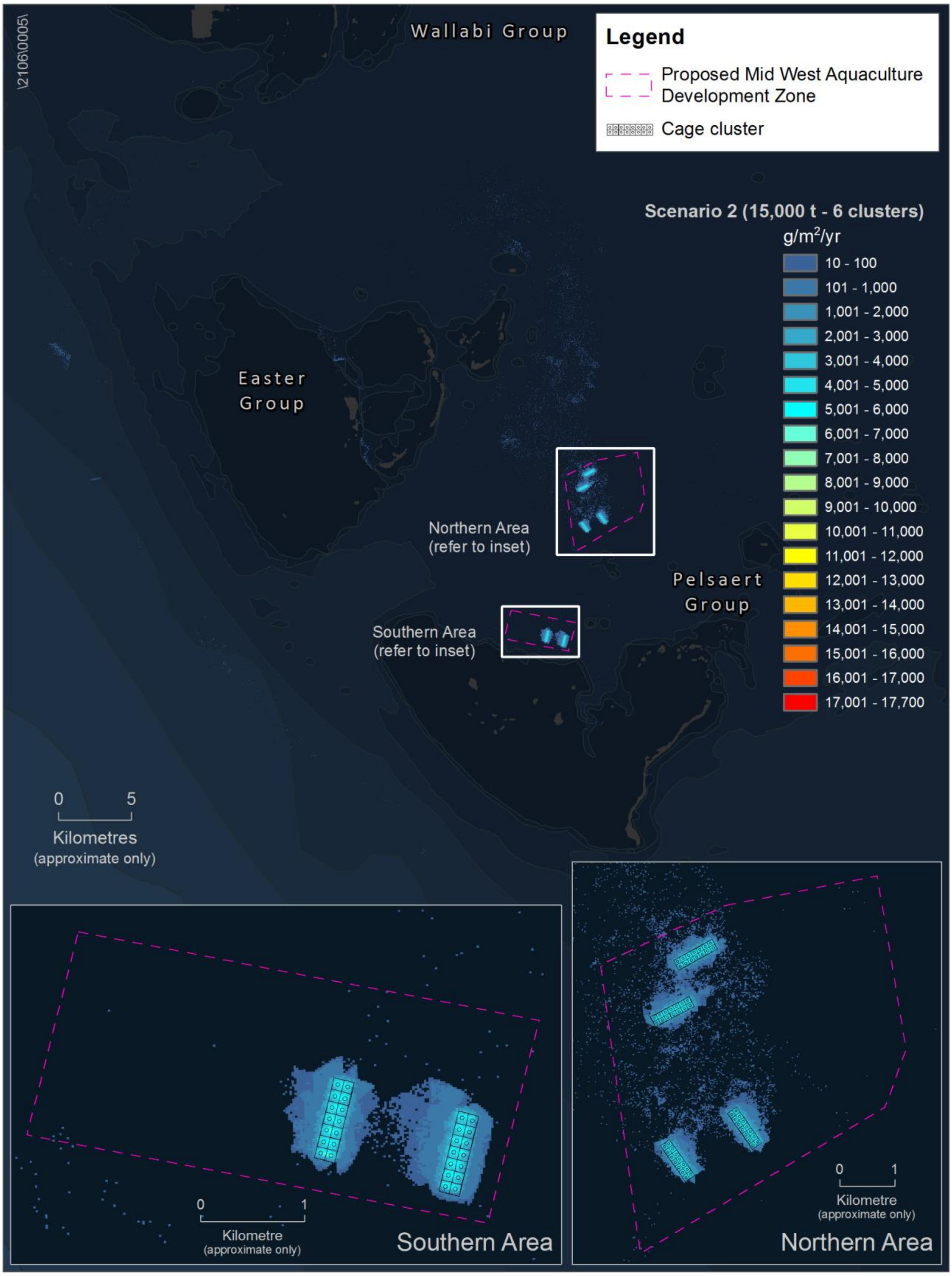


Figure 7-4 Inputs of organic carbon (FOM) under scenario 2 (15 000 t; 6 clusters)

7.3.2 Sediment dissolved oxygen & sulphide content

Figure 7-5–Figure 7-22 show the spatial extents of three zones of impact, following application of the criteria in EAG 7 (EPA 2011). The zones were defined based on the time required for sediment oxygen and sulphide concentrations to return to baseline levels, following two, three and five years of finfish production, and across the full range of production scenarios, 1 to 6 (S1-S6; Table 4.16). As per EAG 7, habitats requiring greater than five years to recover to baseline levels were designated zones of 'high' impact (ZoHI - red colouration), and habitats requiring less than five years were designated zones of 'moderate' impact (ZoMI - amber colouration). Areas expected to receive waste, but not in concentrations great enough to alter the sediment chemistry, were designated zones of influence (ZoI - green colouration). Areas classified as ZoI are expected to maintain sediment oxygen and sulphide levels equivalent to unimpacted sites located beyond the influence of aquaculture activities.

Dispersed effects – nine cage clusters

The aerial extent of the ZoHI, ZoMI and ZoI, in S1, S3 and S5 is illustrated in Figure 7-5–Figure 7-13 and outlined (in hectares) in Table 7.2. These three scenarios captured the effect of spreading the finfish standing biomass across a total of nine cage clusters (simulating a 'dispersed' effect). The effect of concentrating the finfish standing biomass across a reduced number of cage clusters (six) is explored in the subsequent chapter.

Zones of high impact were observed in S3 and S5 after 2, 3 and 5 years production and in S1 after 3 and 5 years production. Under S1, no high impacts were observed after 2 years of production (Figure 7-6 and Figure 7-7). The area occupied by the ZoHI increased in response to increasing standing biomass and the length of finfish production (Table 7.2). After 5 years continuous production, the ZoHI, as indicated by the red coloured pixels in Figure 7-5–Figure 7-13, extended respectively ~70 m, ~55 m and ~40 m from the cage cluster boundaries in S5, S3 and S1, as measured along the maximum radius down-current from the cage clusters.

Further reductions were achieved by reducing the duration of production from 5 to 3 or from 5 to 2 years (Table 7.2). For example, in S3 the ZoHI after 5 years was 132 ha in area, and extended ~55 m from the cage-cluster boundary. By reducing the production period to 3 years the ZoHI contracted to 11 ha, was constrained to small ‘patches’ within the cage cluster boundaries, and did not breach the cage cluster boundary. A further reduction to 3 ha was achieved by reducing the production period from 3 to 2 years production (Figure 7-8 and Figure 7-9). Reducing the production duration also reduced the intensity of the impact. For example, in S1, reducing the production period from 5 to 2 years resulted in a reduction in the impact status from highly (ZoHI) to moderately (ZoMI) impacted (Figure 7-5 and Figure 7-6).

The aerial extent of the ZoHI was smaller areas in the northern area, relative to the southern area. This is likely a result of the higher current speeds in the northern MWADZ, which when simulated in the model, imparted a strong influence on particle transport and resuspension–both processes which affected the retention of organic material near the sea-cages. Particles tended to disperse under higher current speeds, but tended to sink, deposit and remain close to the sea-cages under lower current speeds. This is reflected in Figure 7-5–Figure 7-13, by the greater spread of particles away from the sea-cages in the northern MWADZ, and the greater tendency toward deposition and concentration of particles in the southern MWADZ.

Zones of moderate impact, as indicated by the amber coloured pixels in Figure 7-5–Figure 7-13, were observed in all scenarios irrespective of the length of the production period. With some exceptions, the area occupied by the ZoMI increased with increasing standing biomass and increasing length of production; however, the changes were less dramatic than those predicted for the ZoHI. For example, the area occupied by the ZoHI over the range of modelling treatments was between 0 ha and 177 ha, representing an order of magnitude change; whereas the area occupied by the ZoMI over the same modelling treatments was between 239 ha and 348 ha, representing a smaller, and within order of magnitude change.

The Zone of Influence, as indicated by the green coloured pixels in Figure 7-5–Figure 7-13, was the largest (in area) and the most dispersed of the three impact categories. In the northern area of the MWADZ, the higher current speeds acted to increase the dispersion of organic particles, which in turn increased the area occupied by the ZoI. The prevailing south-easterly currents in the northern area of the MWADZ are reflected in the north-westerly trajectory of particles to the north-west and away from the sea-cages. In the southern area of the MWADZ, the ZoI was generally more constrained, and centred around the individual cage-clusters. Dominant westerly currents in the southern area of the MWADZ resulted in a tendency for particles to disperse to the west of the cage clusters.

Table 7.2 Areas occupied by the zones of high and moderate impact and the zone of influence under scenarios S1, S3 and S5 after 2, 3 and 5 years production

Years of production	Scenario No.	Standing biomass (t)	ZoHI (ha)	ZoMI (ha)	ZoI (ha)
5	S1	15 000	117	239	1150
	S3	24 000	132	235	1005
	S5	30 000	177	270	1226
3	S1	15 000	1	346	1159
	S3	24 000	11	349	1012
	S5	30 000	105	334	1235
2	S1	15 000	0	336	1170
	S3	24 000	3	348	1021
	S5	30 000	91	333	1250

Note:

1. ZoHI = zone of high impact, ZoMI = zone of moderate impact, ZoI = zone of influence

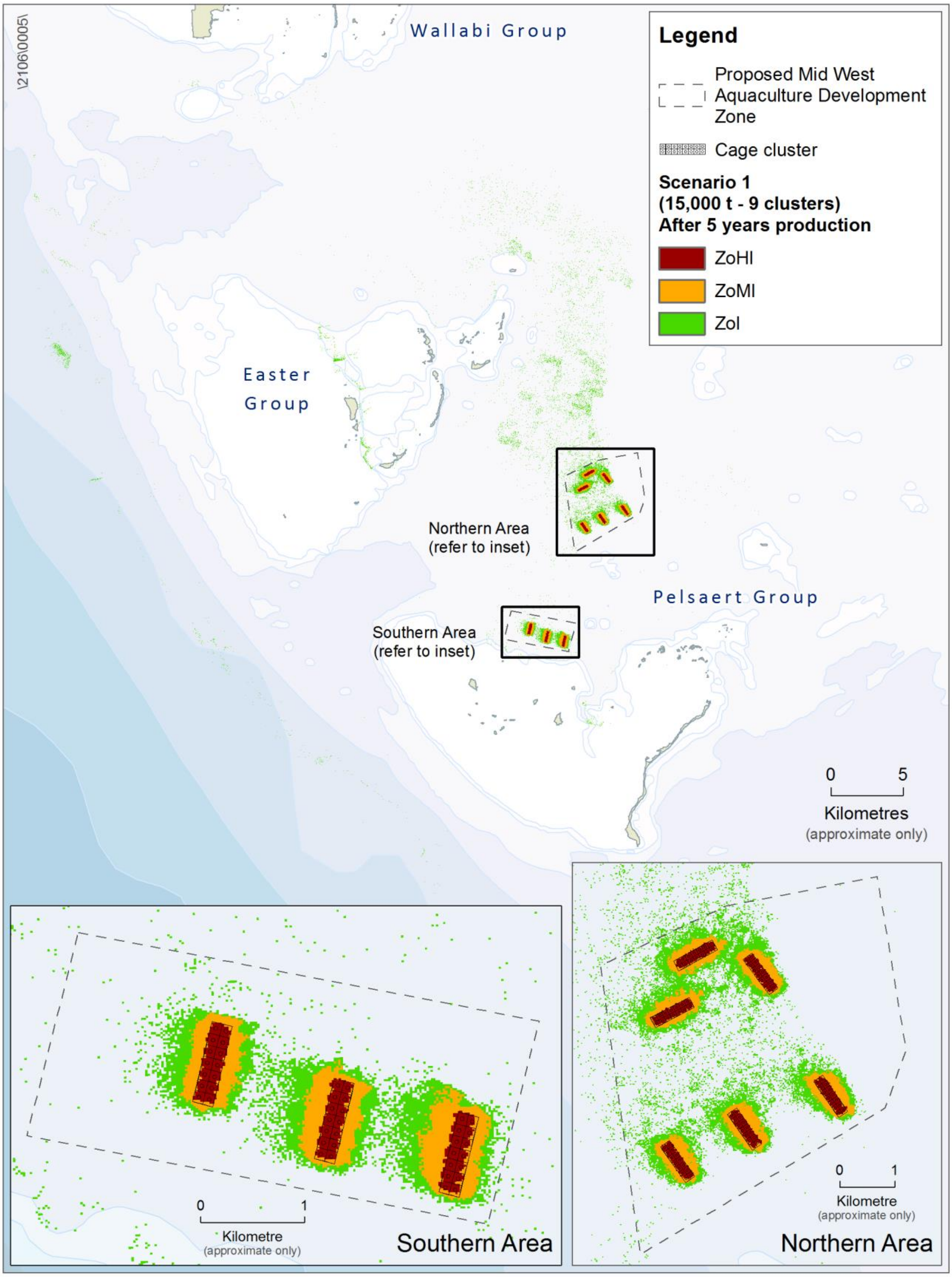


Figure 7-5 Zones of impact under scenario 1 (15 000 t) after 5 years production

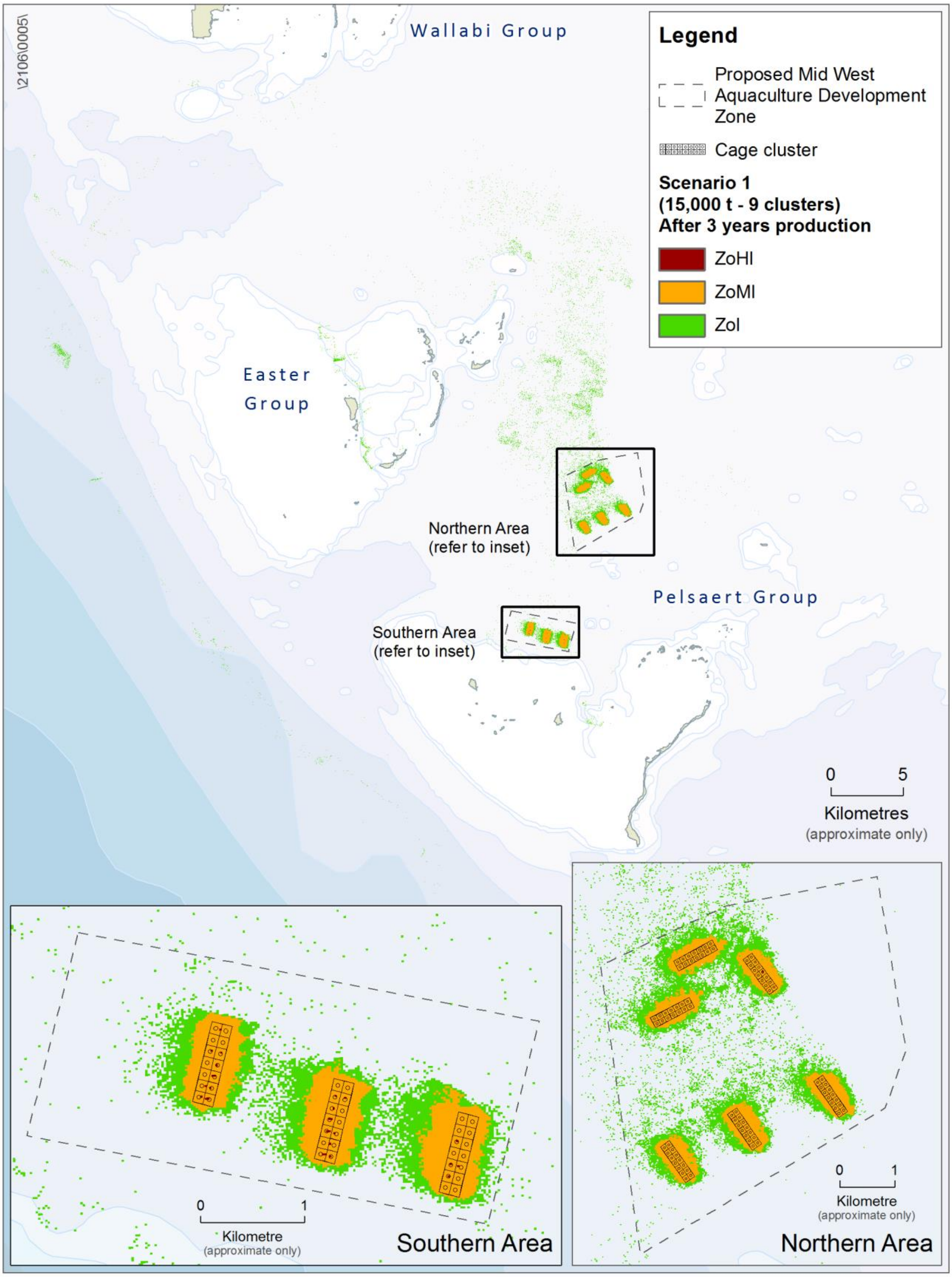


Figure 7-6 Zones of impact under scenario 1 (15 000 t) after 3 years production

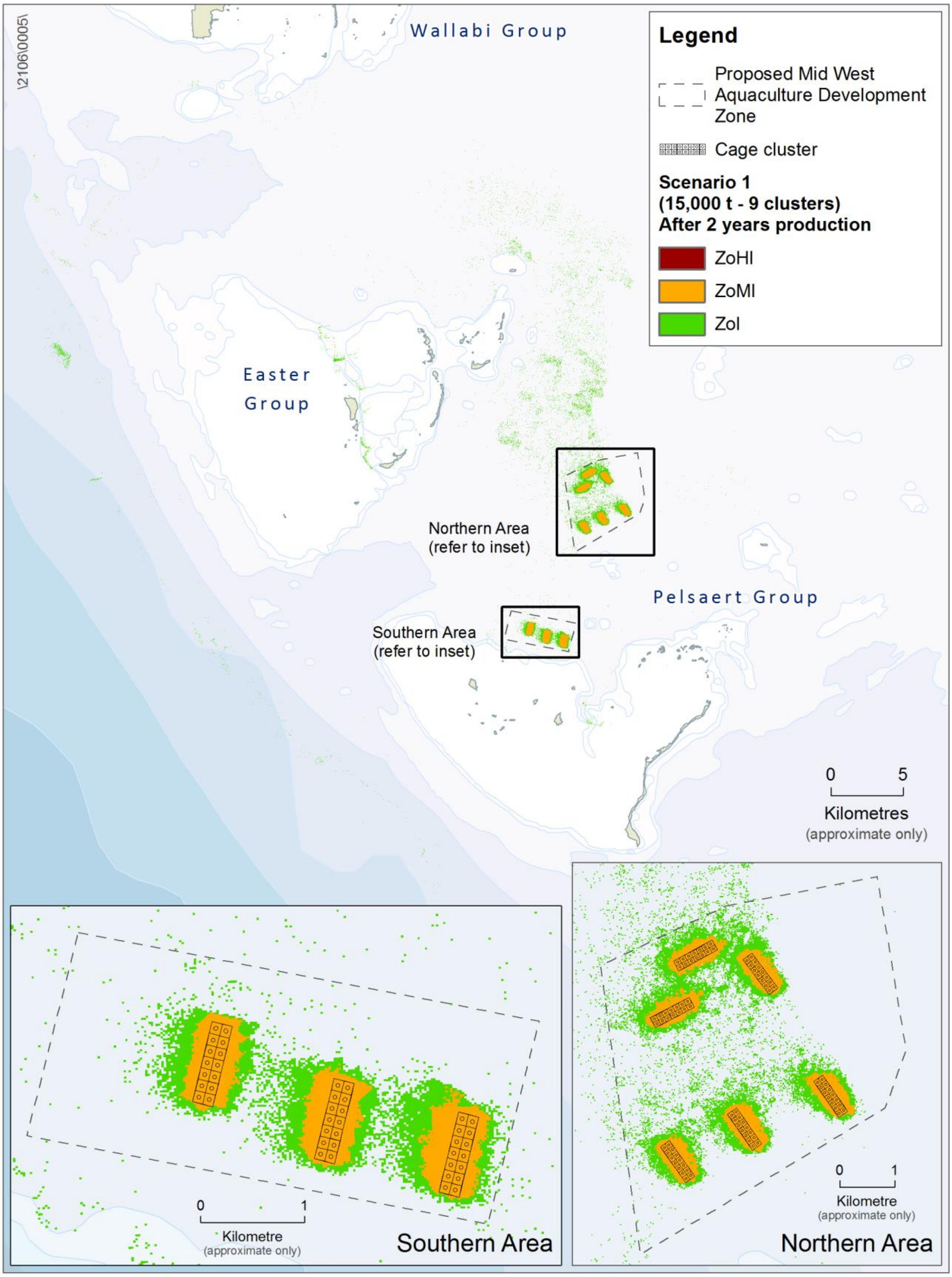


Figure 7-7 Zones of impact under scenario 1 (15 000 t) after 2 years production

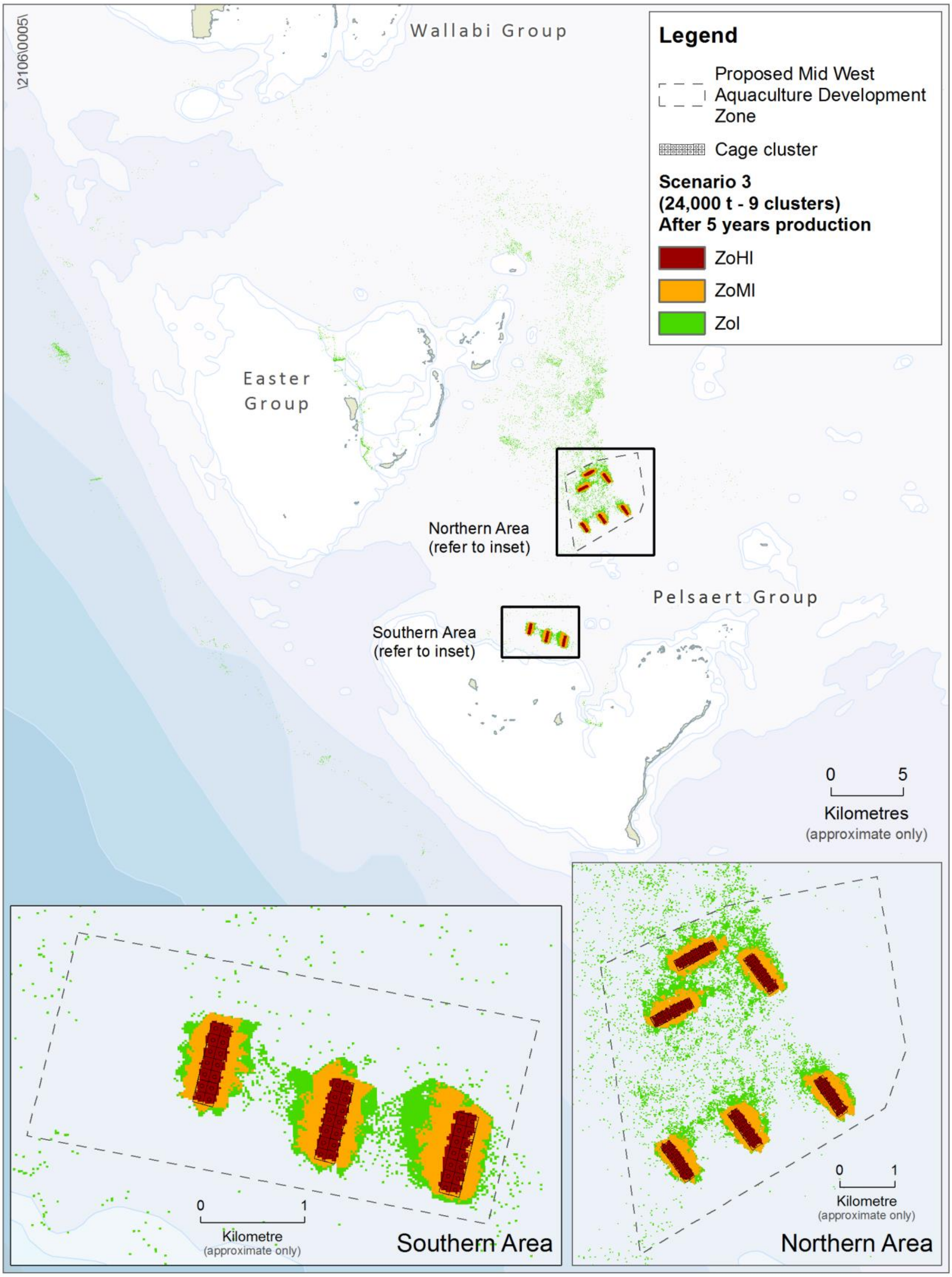


Figure 7-8 Zones of impact under scenario 3 (24 000 t) after 5 years production

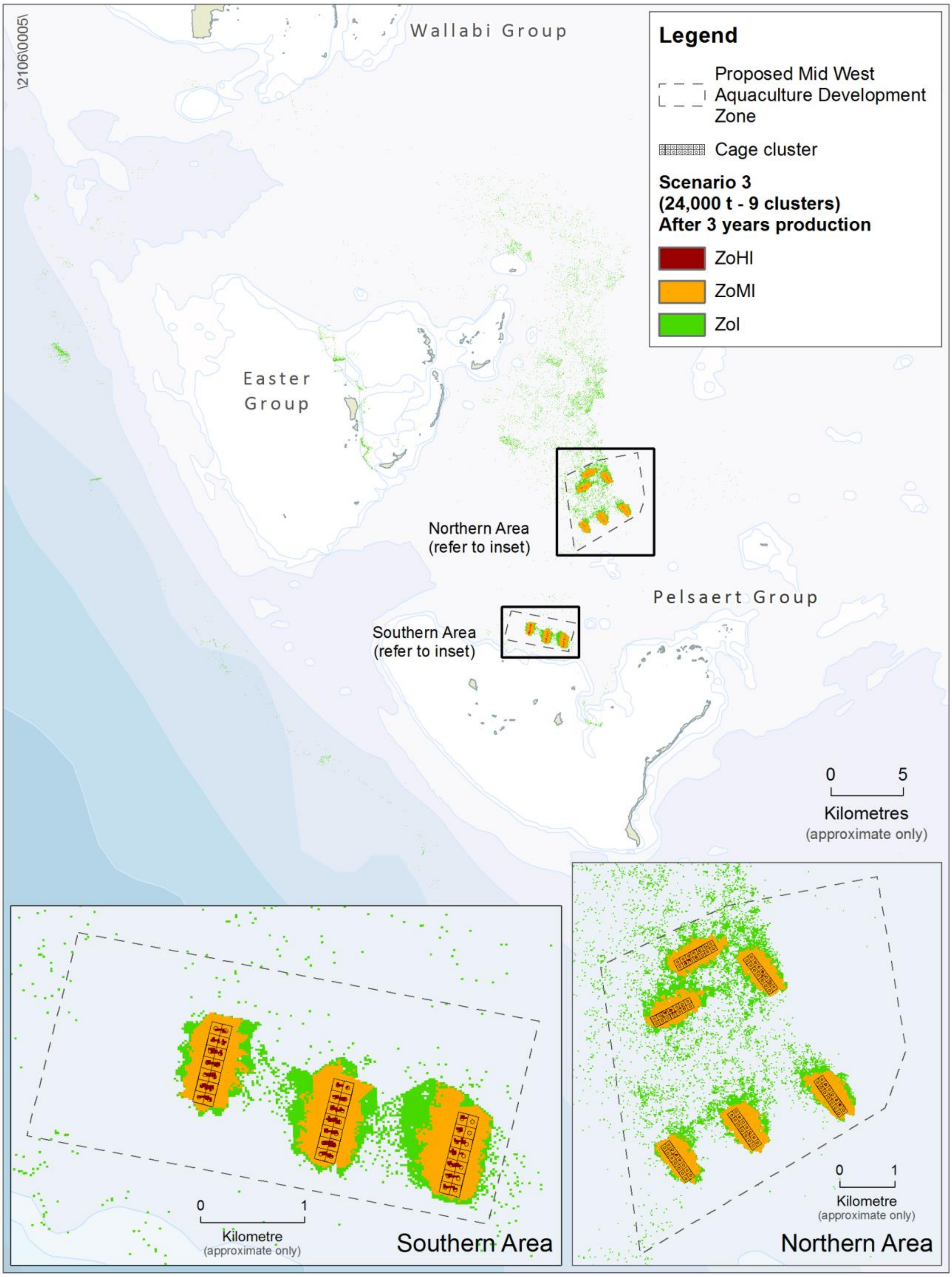


Figure 7-9 Zones of impact under scenario 3 (24 000 t) after 3 years production

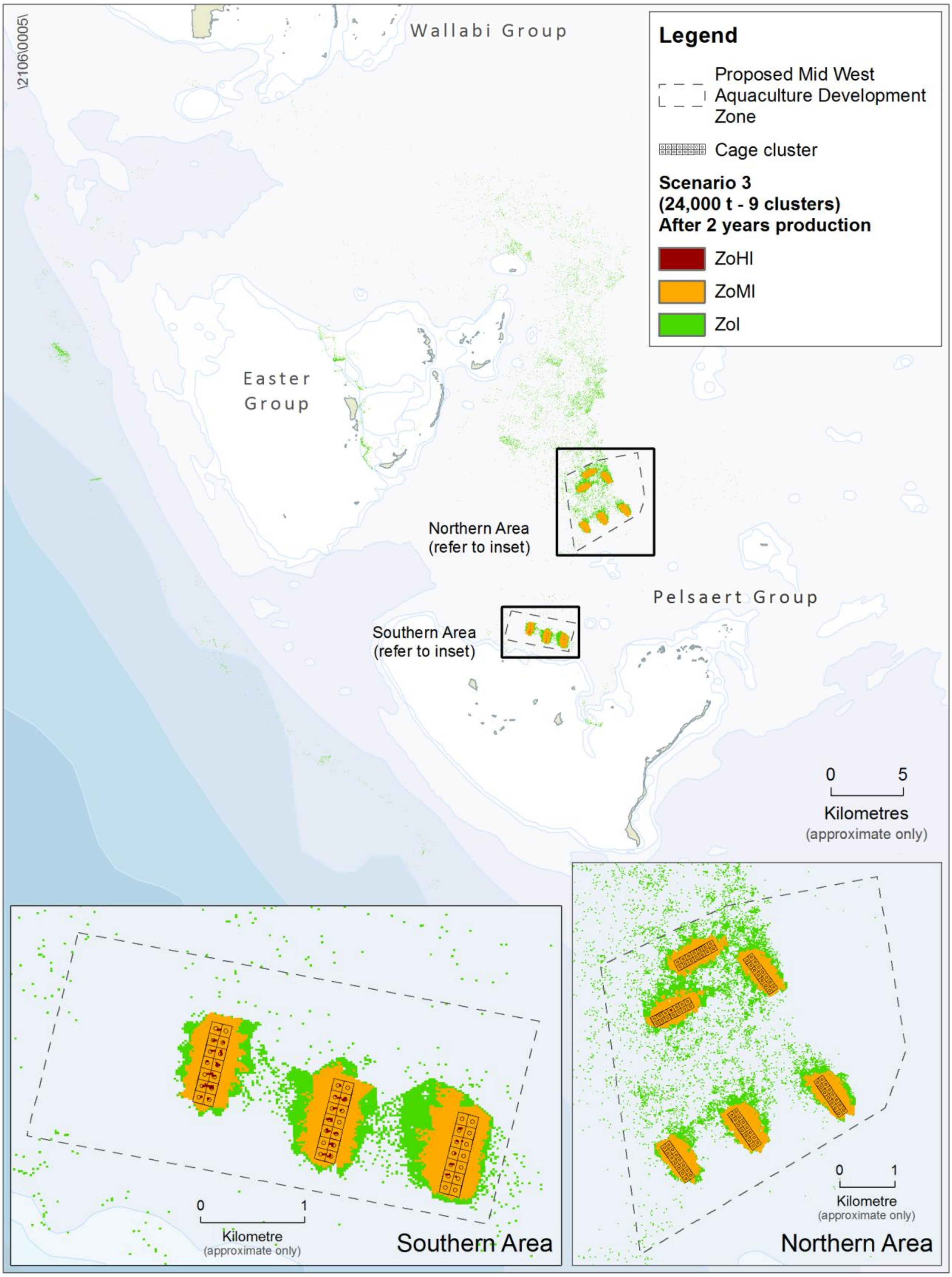


Figure 7-10 Zones of impact under scenario 3 (24 000 t) after 2 years production

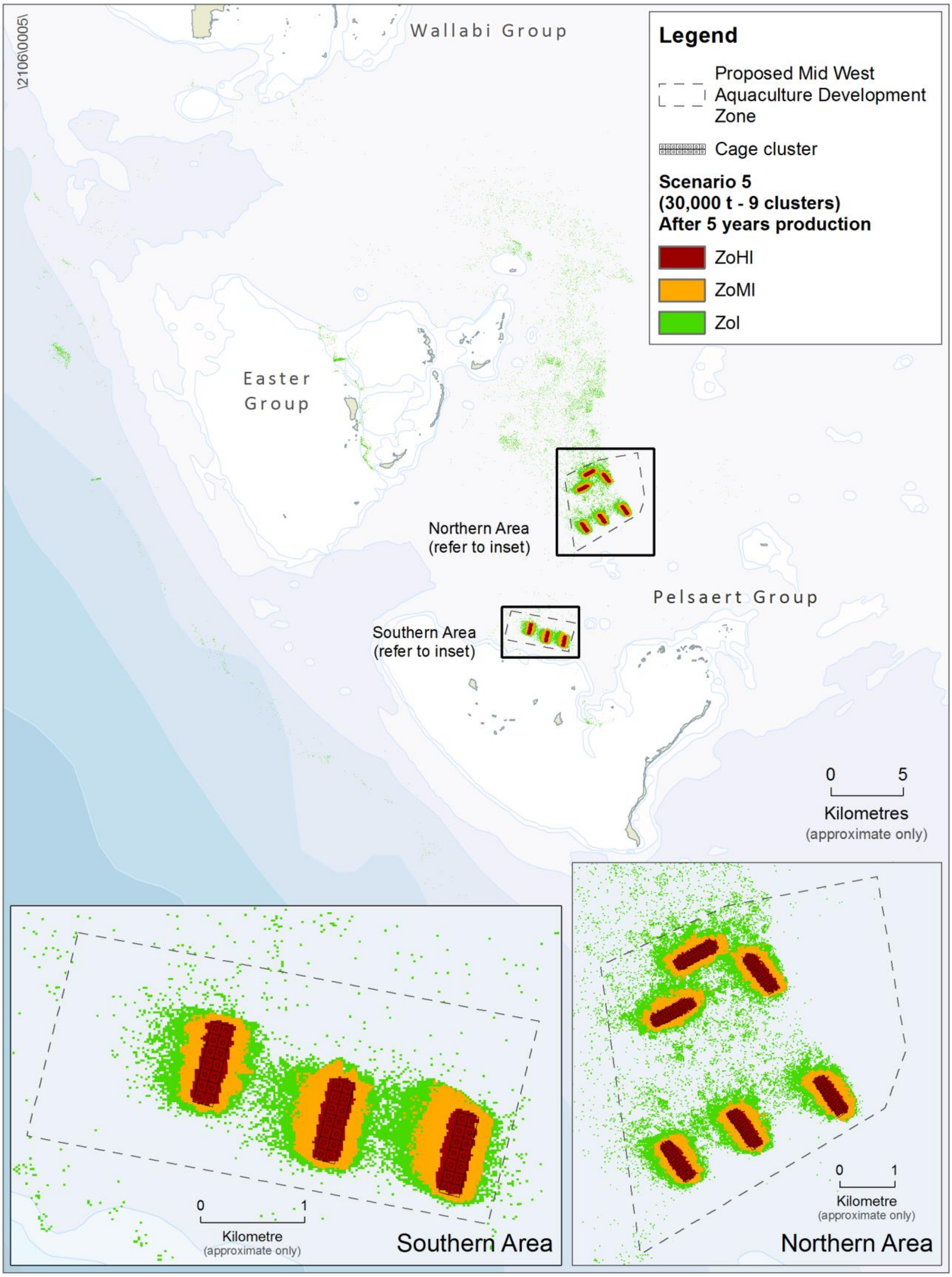


Figure 7-11 Zones of impact under scenario 5 (30 000 t) after 5 years production

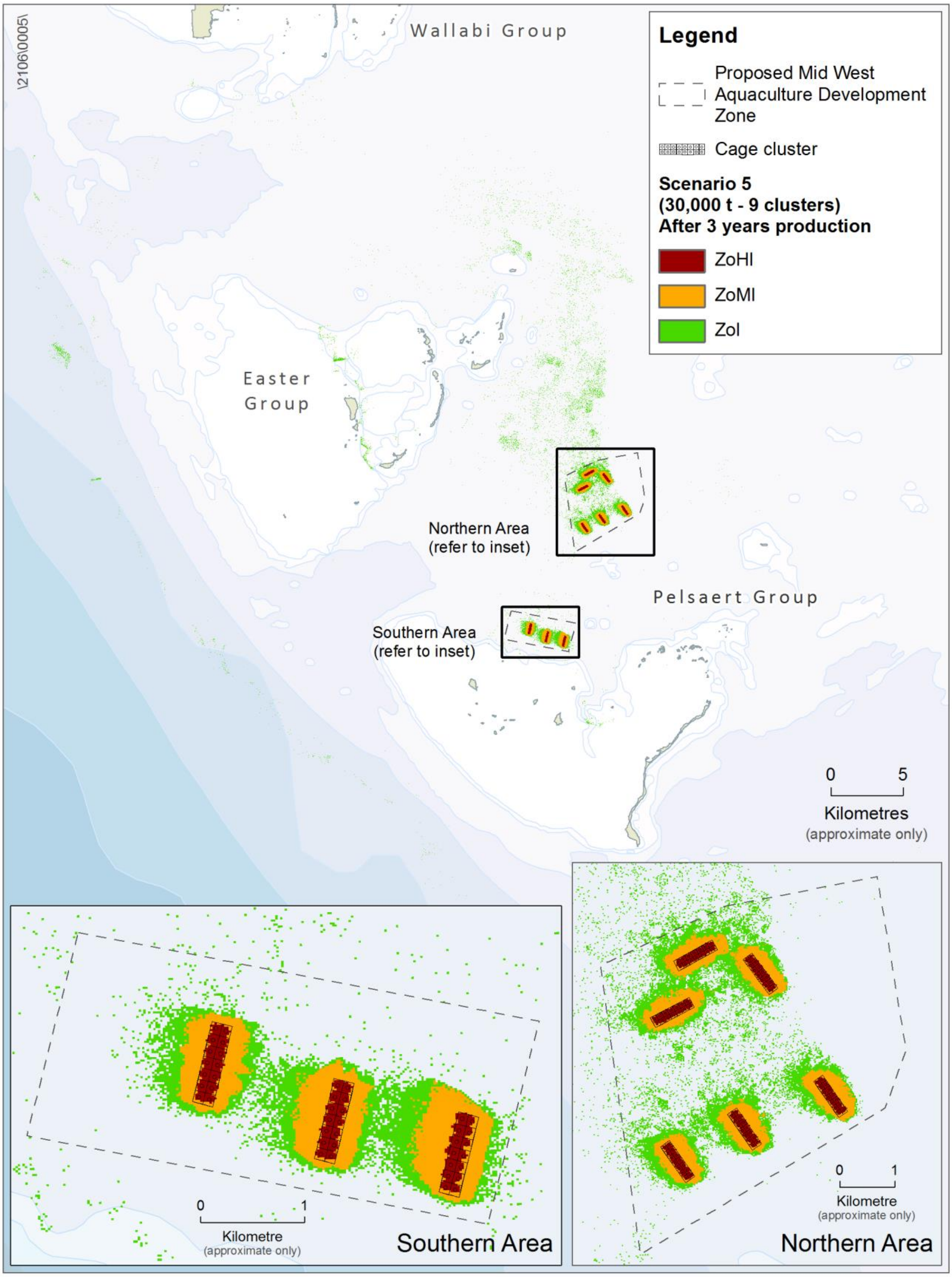


Figure 7-12 Zones of impact under scenario 5 (30 000 t) after 3 years production

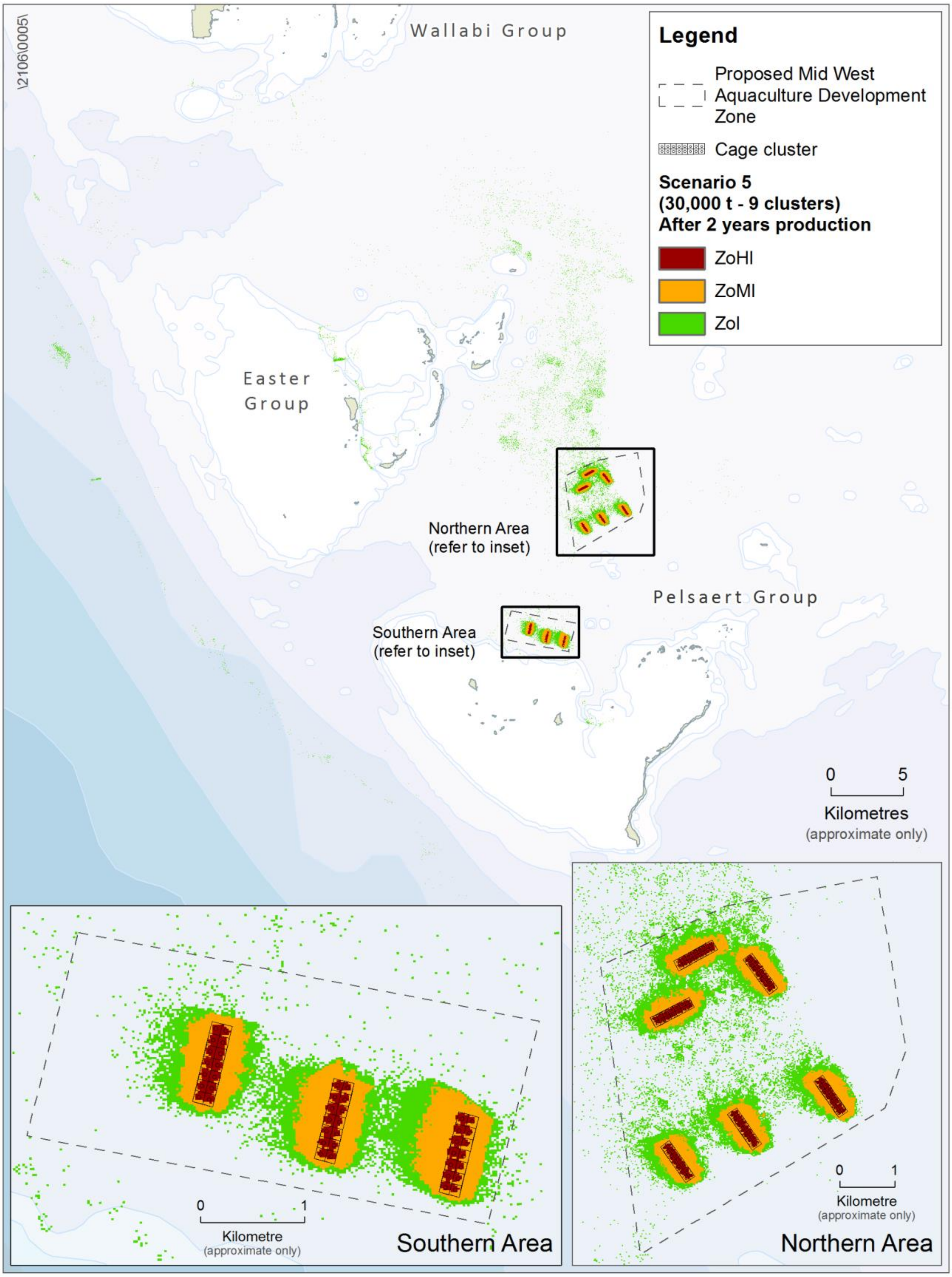


Figure 7-13 Zones of impact under scenario 5 (30 000 t) after 2 years production

Concentrated effects - six cage clusters

The aerial extent of the ZoHI, ZoMI and ZoI, in S2, S4 and S6 is illustrated in Figure 7-14–Figure 7-22 and outlined (in hectares) in Table 7.3. These scenarios captured the effect of concentrating the standing biomass across a total of 6 cage clusters, 3 less than in the 'dispersed' effects simulations (described in the chapter above).

As with the results for the 'dispersed' effects', the ZoHI, as indicated by the red coloured pixels in Figure 7-14–Figure 7-22, increased with standing biomass and the length of finfish production. Zones of high impact were observed in S6, S4 and S2 after 5 and 3 years production and in S6 and S4 after 2 years production. The area occupied by the ZoHI in S2 after 2 years production was marginal at less than 1 ha (Figure 7-14–Figure 7-22).

Significant reductions in the areas of the ZoHI were achieved by reducing the length of production from 5 to 3, and from 3 to 2 years. For example, by reducing the length of production from 5 to 3 years, close to a 100% reduction was achieved in S2, a 45% reduction was achieved in S4 and a 31% reduction was achieved in S6. Greater reductions were achieved for the dispersed effects scenarios, S1, S3 and S5: corresponding to reductions of 100% for S1, 92% for S3 and 41% for S6 (Table 7.2 and Table 7.3).

Reductions in both the standing biomass and the length of production also reduced the maximum extent of the ZoHI, as measured along the maximum radius down-current from the cage clusters. After 5 years continuous production, the ZoHI, as indicated by the red coloured pixels in Figure 7-14–Figure 7-22, extended ~110 m, ~60 m and ~50 m from the cage cluster boundaries in S6, S4 and S2, respectively. However, the maximum distances reduced after 3 and 2 years production: with predictions of 10 m and 15 m respectively under S4, and 55 m and 50 m respectively under S6. Under S2, the ZoHI did not breach the cage cluster perimeter.

Increasing the stocking density, while maintaining the standing biomass (i.e. stocking density S4 > stocking density S3; standing biomass S4 = standing biomass S3), had the effect of reducing the total area occupied by the ZoHI across the zone. This effect was particularly strong after 5 years production (Table 7.2 and Table 7.3), but less so after 3 and 2 years production. For example, after 5 years, the total area occupied by the ZoHI was 177 ha and 139 ha for S5 and S6, respectively; 132 ha and 113 ha for S3 and S4 respectively; and 117 ha and 82 ha for S1 and S2, respectively. After 3 years production, the results were more variable: the total area occupied by the ZoHI was higher in S2 (2 ha) relative to S1 (1 ha); higher in S4 (62 ha) relative to S3 (11 ha) but lower in S6 (95 ha) relative to S5 (105 ha). Similar variable results were achieved after 2 years production (Table 7.2 and Table 7.3).

Reducing the number of cage clusters also reduced the total area occupied by the ZoMI and the ZoI. By reducing the number of cage clusters, reductions in the footprints of both zones were achieved irrespective of the standing biomass or the production period modelled (Table 7.2 and Table 7.3). This is a useful finding indicating that reductions in the spatial extent of impacts, as measured under EAG 7 (ZoHI, ZoMI and ZoI), can be achieved by concentrating finfish in individual cage clusters, without a corresponding need to reduce the total standing biomass across the zone. It was noted, however, that while the spatial extent of the impacts can be reduced based on the criteria in EAG 7, the effect of this is to increase the intensity of impacts immediately under the sea-cages. Intensifying the impacts, as S2, S4 and S6, translate to longer recovery periods, as shown in Figure 7-23–Figure 7-31. The difference in the areas occupied between the dispersed (9 clusters) and concentrated (6 clusters) scenarios is shown in Table 7.2 and Table 7.3, and illustrated in Figure 7-14–Figure 7-22.

As observed in S1, S3 and S5, the area occupied by the ZoHI in S2, S4 and S6 also increased in response to increasing standing biomass and the length of finfish production. Zones of high impact were observed in S6, S4 and S2 after 5 and 3 years production and in S6 and S4 after 2 years production. The area occupied by the ZoHI in S2 after 2 years production was marginal at less than 1 ha (Figure 7-14–Figure 7-22).

The area occupied by the ZoHI after 2, 3 and 5 years production increased proportionally with increases in standing biomass, increasing from 82 ha in S2 to 139 ha in S6 after 5 years, 2 ha in S2 to 95 ha in S6 after 3 years and 0.2 ha in S2 to 88 ha in S6 after 2 years. Similar increases were apparent with the ZoMI, which increased in size from 160 ha in S2 to 203 ha in S6, after 5 years. The area occupied by the ZoI was also observed to increase in response to increasing standing biomass, reaching a maximum coverage in S6, irrespective of the length of production (Table 7.3).

Significant reductions in the areas of the ZoHI were achieved by reducing the length of production from 5 to 3, and from 3 to 2 years. For example, by reducing the production period from 5 to 3 years close to 100% reductions were achieved in S2, 45% reductions were achieved in S4 and 31% reductions were achieved in S6. Greater reductions were achieved for the dispersed effects scenarios, S1, S3 and S5: corresponding to reductions of 100% for S1, 92% for S3 and 41% for S6.

Table 7.3 Areas occupied by the zones of high and moderate impact and the zone of influence under scenarios S2, S4 and S6 after 2, 3 and 5 years production

Years of production	Scenario No.	Standing biomass (t)	ZoHI (ha)	ZoMI (ha)	ZoI (ha)
5	S2	15 000	82	160	616
	S4	24 000	113	173	697
	S6	30 000	139	203	861
3	S2	15 000	2	234	621
	S4	24 000	62	219	701
	S6	30 000	95	241	868
2	S2	15 000	0.2	229	628
	S4	24 000	51	222	710
	S6	30 000	88	237	879

Note:

1. ZoHI = zone of high impact, ZoMI = zone of moderate impact, ZoI = zone of influence

Zones of moderate impact, as indicated by the amber coloured pixels in Figure 7-14–Figure 7-22, were observed in all scenarios irrespective of the length of the production period. The ZoMI was restricted to the area immediately adjacent to the sea-cage clusters, but extended further than the ZoHI. As with the ZoHI, the area occupied by the ZoMI increased with increasing standing biomass and the length of production; however, the changes were less distinct than those observed for the ZoHI. Unlike the ZoHI, which was near absent in S2 after 2 years production, moderate impacts were detected irrespective of the modelled treatment.

The Zone of Influence, as indicated by the green coloured pixels in Figure 7-14–Figure 7-22, was the largest (in area) and the most dispersed of the three impact categories. In the northern area of the MWADZ, the higher current speeds acted to increase the dispersion of organic particles, which in turn increased the area occupied by the ZoI. The prevailing south-easterly currents in the northern area of the MWADZ are reflected in the north-westerly trajectory of the ZoI, which was predicted to advect away from the sea-cages. In the southern area of the MWADZ, the ZoI was generally more constrained, and centred on the individual cage-clusters.

The ZoHI is the area where impacts on benthic habitats are predicted to be irreversible, as per EAG 7. The term irreversible is defined as ‘lacking a capacity to return or recover to a state resembling that prior to being impacted within a timeframe of five years or less’. Despite the use of the term irreversible, it is noted that sea-cages are not permanent structures and can be moved to facilitate benthic rehabilitation. Recovery times in the ZoHI and ZoMI ranged between 1 and 7+ years, depending on the scenario and distance from the sea-cages. Immediately under the sea-cages, sediments required greater than 7 years to achieve full recovery. However, this reduced to 6 and 5-6 after 3 and 2 years production respectively (Figure 7-23–Figure 7-31).

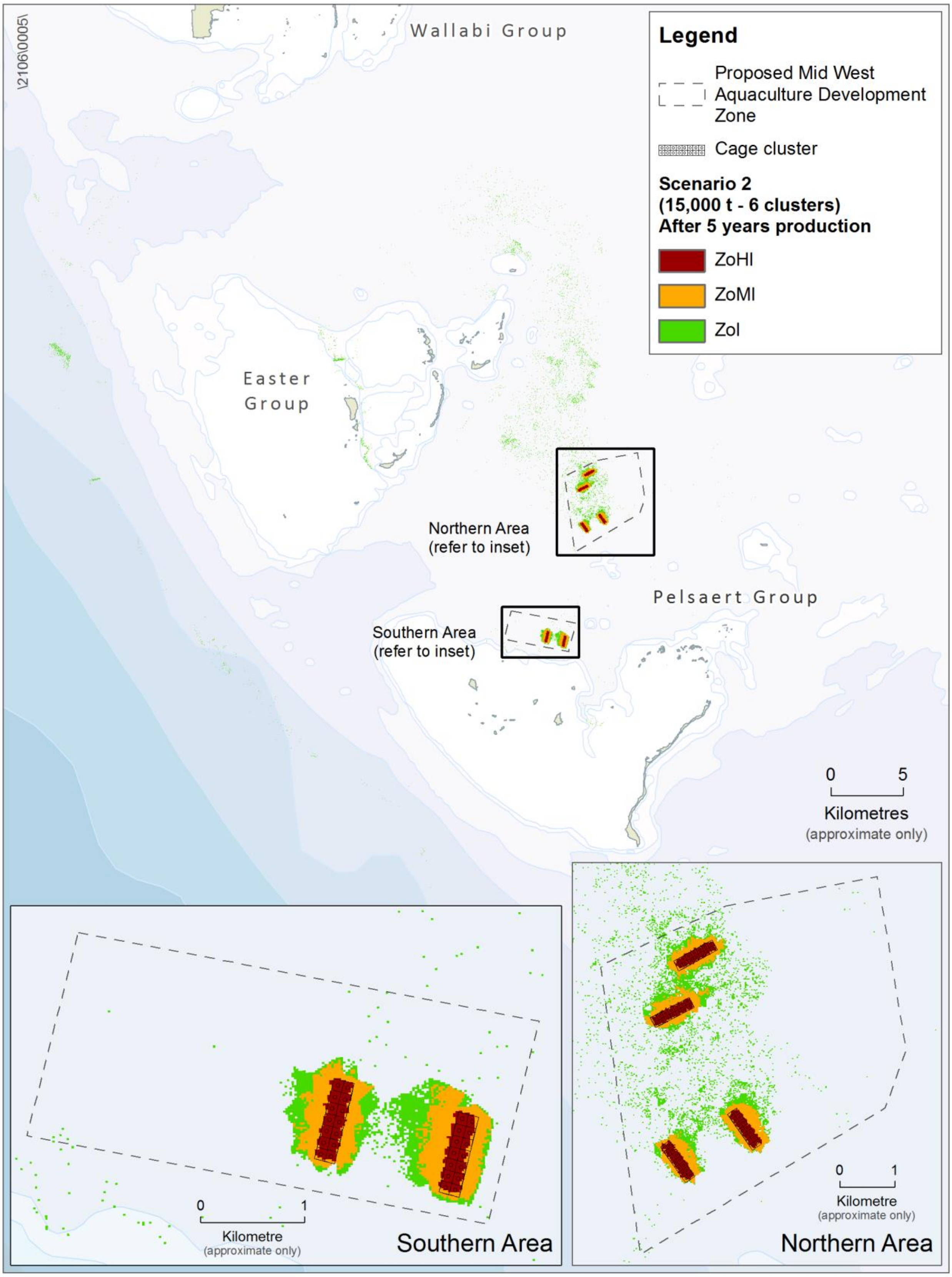


Figure 7-14 Zones of impact under scenario 2 (15 000 t) after 5 years production

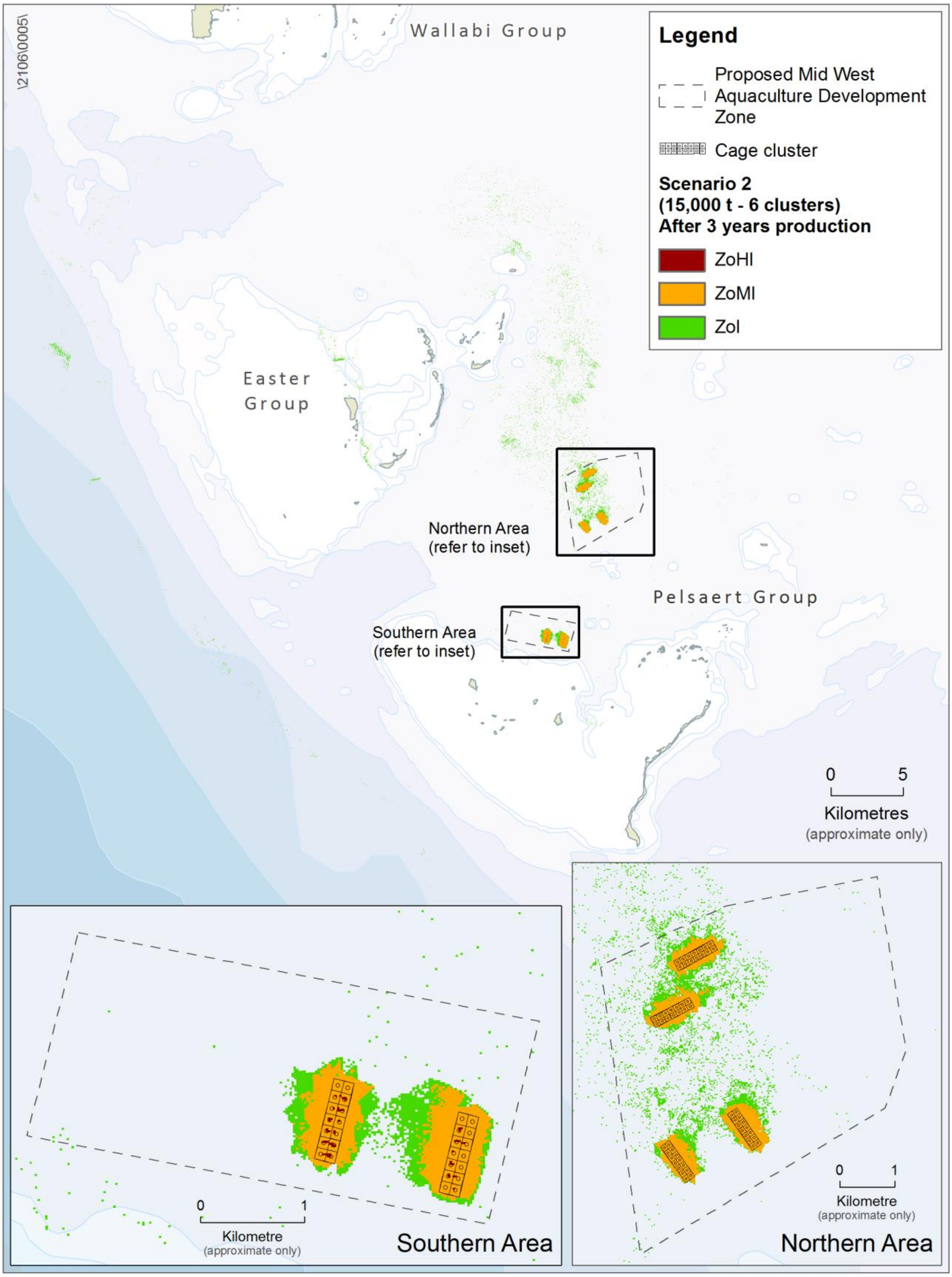


Figure 7-15 Zones of impact under scenario 2 (15 000 t) after 3 years production

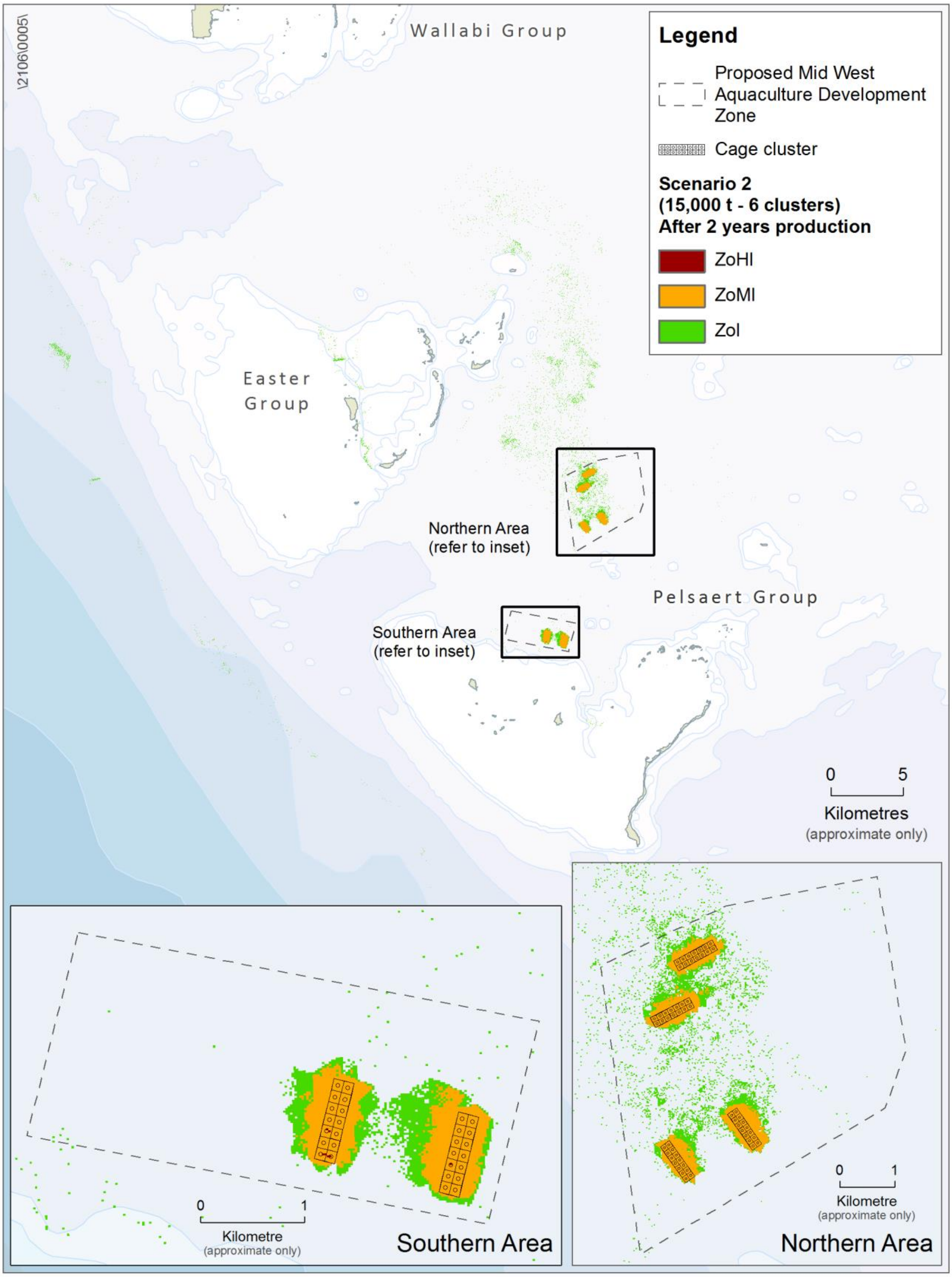


Figure 7-16 Zones of impact under scenario 2 (15 000 t) after 2 years production

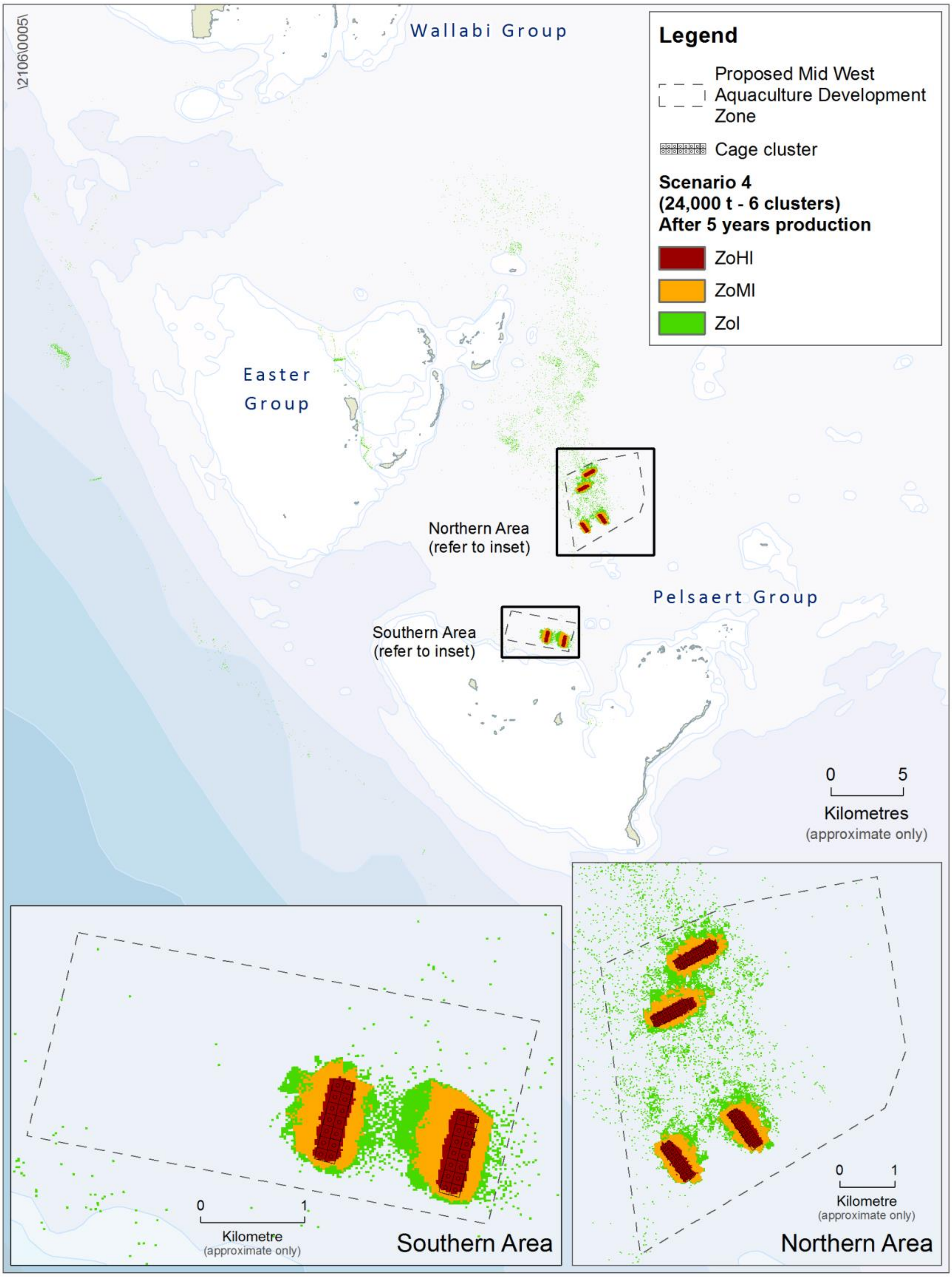


Figure 7-17 Zones of impact under scenario 4 (24 000 t) after 5 years production

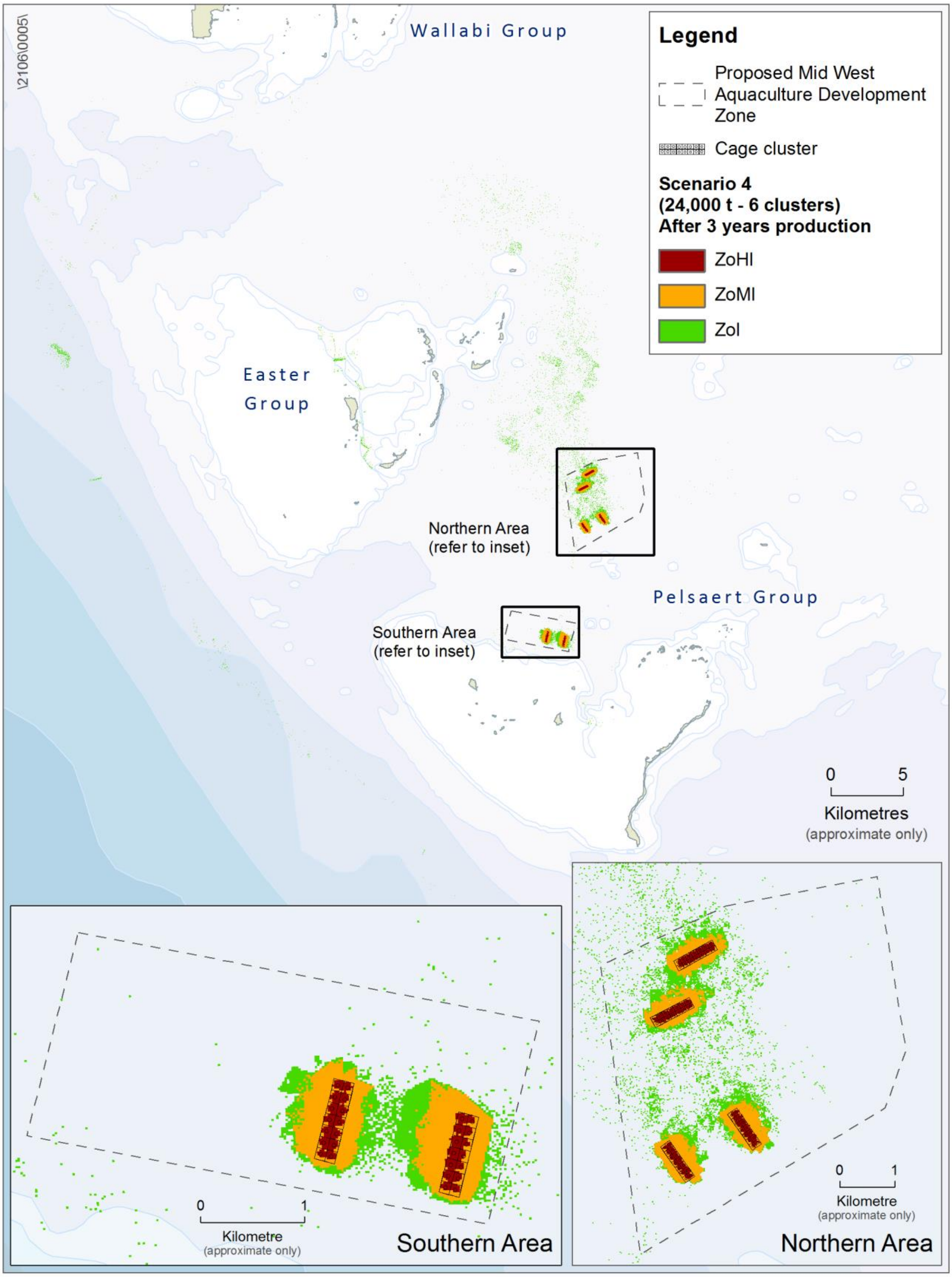


Figure 7-18 Zones of impact under scenario 4 (24 000 t) after 3 years production

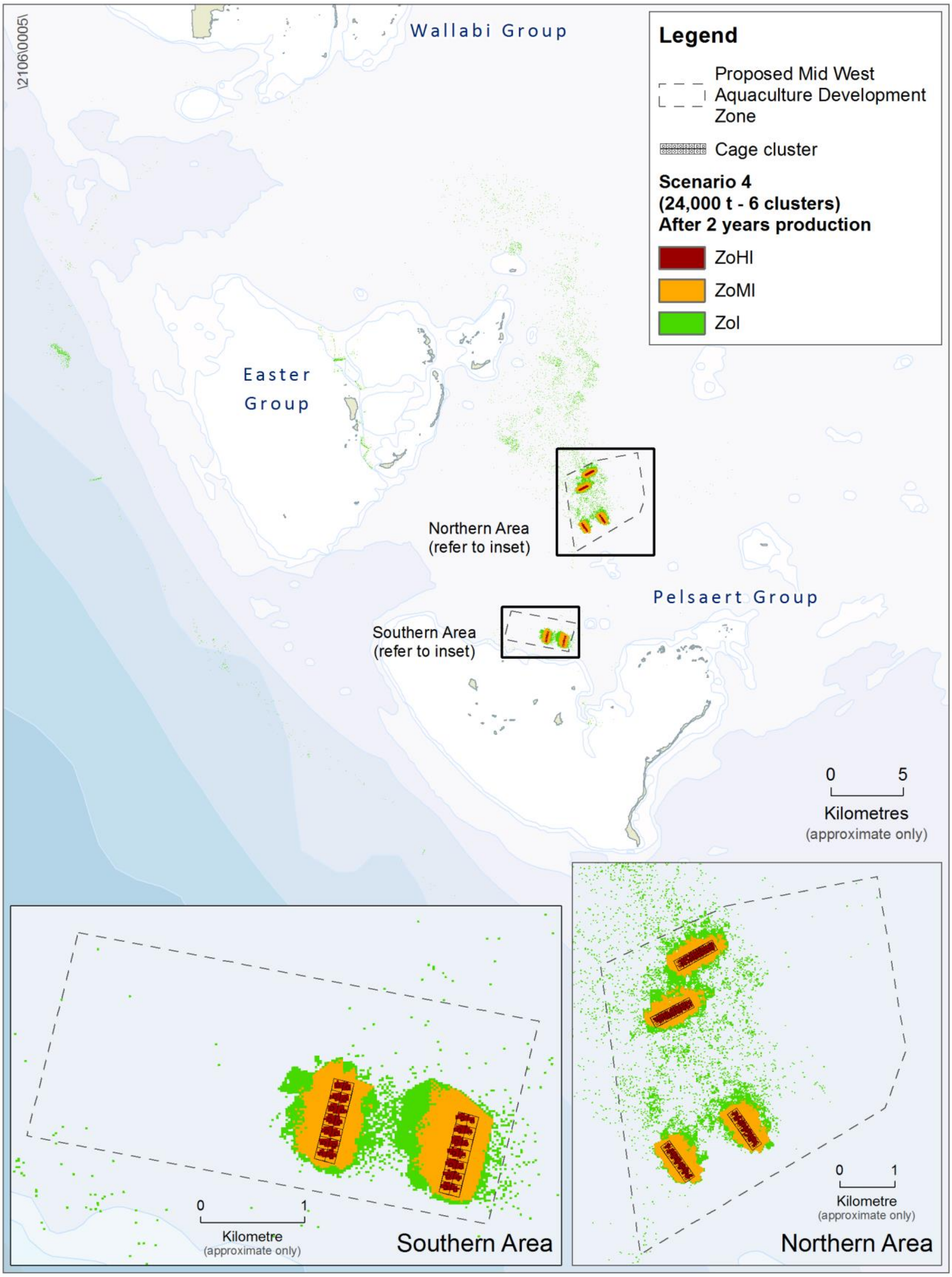


Figure 7-19 Zones of impact under scenario 4 (24 000 t) after 2 years production

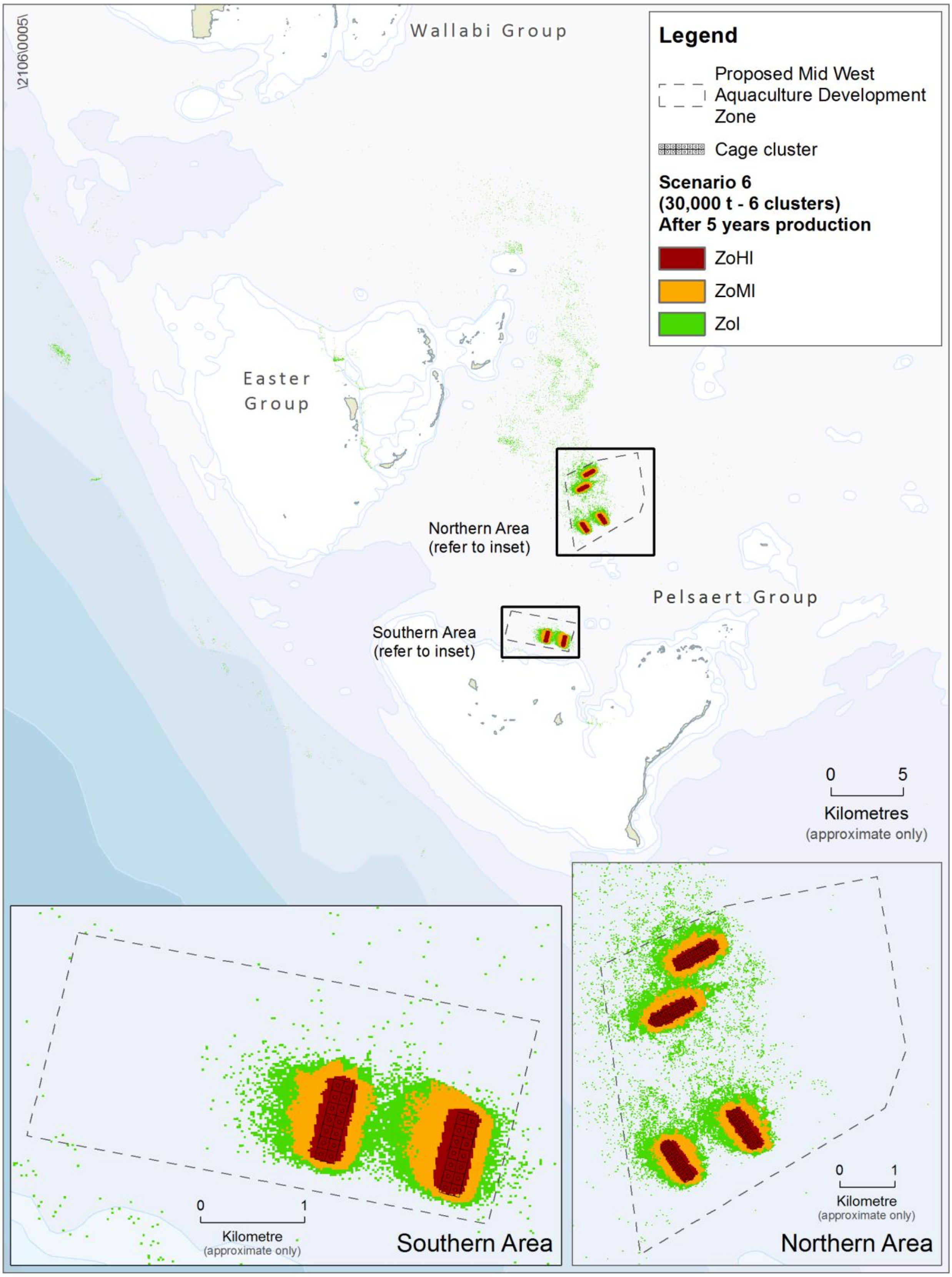


Figure 7-20 Zones of impact under scenario 6 (30 000 t) after 5 years production

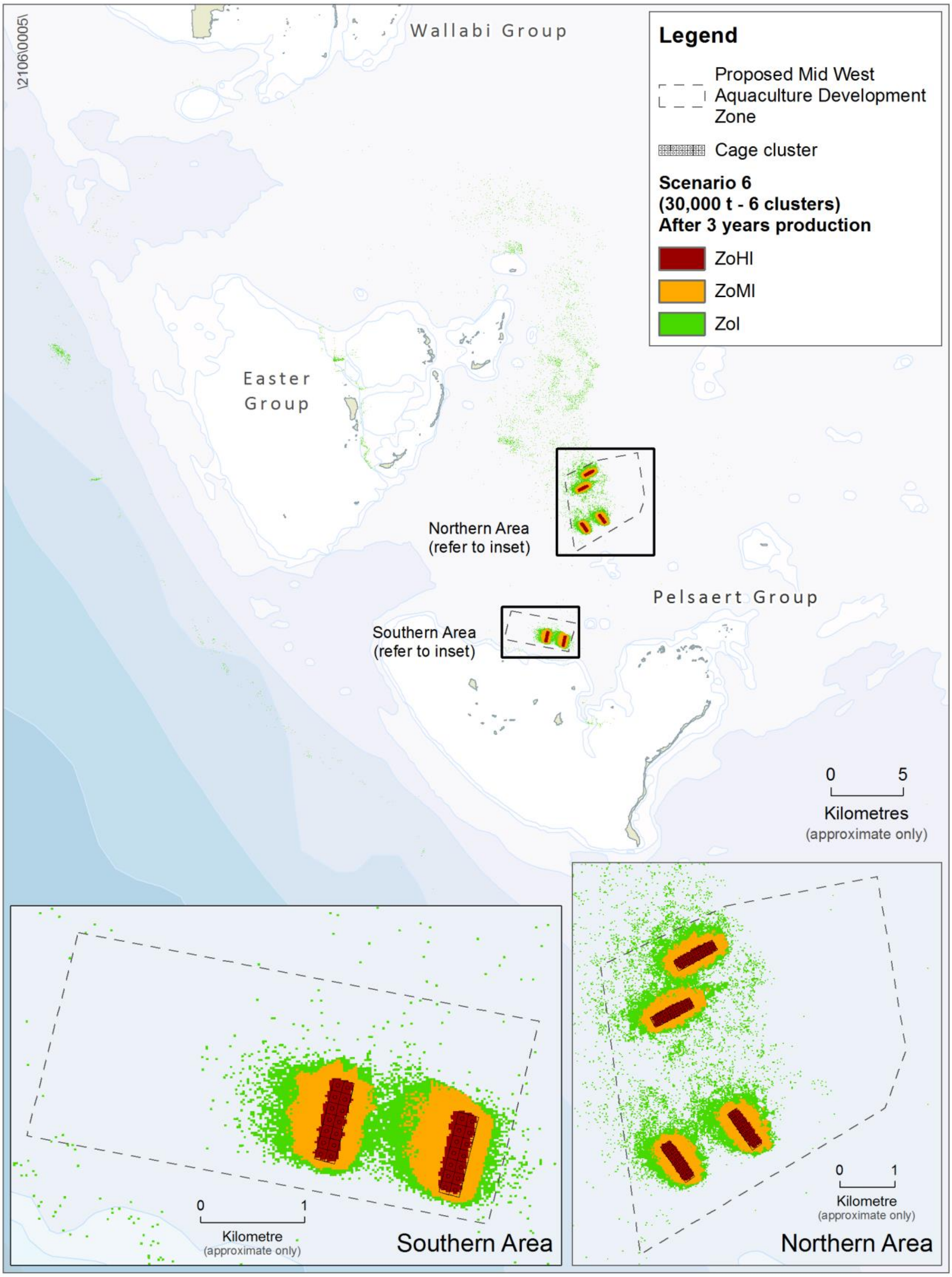


Figure 7-21 Zones of impact under scenario 6 (30 000 t) after 3 years production

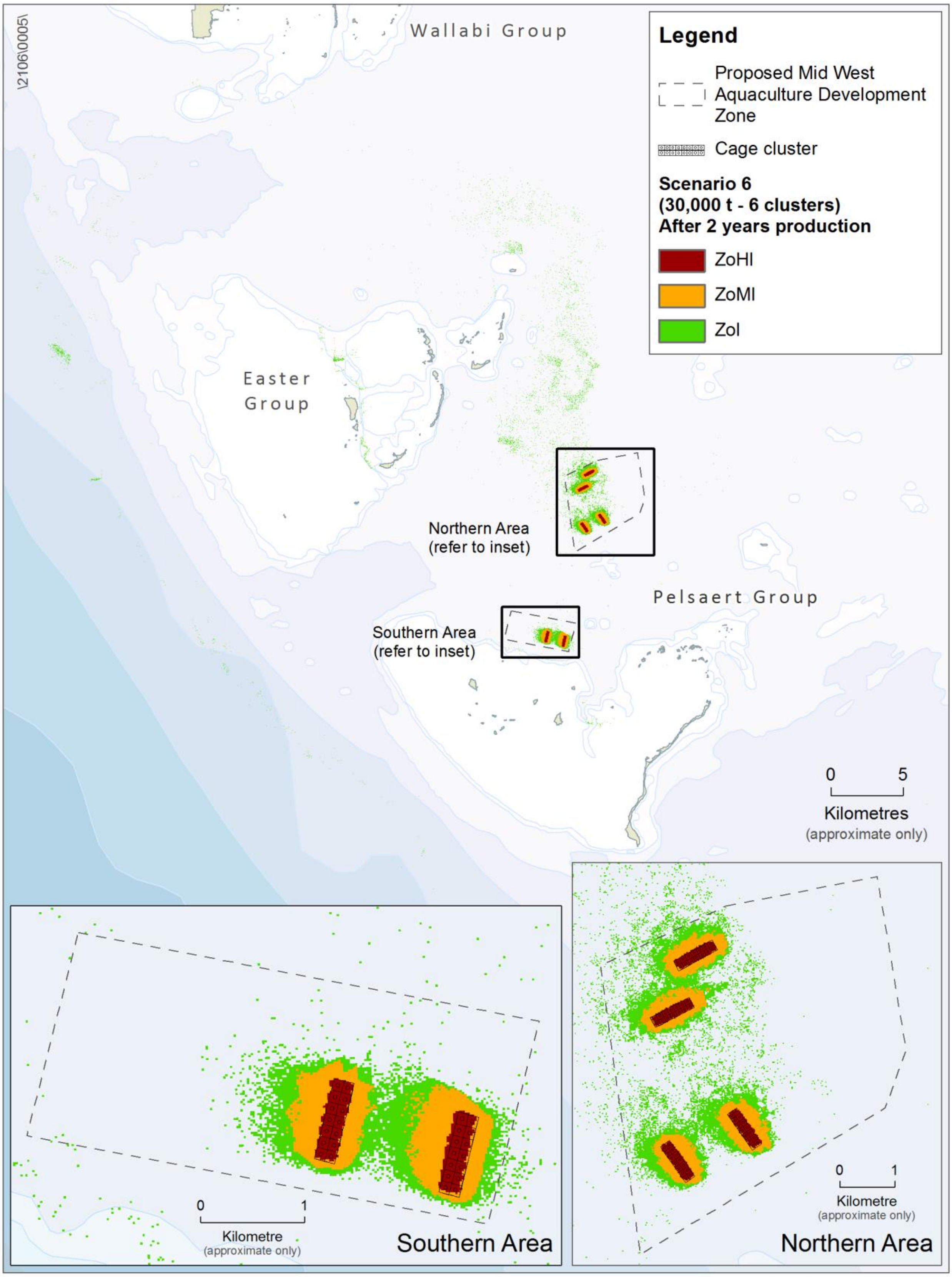


Figure 7-22 Zones of impact under scenario 6 (30 000 t) after 2 years production

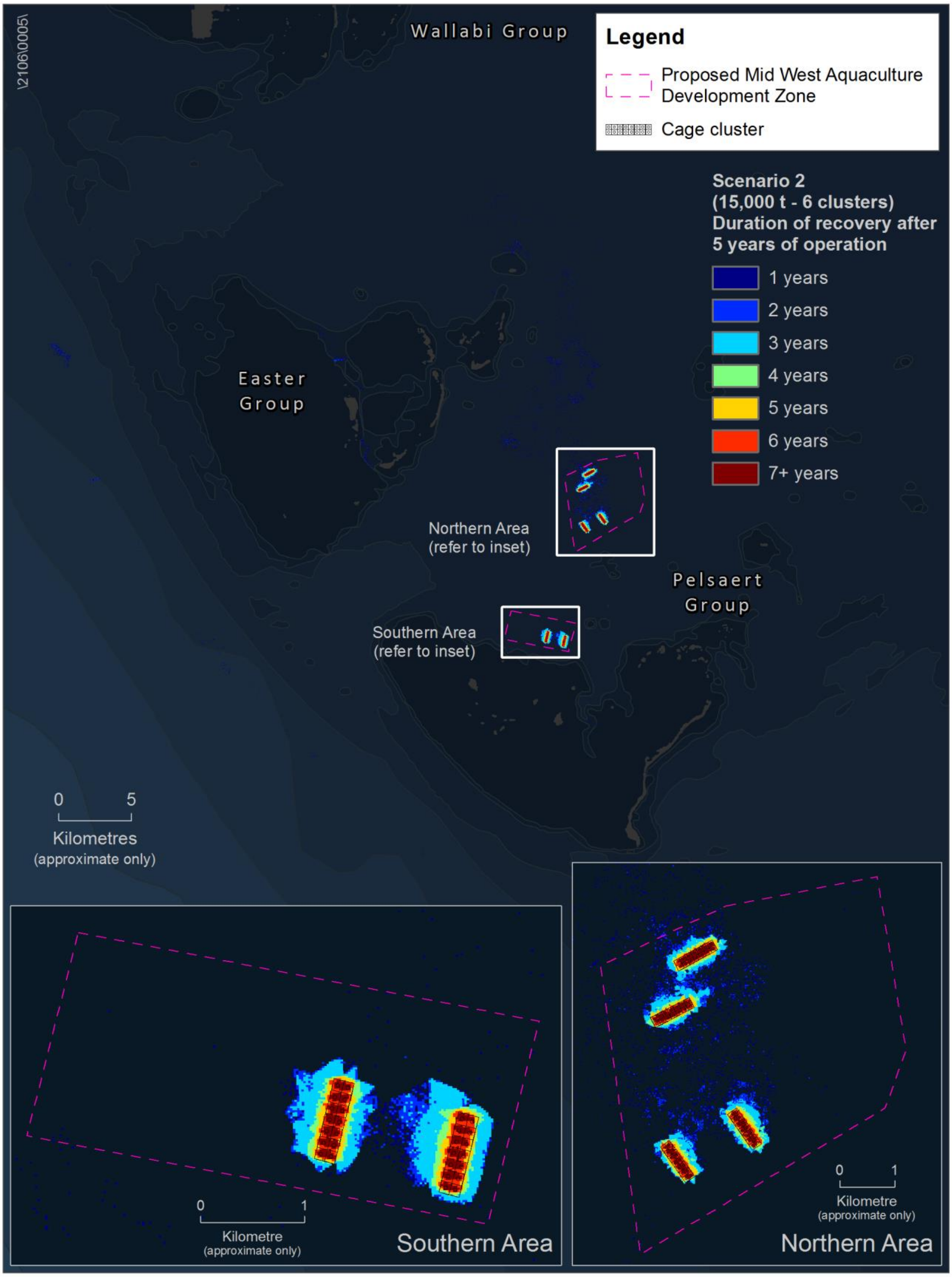


Figure 7-23 Duration of recovery under scenario 2 (15 000 t) after 5 years of operation

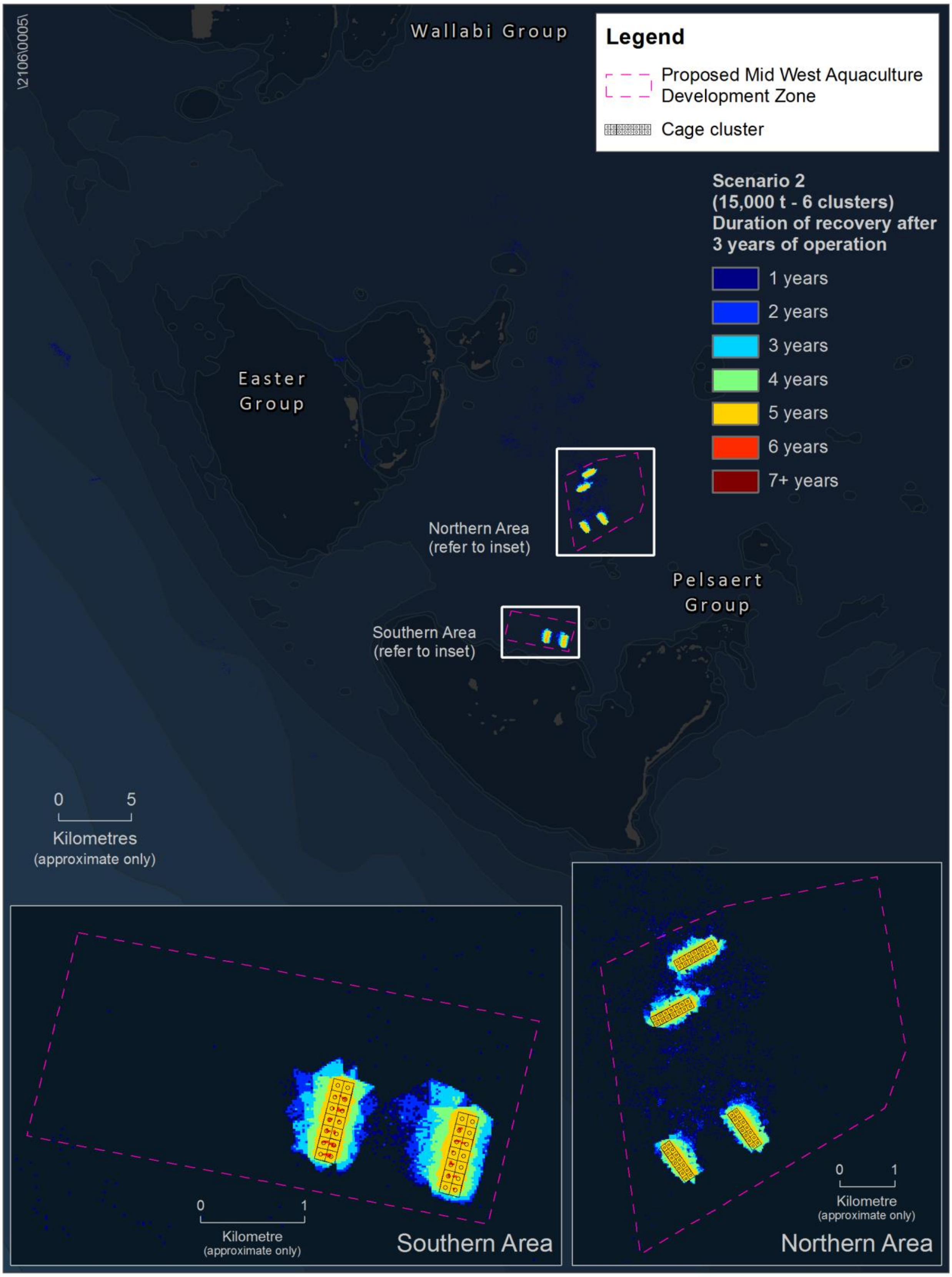


Figure 7-24 Duration of recovery under scenario 2 (15 000 t) after 3 years of operation

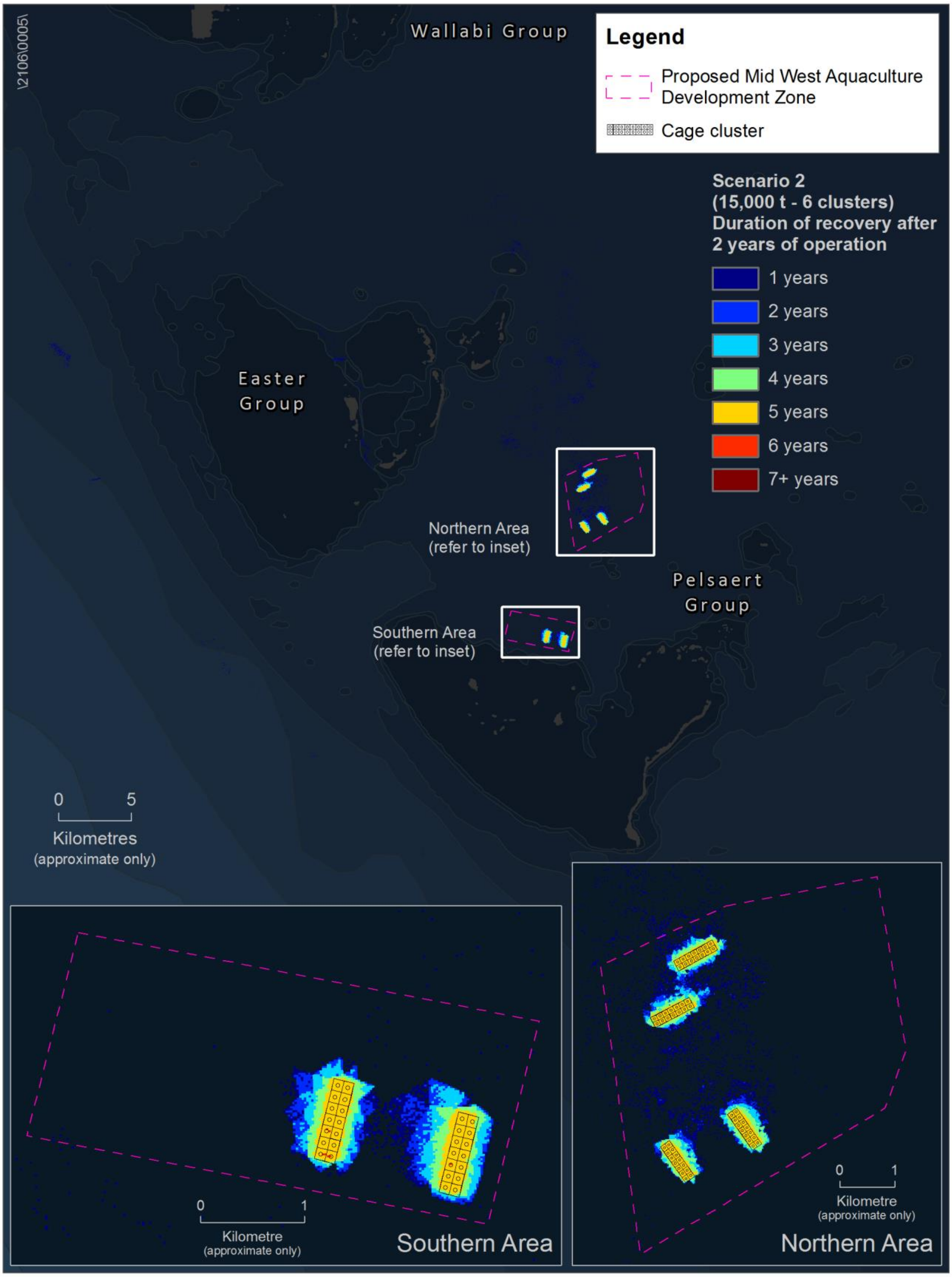


Figure 7-25 Duration of recovery under scenario 2 (15 000 t) after 2 years of operation

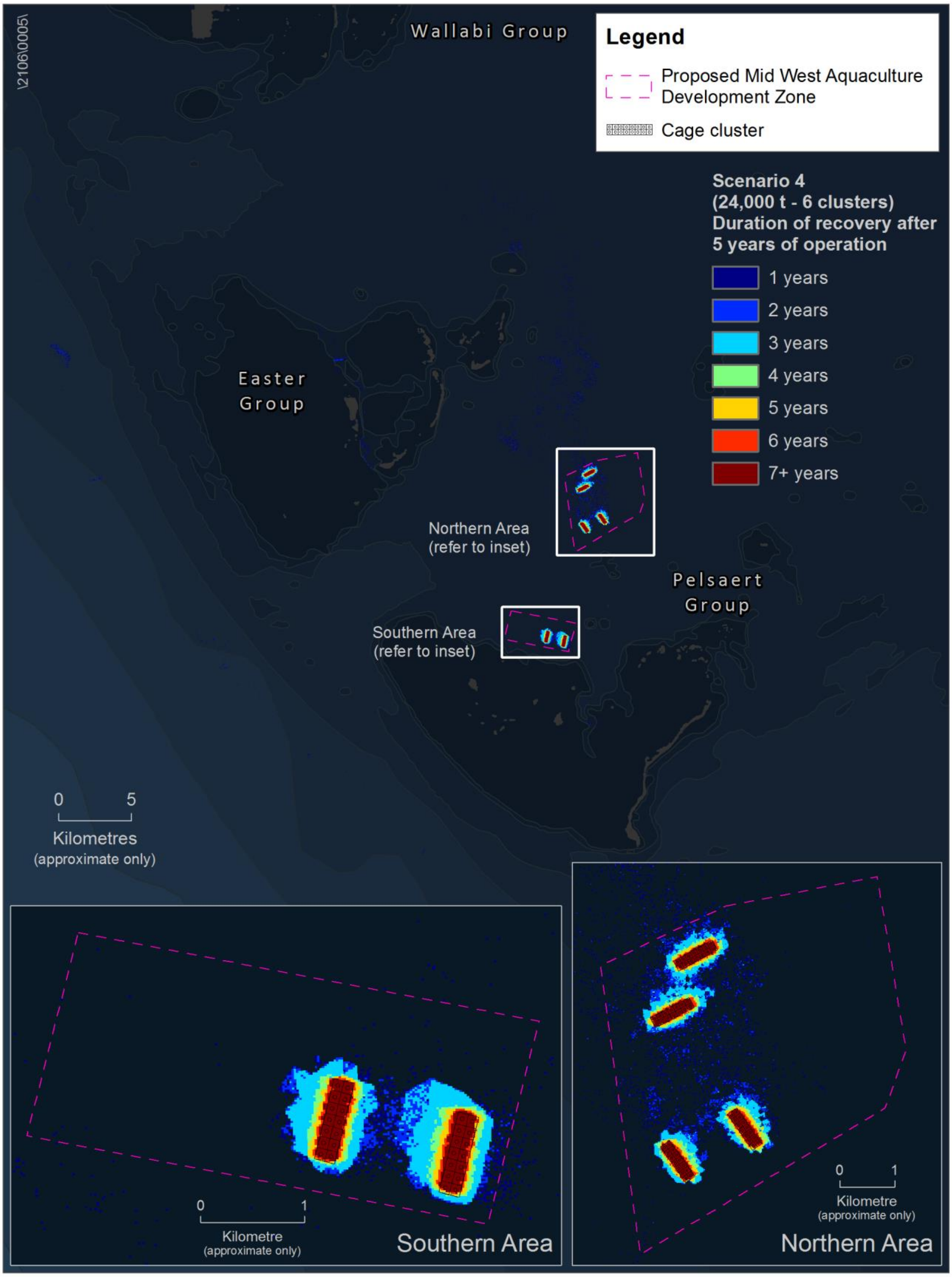


Figure 7-26 Duration of recovery under scenario 4 (24 000 t) after 5 years of operation

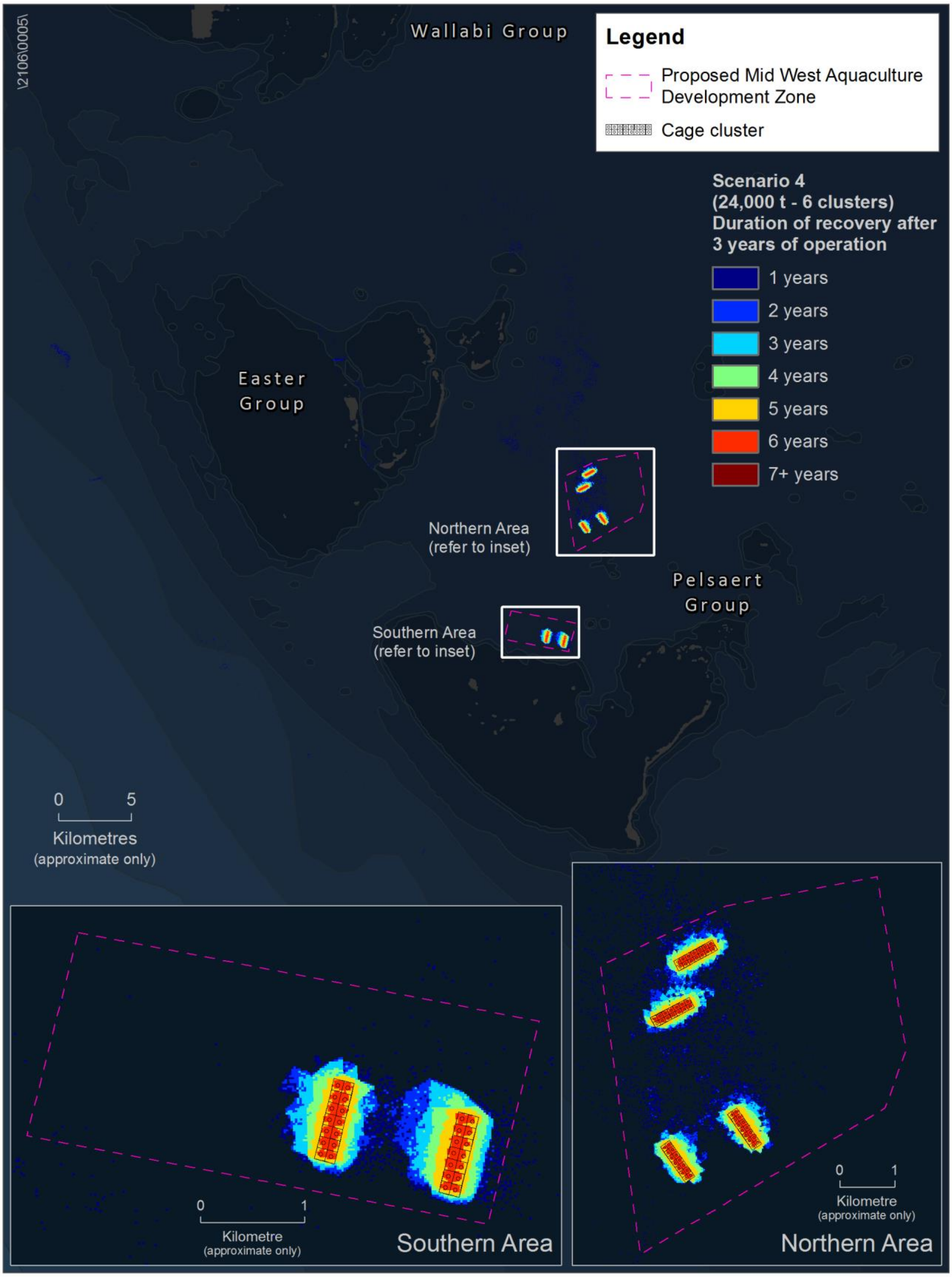


Figure 7-27 Duration of recovery under scenario 4 (24 000 t) after 3 years of operation

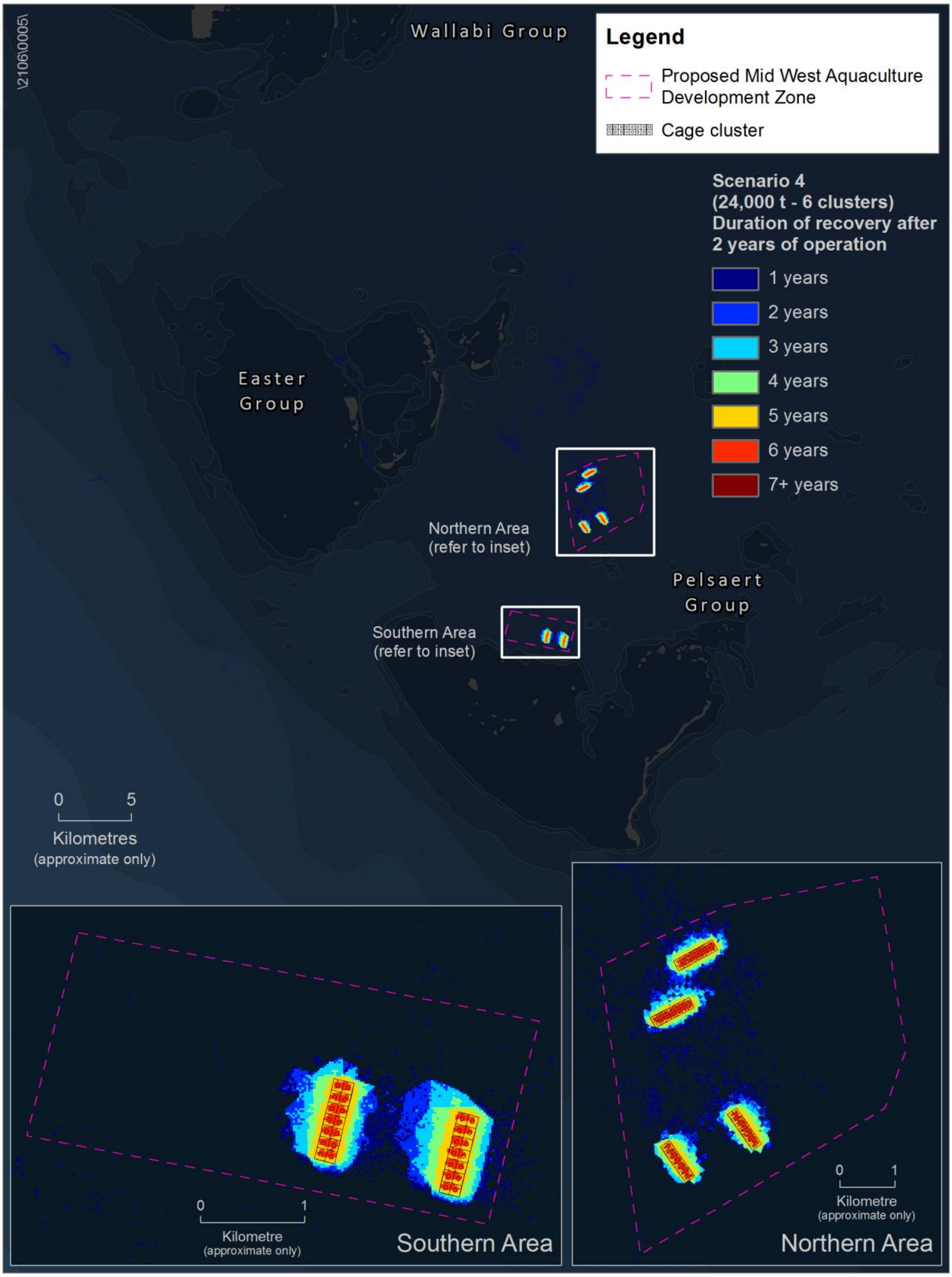


Figure 7-28 Duration of recovery under scenario 4 (24 000 t) after 2 years of operation

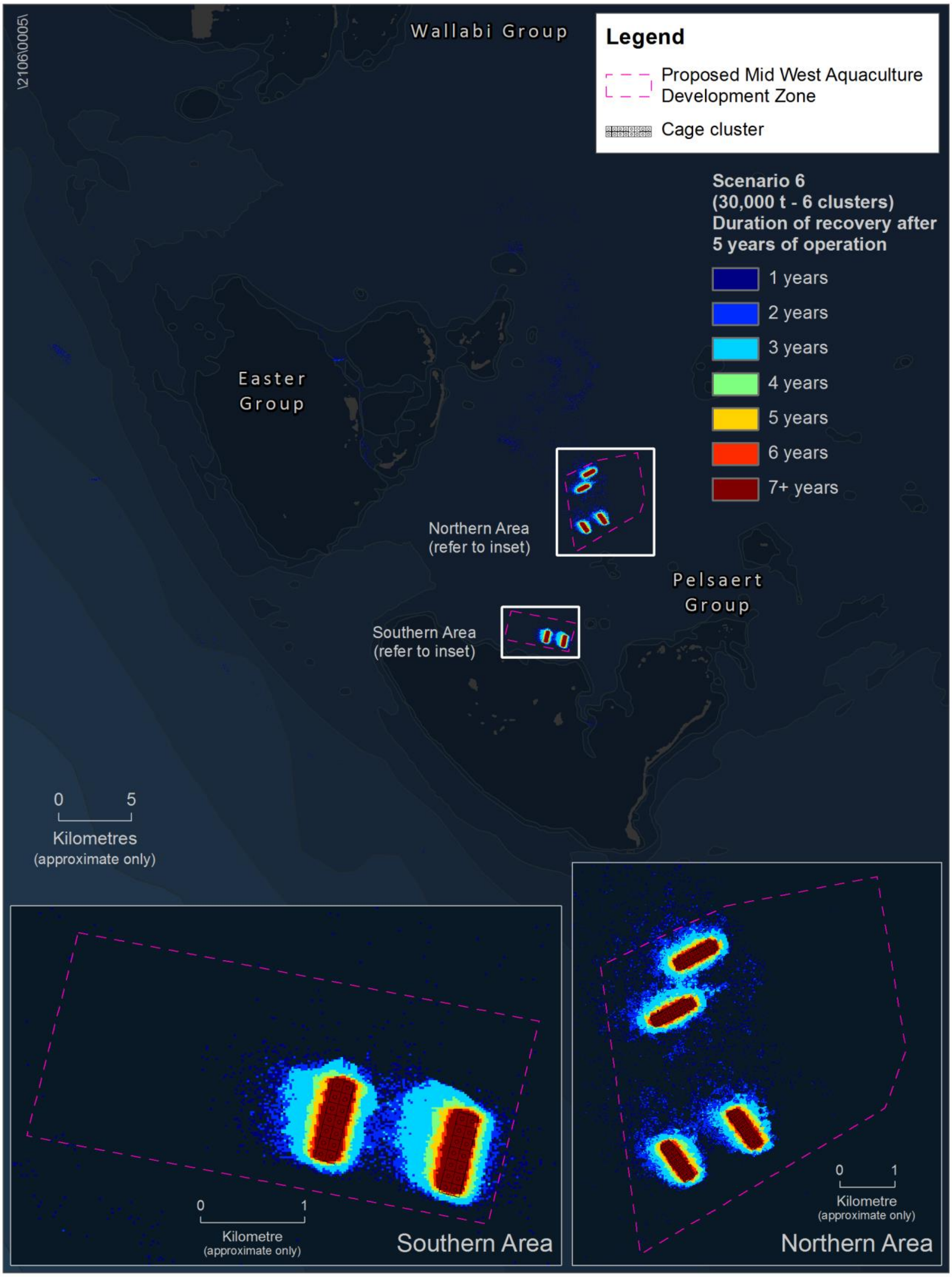


Figure 7-29 Duration of recovery under scenario 6 (30 000 t) after 5 years of operation

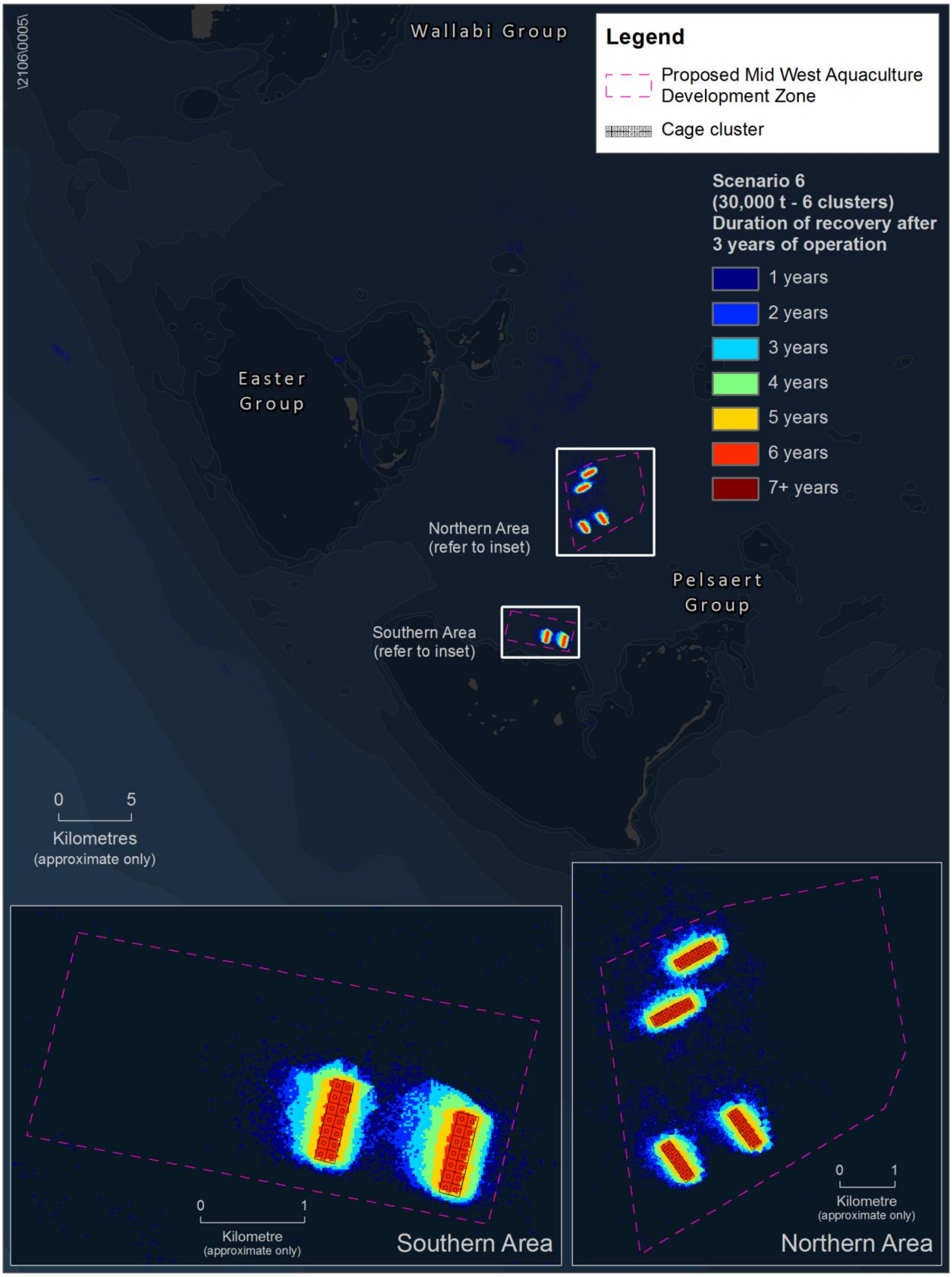


Figure 7-30 Duration of recovery under scenario 6 (30 000 t) after 3 years of operation

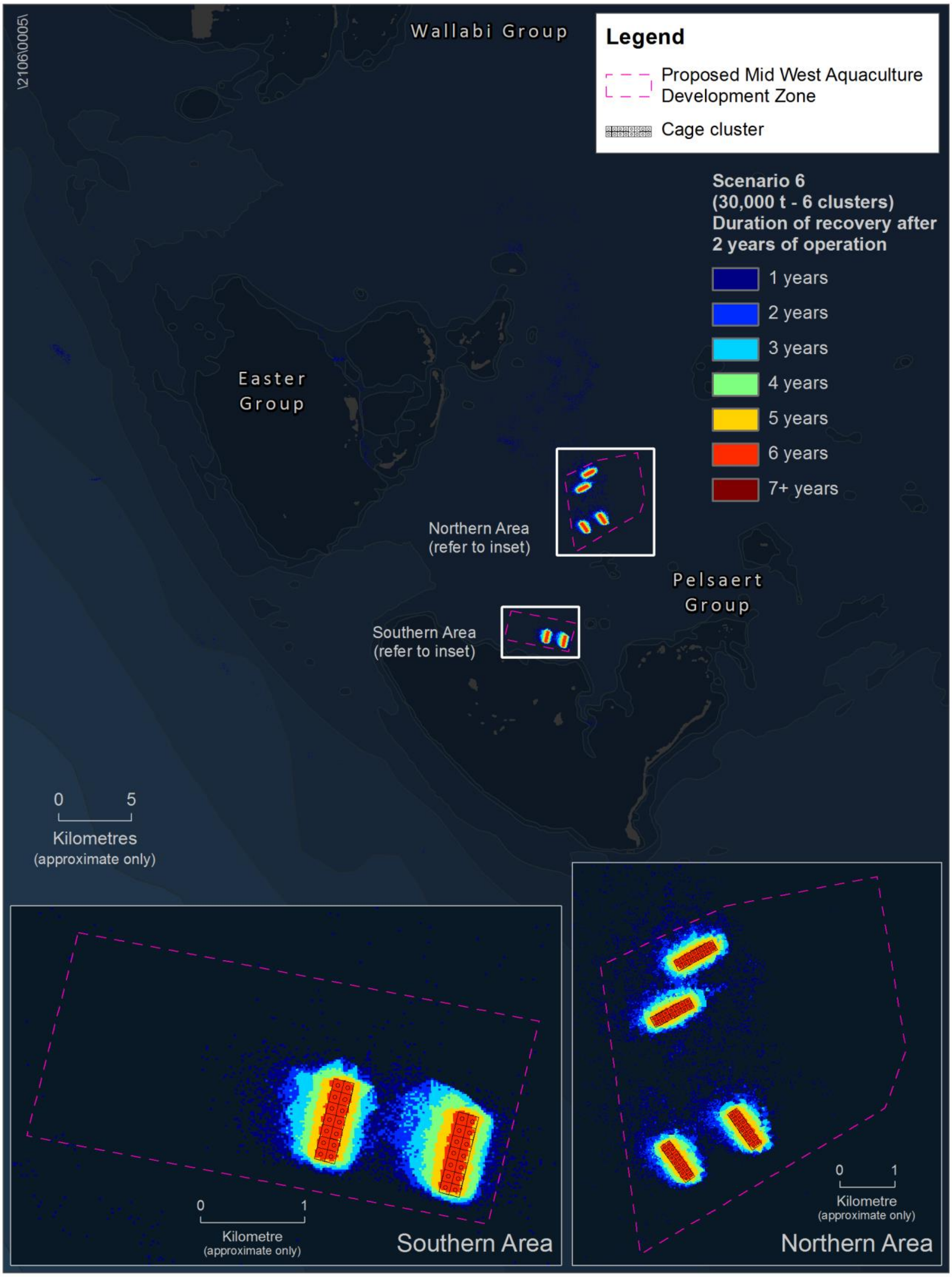


Figure 7-31 Duration of recovery under scenario 6 (30 000 t) after 2 years of operation

Comments on the zone of influence

The spatial extent of the ZoI, and particularly its outer limits of distribution, was driven largely by the dispersion of the smallest faecal fraction (see Section 4.6.1). The extremities of its distribution in the north, the south-west, and particularly in the deeper lagoonal areas of the Easter Group, are an artefact of the modelling. Particles may travel this distance from the cages through resuspension, but they are unlikely to accumulate in the densities shown in the Figures because the model understates dispersive processes at very low deposition rates.

The model does not simulate every single particle released during operations, as to do so would exceed hardware limits such as memory and disk space. Instead, multiple particles are packaged together in a single discrete unit of 10kg, which means that the lowest deposition rate that can be resolved is 10 kg/year. This 'package' will have all the physical characteristics of the particles it is representing (e.g. settling velocities, resuspension dynamics, density) but using it greatly reduces computational overhead. At high deposition rates (e.g. in the vicinity of cages), packaging particles in this manner will not change overall model results, but in areas with low deposition rates (e.g. the lagoonal area of the Easter Group) deposition will be overstated if only a few packages are deposited at the same location.

The accumulations of FOM in the lagoon of the Easter Group (Figures 6.1–6.4) were due in part to the deeper water in this area (leading to reduced wave-driven bed shear stress and, hence, resuspension), but also due to the packaging of particles for modelling purposes as noted above (see also Section 4.6.1). This was only an issue where smaller numbers of particles were involved and the model predicted the spatial extent of the ZoI nearer to the cage clusters much more precisely. The higher precision in this case was driven by the higher number of particles near to the cages compared to the extremities of the zone. The Monte Carlo approach used to predict particle transport is more precise when dealing with large numbers of particles.

Comments on the modelled rate of chemical remediation

Rates of organic matter mineralisation are site-specific and depend, among other things, on the assimilative capacity of the system (Findlay et al. 1995). A review by Brooks et al. (2003) found that biological remediation times varied significantly from a few months to several years (Mahnken 1993, Morrisey et al. 2000, Karakassis et al. 1999). Recovery typically proceeded rapidly in the months directly after fallowing but often slowed as time progressed, presumably because the recolonisation rates of infauna differ (e.g. Mahnken 1993).

Brooks et al. (2004) examined recovery in sediments after >2000 t of salmon were harvested and the cages left to fallow. At peak farming biomass, benthic sediments at the study site were black in colour and characterised by bubbles of hydrogen sulphide and beds of the sulphide-oxidising bacterium *Beggiatoa* spp, with the effects extending between 18 and 145 m down-current of the sea-cage perimeter. In this worst-case scenario, and following four years of fallowing, biological remediation was nearing completion at distances >80 m from the sea-cages but was not complete within this distance. Within 80 m, it was predicted that that chemical remediation sufficient to support half of the common taxa observed at reference sites would be complete 5.4 years post-fallowing, with complete biological remediation requiring a longer period.

The observations described in Brooks et al. (2004) validate in part the recovery times reported here, in which it was predicted that between 6 and 7+ years would be required for sediments directly beneath the sea-cages to achieve chemical remediation (see above). The longer periods of recovery reported in this assessment are perhaps not surprising given the levels of standing biomass examined (between 2600 and 5000 t of finfish per cage-cluster), and the fact that we adopted a highly conservative approach for estimating the volumes of fish waste (see Section 4.6.1).

Variability in the timing of recovery is widely reported in the literature: Macleod et al. (2002) reported chemical remediation after two years (with sulphide levels returning to reference levels), but incomplete biological remediation (infauna were in a transitionary recovery phase, and still significantly different from reference sites). Subsequent work by these authors (Macleod et al. 2006) found that sediment returned to pre-stocking conditions after a three-month period, but did not return to reference conditions. Despite similarities in the way the impact sites were treated in these studies (i.e. stocking levels and feed inputs), there were differences in the recovery response and in the rate of change in infauna community structure. This implies that the link between sediment organic load and recovery is not straightforward and that different locations may need different management strategies, particularly with regard to timing of fallowing (Macleod et al. 2006).

As indicated in Section 4.6.2, rates of chemical remediation as predicted by the sediment diagenesis model were assumed to proceed free of major physical disturbances. Although the model incorporated some capacity for bio-physical disturbance and biological reoxygenation via biodiffusion and irrigation (based on a constant of 20 m^2/y to a depth of 15 cm), neither of these processes accounted for the potential ‘resetting’ of the sediment during major scour events i.e. such as those which may occur during major storm events or cyclones, the latter of which affects the MWADZ approximately every 2.5 years. The recovery times presented herein are therefore conservative and longer than those which may occur in reality, especially if the 5-7 year recovery period modelled in this assessment was affected by a significant storm event.

7.3.3 Metals

The sediment diagenesis model was also used to determine the time taken for sediments to recover following inputs of waste, including trace elements (Zn and Cu). Triggers were set following the EPAs EQG for high ecological protection (EPA 2014). Although present in commercial feeds and therefore also present in fish faeces, the low molar ratios of Zn and Cu in the fish waste were insufficient to result in sediment concentrations in excess of the EQG, even after five years production at the upper end of the scenarios modelled (S6).

7.4 Mixed assemblages / Water column

7.4.1 Dissolved oxygen

The potential for deoxygenation of the water column beneath and near the sea-cages was investigated using the integrated hydrodynamic, water and sediment diagenesis model. Simulations focused on the bottom half of the water column, which for the project area ranged between 12–25 m and 25–50 m depth. Simulations also included deeper areas (>50 m depth) to the west of the MWADZ, including the leading edge of continental shelf slope. Median dissolved oxygen concentrations at the edge of the continental shelf were lower than the 80th percentile of background concentrations. Oxygen concentrations in the MWADZ maintained normal levels across the scenarios, with no evidence of significant oxygen drawdown even at peak standing biomass (i.e. S6). Results of the sediment diagenesis model, however, point to high levels of biological oxygen demand (BOD) at the sediment water interface (Appendix G). Under the anoxic sediment conditions predicted by the model, waters at the sediment water interface (and in some cases, the layers above the sediment water interface) are likely to experience some oxygen drawdown. However, the extent of water movement through the system is such that the level of drawdown is unlikely to be of any ecological consequence, as oxygen levels are quickly resupplied by new seawater inputs.

7.4.2 Suspended particles

Sea-cage aquaculture produces volumes of organic wastes which when expelled from the sea-cages, settle to the sea-floor. A proportion of these wastes retain potential for resuspension, creating potential for mechanical interference to filter feeding processes. The potential for suspended particles to exceed the thresholds in Table 4.14 was investigated using the hydrodynamic model coupled to the particle transport model (refer to Section 4.6.1).

Under the range of production scenarios (S1–S6) simulated by the model, none produced TSS is concentrations high enough, or over sufficient durations of time (i.e. 50% given the criteria are based on the median value in time) to exceed the thresholds in Table 4.14. Under these thresholds, the EPAs criteria for moderate and high levels of ecological protection were met. However subsequent contextual investigations using a higher time threshold (i.e. 95%) revealed potential for short-term exceedances (5% of the time) of both the high and moderate protection criteria, but only in the northern area. Hence, although there was potential for TSS concentrations in the MWADZ to reach levels higher than background on occasion, the duration and level of exceedance was not sufficient to exceed the published major impact thresholds for filter feeding communities (PIANC 2010).

7.4.3 Smothering

Anecdotal observations, and the results of modelling presented here, suggest that the majority of aquaculture waste settles to the sea-floor immediately beneath the sea-cages (Section 7.3.1). Under conditions of low shear stress, some of this material may accumulate, leading to smothering of resident benthic communities.

The potential for impacts from smothering was investigated using the hydrodynamic model coupled to the particle transport model (refer to Section 4.6.1) and was assessed using thresholds developed for corals (PIANC 2010) (Table 4.11). Corals were chosen because they exhibit poor tolerance to sedimentation relative to other invertebrates (Oceanica 2013), thus providing for a conservative assessment. Rates of sediment deposition were calculated on a square meter basis over a 12 month period, and averaged over a 365 day period. Because modelling assumed constant rates and volumes of deposition, changes related only to variation in current speed (as captured by the hydrodynamic model).

Modelling indicated potential for exceedances of both the minor and moderate impact categories, but there were no exceedances of the major impact category (Table 4.12). Moderate impacts were restricted to S6, and were confined to very small areas immediately under the sea-cages (Figure 7-33). Minor impacts were more prevalent, and were recorded in S5 and S6 (Figure 7-32 and Figure 7-33). The zone of minor impact although proportionally larger than the zone of moderate impact, was nevertheless predicted to be confined to area of seafloor corresponding to the outer boundary of the sea-cage structures.

Under the PIANC (2010) criteria, areas of the seafloor subjected to exceedances of the minor impact criteria could be expected to result in localised mortalities of coral, but not at a spatial scale expected to flow on to more serious secondary consequences. Under the same criteria, areas subjected to exceedances of the moderate impact criteria, could result in locally significant mortalities. From the results, both the zones of minor and moderate impact were predicted to be restricted to area occupied by the sea-cages. While no significant corals reefs were observed in the MWADZ (Section 5.4.5), the potential for impact to sensitive filter feeding communities under the sea-cages should be considered during placement.

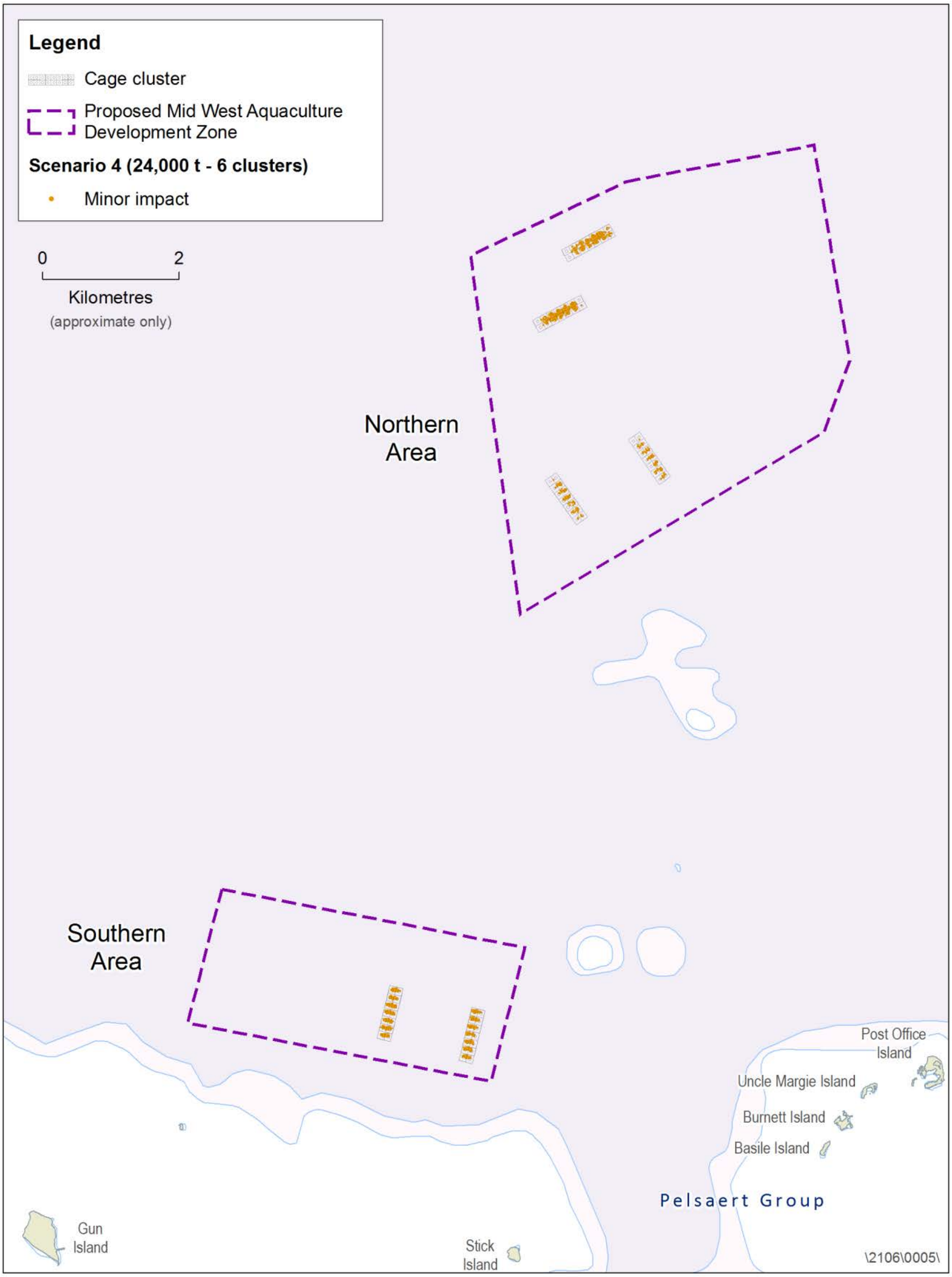


Figure 7-32 Zones of impact based on the rate of material deposition under scenario 4 (24 000 t)

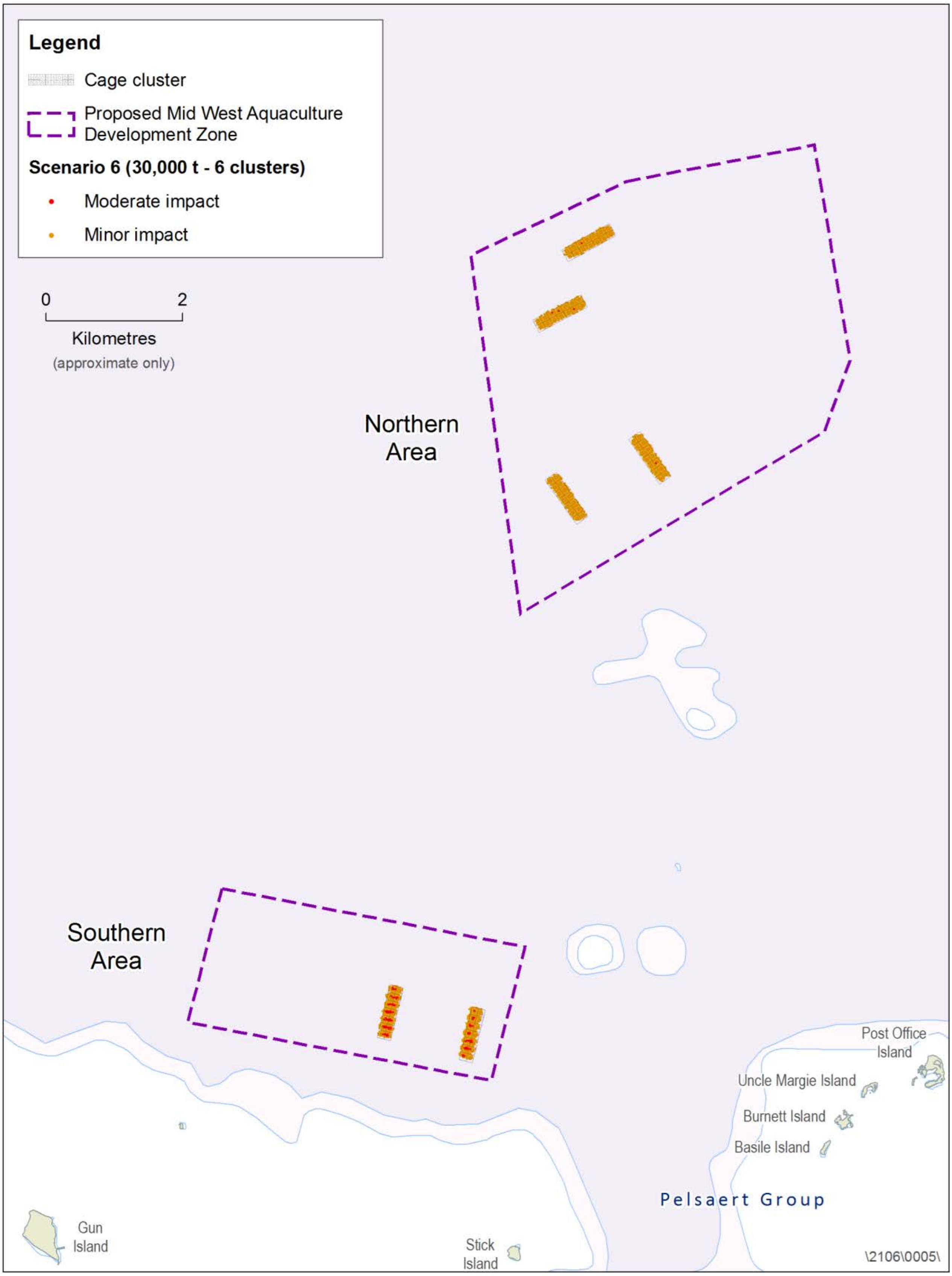


Figure 7-33 Zones of impact based on the rate of material deposition under scenario 6 (30 000 t)

7.4.4 Light intensity

Sea-cage aquaculture has the potential to lead to increased light attenuation (at the benthic level) via a number of cause effect pathways: typically via increases in suspended particles and/or increases in phytoplankton biomass. The potential for light intensity reduction in the bottom strata of the water column was investigated using the coupled TUFLOW FV - AED hydrodynamic and water quality model. The potential for impacts was investigated in the context of the thresholds listed in Table 4.14.

Reductions in PAR of ~15% and ~4% were respectively observed immediately under the sea-cages and to a distance of 100 m from the sea-cage perimeter. However, under the range of production scenarios (S1–S6) simulated by the model, none produced conditions sufficient to reduce PAR to levels exceeding the moderate and high protection thresholds in Table 4.14. The observed reductions in PAR near the sea-cages were the combined result of shading of the sea-cage infrastructure, and the shading effect of suspended particles (fish wastes). None of the observed declined in PAR resulted from increases in phytoplankton. The response of phytoplankton to the varying inputs of nitrogen, as simulated across the range of scenarios, is discussed further in Section 7.4.6.

7.4.5 Algal growth potential (DIN)

The spatial extent and concentration of DIN released from sea-cage infrastructure was investigated under the higher range of production scenarios (S6-S4); Section 4.5.4). Concentrations of DIN near the sea-cages increased with increasing biomass, and increasing stocking density. Scenario S6 produced the highest concentrations and the largest DIN 'footprint', while scenario S4 produced lower DIN concentrations and a smallest environmental 'footprint' (Figure 7-34 and Figure 7-35). The decrease in DIN with distance was driven partly by far-field dilution processes and partly by biological assimilation, both processes simulated in the CANDI-AED-model.

For the purposes of defining zones of impact, acute thresholds were developed following the criteria for high and moderate levels of ecological protection, under which large and moderate changes to ecosystem health, respectively, could be expected (Section 4.5.2). Concentrations of DIN in and immediately adjacent to the sea-cage structures exceeded the moderate ecological protection criterion (95th percentile of background) in both scenarios (S4 and S6), though the areas occupied by this zone were small, and typically restricted to within 150 m of the sea-cage perimeter. The spatial extent of the area exceeding the high protection criterion (80th percentile of background) was more extensive, but varied markedly depending on the scenario, and the position of sea-cages within the zone. The area exceeding the high protection criterion was greater in the northern MWADZ, where the stronger currents acted to carry the plume farther and more rapidly.

Although the area exceeding the moderate protection criteria was small and restricted to the MWADZ, the area exceeding the high protection criteria encroached (and in some cases breached) the boundaries of the northern MWADZ. This was most pronounced in S6 (Figure 7-34), but was mitigated in S4 by reducing the stocking density (Figure 7-35). The area exceeding the combined moderate and high protection criteria represents the area not expected to meet a high level of ecological protection, and highlights the potential for algal growth. The extent to which the simulated elevations in DIN translated to algal growth were examined using the water quality model packages (Section 7.4.6).

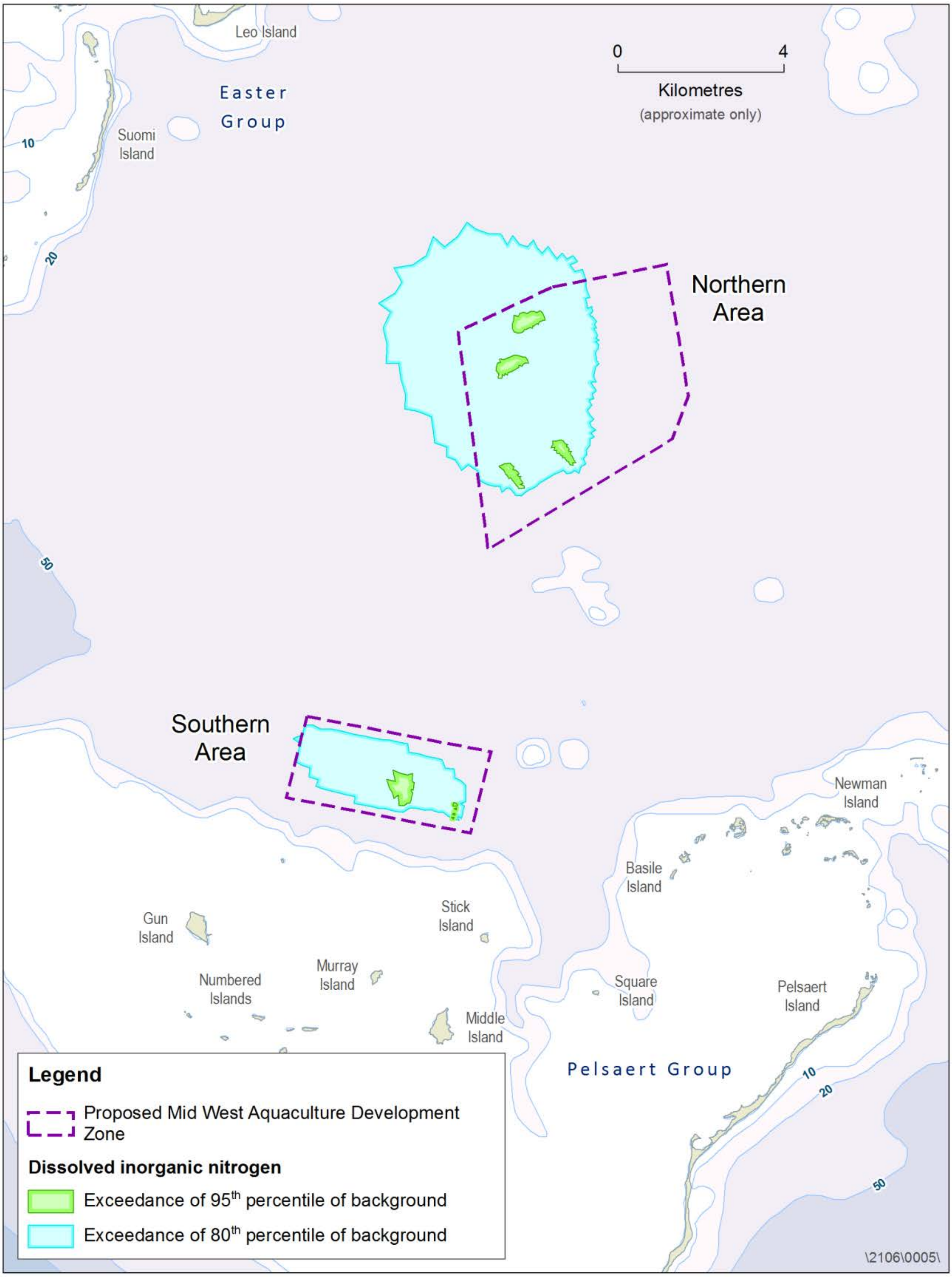


Figure 7-34 Zones of impact based on dissolved inorganic nitrogen in the water column under scenario 6

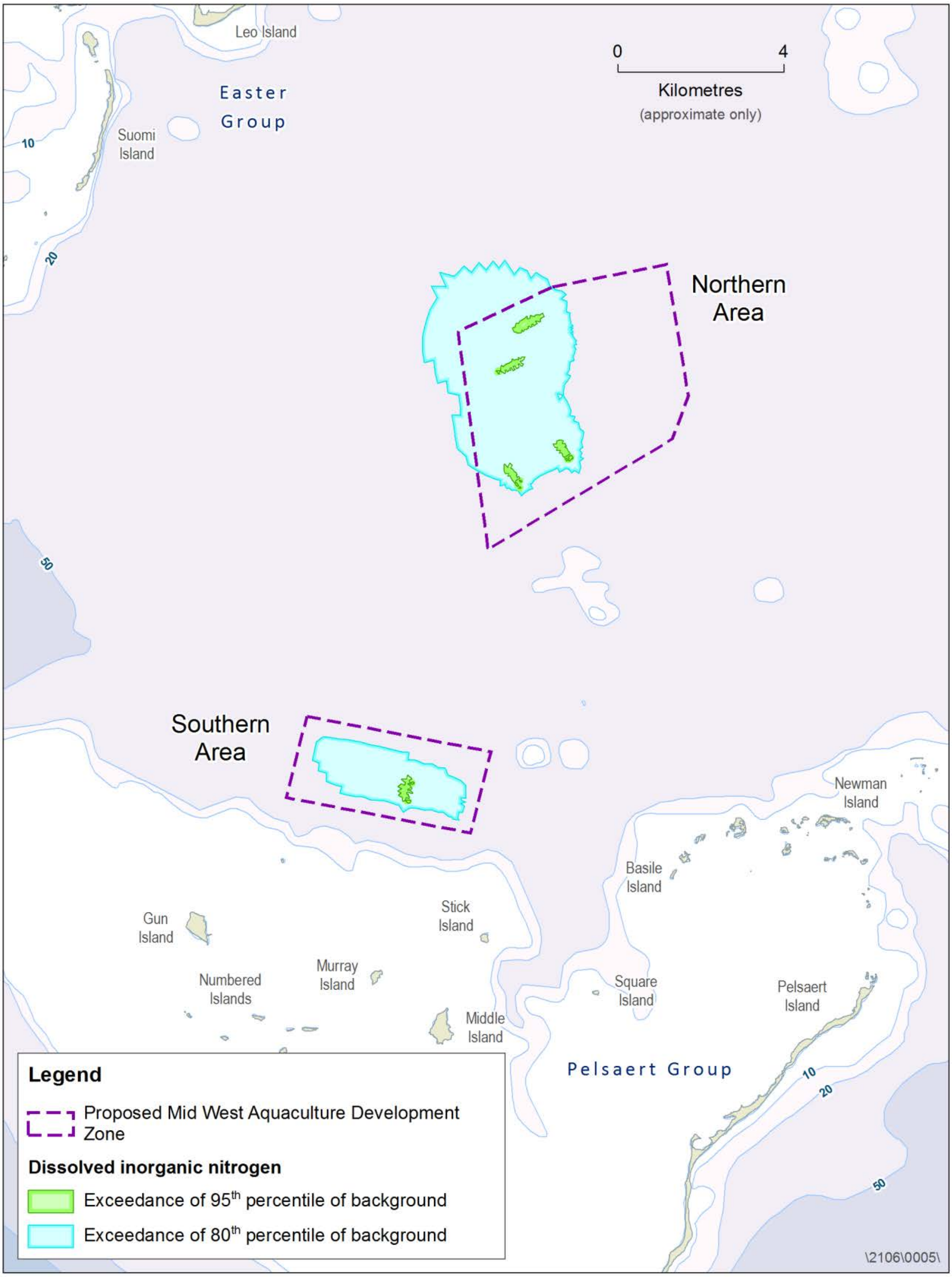


Figure 7-35 Zones of impact based on dissolved inorganic nitrogen in the water column under scenario 4

7.4.6 Nutrient enrichment (chlorophyll-a)

Despite significant inputs of DIN, there were no discernible increases in chlorophyll-a (the proxy for phytoplankton biomass) that could be attributed to sea-cage production, and no exceedances of the moderate and/or high ecological protection criteria in the waters surrounding the MWADZ. A natural gradient of chlorophyll-a was detected between deep waters of the MWADZ and shallow waters of the mainland. Chlorophyll-a in coastal waters sustained concentrations higher than the 95th percentile of background oceanic conditions, even when simulated under baseline conditions, confirming the observed pattern was not a result of aquaculture activities.

The results achieved via simulation are perhaps not surprising given the volume and level of water movement through the project area. Inputs of DIN for scenarios S1-S2 are roughly equivalent to the annual total DIN inputs to Perth's coastal waters via three widely separated ocean outfalls (BMT Oceanica 2015c). Perth's coastal waters, like those of the project area, are oligotrophic and well flushed (but differ in that that they are shallower; 10–20 m depth). Over ten years of intense summer water quality monitoring near these outfalls has failed to detect long-lasting increases in chlorophyll-a due to these regular DIN inputs. Where chlorophyll-a increases have been detected, they have only persisted for a short time (days) and were typically associated with extended periods of low wind (Oceanica, unpublished data). Scenarios S3–S6, although contributing DIN in higher volumes than those contributed to Perth's coastal waters by the ocean outfalls, are indicative of the very high assimilative capacity of the water within the project area, an attribute which is likely enhanced by the depth of the water column (and associated large receiving volume).

8. Impact Assessment – Supported by Literature

8.1 Threatened, endangered and protected finfish

8.1.1 Approach

The potential for adverse interactions between finfish populations and the proposed MWADZ was investigated via two desktop assessments: one focussing on potential impacts to the sustainability of threatened, endangered and protected fish species (sharks and rays) (this section) and the other focussing on potential impacts to invertebrate and finfish species and fisheries (Section 8.2). Section 8.1 provides a summary of the key risks presented by the proposal to the sustainability of threatened, endangered and protected fish populations, focussing particularly on sharks. Text included in this section is excerpted from DoF (2015a). Full details are provided in Appendix B.

8.1.2 Potential adverse interactions

Threatened, endangered and protected fin- fish with potential to be adversely affected by the proposal are outlined in Table 8.1. Although all of these species may be affected by the proposal, locally relevant data for the majority of the species listed in Table 8.1 is scarce. The review was therefore centred on species for which there was available information. The review hence focused on the white shark, grey nurse shark, tiger shark and whale shark.

Table 8.1 Threatened, endangered and protected species of fish potentially affected by the MWADZ proposal

Common name	Family	Species
White shark	Lamnidae	*Carcharodon carcharias*
Shortfin mako		*Isurus oxyinchus*
Longfin mako		*Isurus paucus*
Grey nurse shark	Odontaspididae	*Carcharias Taurus*
Tiger shark[1]	Sphyrnidae	*Galeocerdo cuvier*
Smooth hammerhead		*Sphyrna zygaena*
Scalloped hammerhead		*Sphyrna lewini*
Great hammerhead		*Sphyrna mokarran*
Green sawfish	Pristiophoridae	*Pritis zijsron*
Whale shark	Rhincodontidae	*Rhincodon typus*
Manta ray	Mobulidae	*Manta birostris*

Note:

1. Tiger sharks are not considered threatened, endangered or protected; however, as an iconic species it was included in this assessment.
2. Blue highlighted sections pertain to taxa considered representative of the broader threatened, endangered and protected shark and ray species, and the taxa included in the assessment

Sea-cage farming may adversely affect threatened, endangered and protected species through interactions with the aquaculture related activities (mainly feeding) and infrastructure (sea-cages, vessels). Organic wastes, including fish faeces and feeds, are predicted to exit the cages and accumulate immediately under and adjacent to sea-cages (Section 7.3). Aquaculture waste products in particular are likely to attract smaller fish, which in turn may attract larger predatory species, including sharks.

The key cause cause-effect-response pathways identified in the risk assessment (Appendix B) are summarised in Figure 8-1. Risks are considered particularly in the context of the potential for sea-cage aquaculture to act as an attractant, leading to secondary changes in the behaviour and abundance of threatened, endangered and protected species.

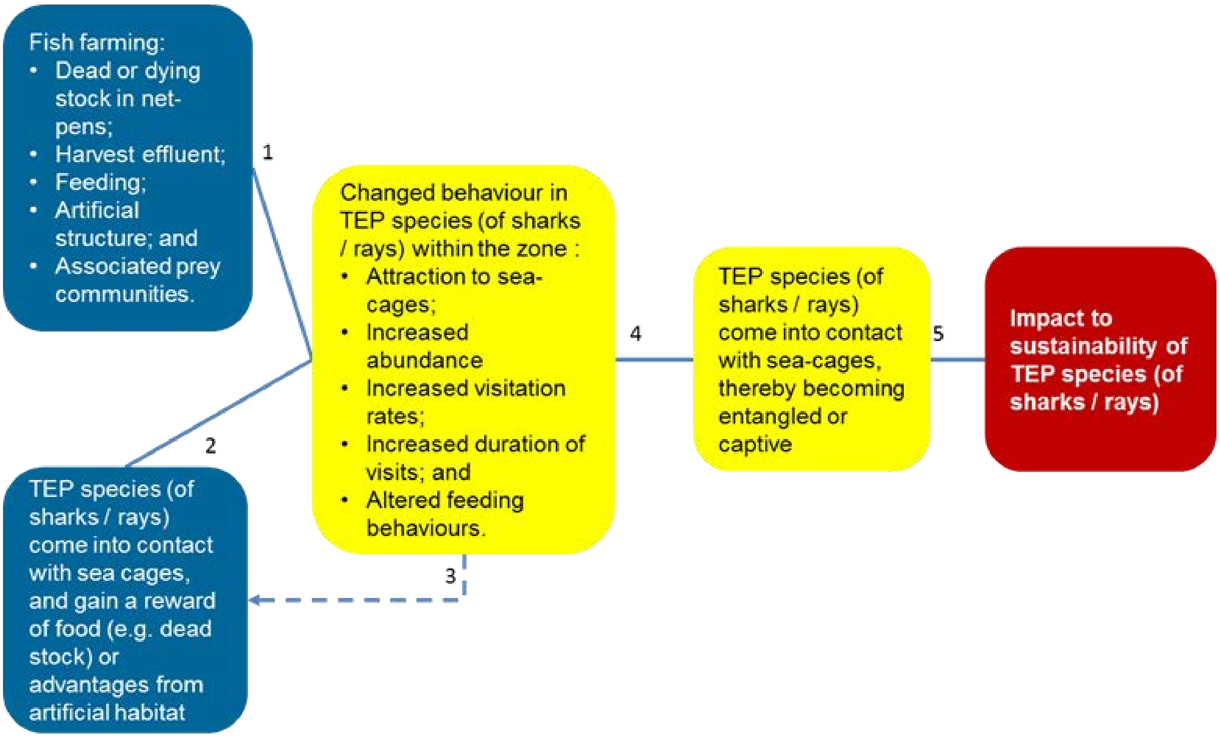


Figure 8-1 **Conceptual model of hazards associated with aquaculture and the potential cause-effect pathways which could affect the sustainability of threatened, endangered or protected species of finfish**

8.1.3 Possible behavioural responses

Significant populations of sharks currently reside in and in close proximity to the MWADZ (Appendix B). Sea-cages are likely to attract threatened, endangered and protected fish species, leading to localised changes in population structure. Key attractants include: live and dead (or dying) finfish stock, availability of artificial feeds (both pellets and fish waste), harvest activities (blood in the water), and the artificial sea-cage structures themselves, which may serve as shelter, and artificial habitat.

Behavioural responses are likely to include attraction and higher rates of visitation. The increased presence of sharks and rays in the MWADZ is also likely to increase the probability of fauna interactions. Success in gaining provision (via feeding reward) is likely to exacerbate the issue, leading to repeat visitation and increased probability of adverse interactions. At a local scale, the increased presence of sharks in the MWADZ is likely to increase the potential for entanglement or capture.

8.1.4 Major findings and recommendations

Modern fish farms alone are unlikely to cause levels of mortality that will impact the sustainability of threatened, endangered and protected species of sharks or rays. However, fish farms could contribute, by way of a small number of deaths, to the total number of anthropogenic shark mortalities within the region. The review found that the probability of adverse impacts could be reduced (to 'minor') by eliminating, or reducing the probability of interactions, through best-practice mitigation and management strategies, as follows:

- Use of appropriate anti-predator netting materials
- Use of well-designed and durable sea-cages suited to the local environment
- Containment of all post-harvest blood water
- Prevention of food provision through regular removal of dead and moribund stock
- Regular inspections using submerged cameras to detect tears in the mesh
- Controlled feeding regimes and
- Compliance with the industry benchmark of less than 1% feed wastage.

The review indicated that the risk posed to threatened endangered and protected species is low and that the residual risks are manageable, provided the mitigation strategies listed in the bullet points above are implemented and followed for the life of the project. For the full assessment refer to Appendix B.

8.2 Invertebrate and finfish species and fisheries

8.2.1 Approach

Section 8.2 summarises the risks to invertebrate and finfish species and fisheries at the Abrolhos Islands, posed by the introduction of aquaculture sea-cages and associated activities. Text included in this section is excerpted from DoF (2015b). For the full assessment refer to Appendix C.

8.2.2 Potential adverse interactions

The potential for impacts to invertebrate and finfish species and fisheries was assessed via a comprehensive risk assessment. Following the identification of key threats and detailed analysis of hazard pathways leading to potential realisation of these threats, four overarching risks of most relevance to the activities proposed in association with the MWADZ were identified. These were:

- Aquaculture activity in the zone has a significant impact on the populations of invertebrate species (i.e. saucer scallop) in the Abrolhos Islands FHPA
- Aquaculture activity in the zone has a significant impact on the populations of finfish species in the Abrolhos Islands FHPA
- Aquaculture activity in the zone has a significant impact on the invertebrate fishery (i.e. Abrolhos Islands and Mid West Trawl Managed Fishery) and
- That aquaculture activity in the zone has a significant impact on finfish fisheries in the Abrolhos Islands FHPA

The first two risks are risks associated with potential ecological impacts on the species populations. By comparison, the last two risks are risks that essentially comprise the effects of the first two risks (i.e. the ecological impacts) in addition to the potential resource access impacts resulting from the physical presence of aquaculture infrastructure within the MWADZ.

All the above risks were assessed with a consideration of potential cumulative impact using the precautionary approach described in the methodology. This process investigated pathways or cause-effect linkages between hazards and key factors that contribute to a broad risk category.

Results from the risk assessment concluded that the proposal poses a negligible and acceptable risk to three of the four key risks identified. The MWADZ proposal is anticipated to generate negligible impacts on saucer scallop and finfish populations within the Abrolhos Islands FHPA. With respect to the Abrolhos Islands and Mid West Trawl Managed Fishery, the risk assessment identified that the MWADZ proposal poses a low risk, due to the potential to limit the amount of available fishing ground in the fishery.

The key cause-effect-response pathways considered in the review are summarised in Figure 8-2–Figure 8-6.

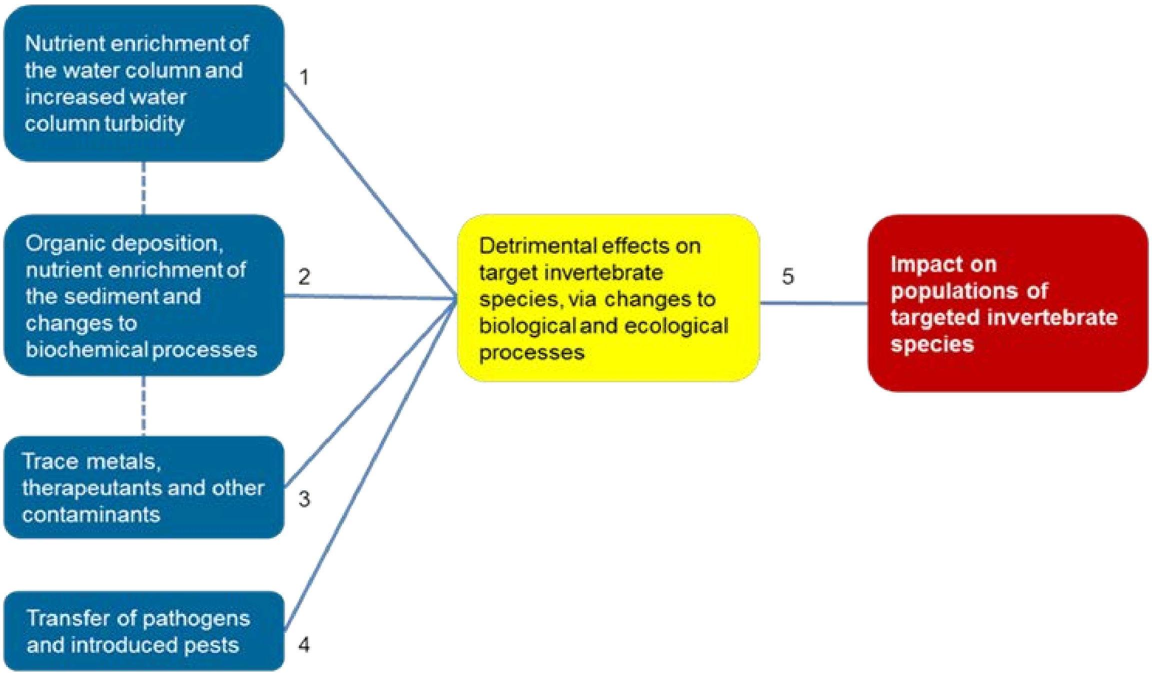


Figure 8-2 **Conceptual model illustrating potential cause-effect pathways of possible impacts from finfish aquaculture on invertebrate species populations**

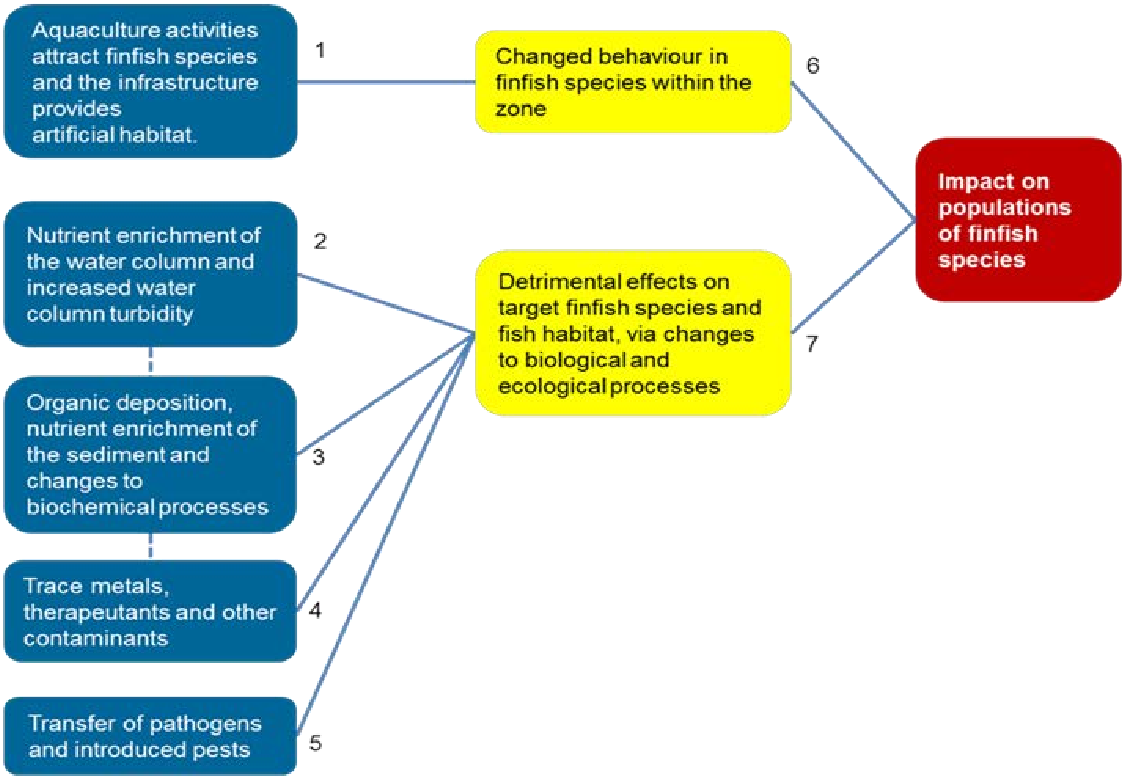


Figure 8-3 **Conceptual model illustrating potential cause-effect pathways of possible impacts from finfish aquaculture on wild finfish species populations**

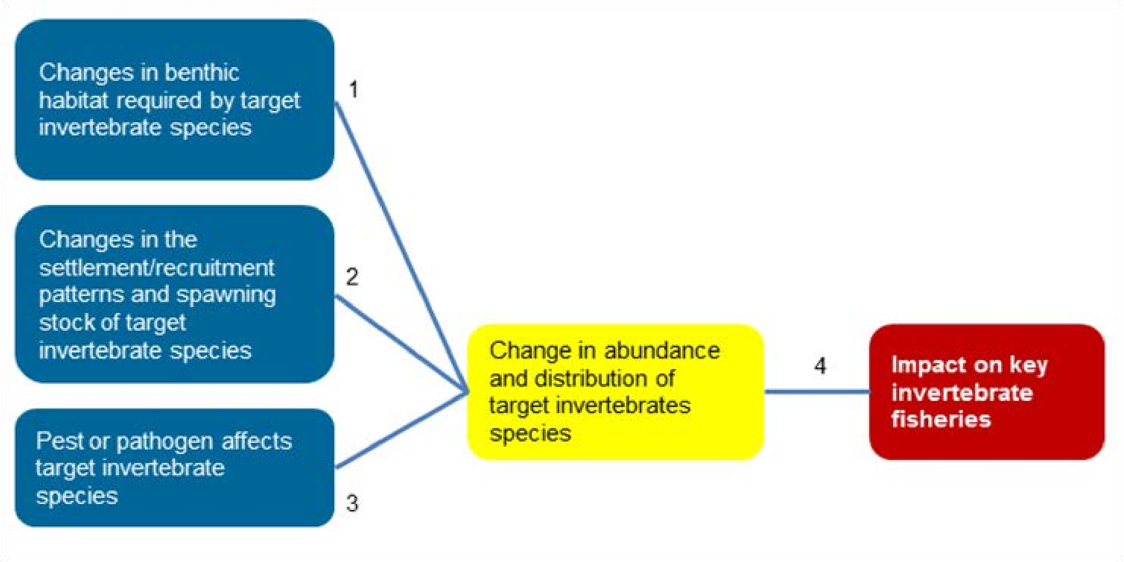


Figure 8-4 Conceptual model illustrating potential cause-effect pathways of possible impacts from finfish aquaculture on invertebrate fisheries

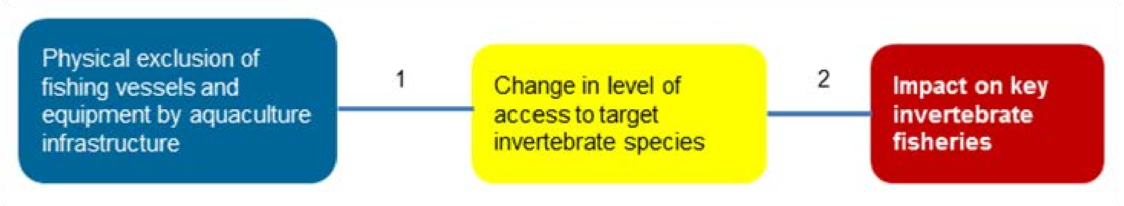


Figure 8-5 Conceptual model illustrating potential cause-effect pathways of possible resource access impacts from finfish aquaculture on invertebrate fisheries

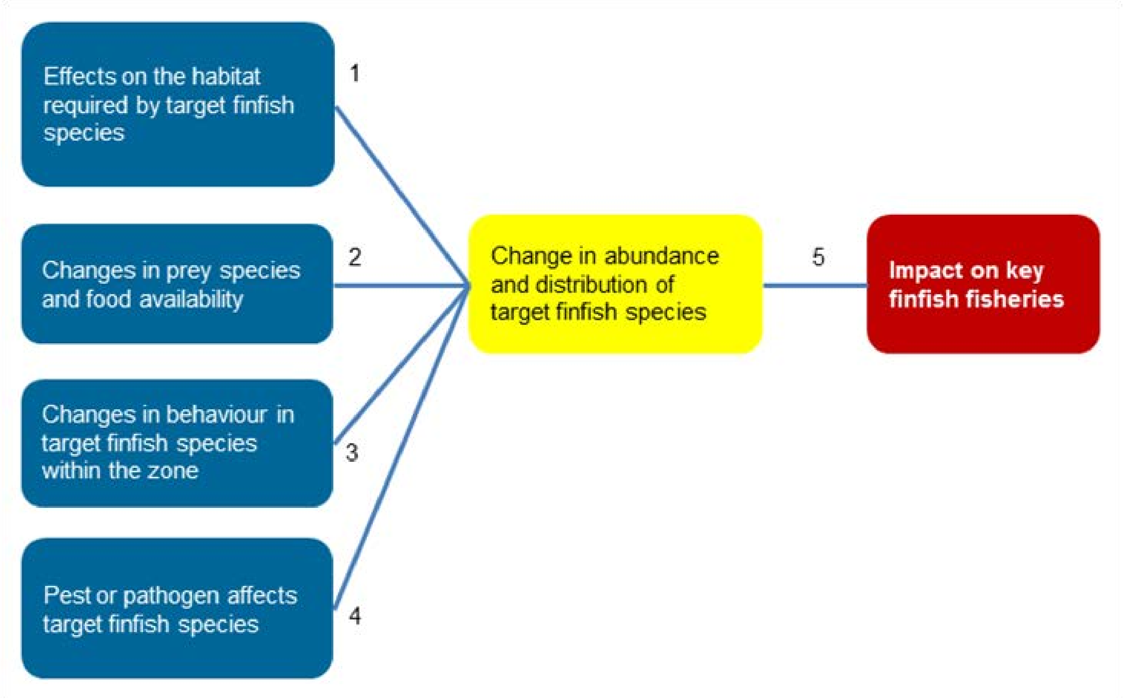


Figure 8-6 Conceptual model illustrating potential cause-effect pathways of possible ecological impacts from finfish aquaculture on finfish fisheries

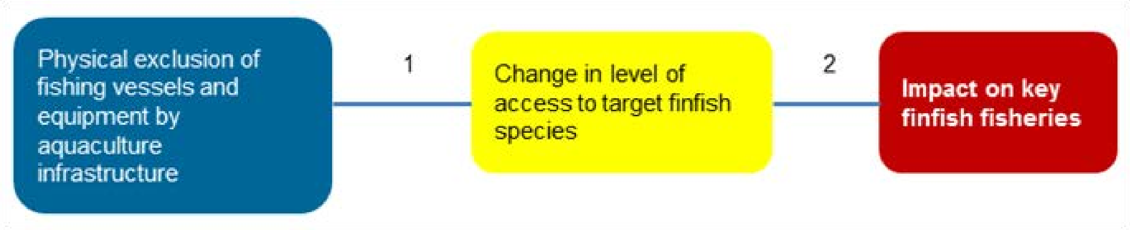


Figure 8-7 Conceptual model illustrating potential cause-effect pathways of possible resource access impacts from finfish aquaculture on finfish fisheries

8.2.3 Possible behavioural responses

Invertebrate populations

Impacts to benthic invertebrates are likely to be limited to very small areas beneath the sea-cages, where rates of organic matter deposition are predicted to be high, irrespective of the production scenario (Section 7.3). Modelled rates of organic matter deposition were considered in the context of the smothering thresholds listed in Table 4.11; Section 4.5.3. Results indicated that the minor and moderate level impacts would be confined to within the cage cluster boundaries (Section 7.4.3).

Under the sea-cages, invertebrates may be adversely affected by organic matter deposition, smothering, interruption to filter feeding processes and changes to sediment biochemical processes. In some circumstances, this may lead to avoidance behaviour in some target species, mortality of sensitive species and/or a change in species composition.

Invertebrate fisheries

Changes in sediment characteristics beneath the sea-cages may adversely affect the survivorship of settled invertebrate juveniles, including scallops. However, as predicted by the modelling (Section 7.4.3), impacts are expected to be limited to the area immediately under the sea-cages.

It is also expected that the presence of aquaculture infrastructure including, anchors, sea-cages and feeding systems may in some circumstances prevent access to potential scallop fishing grounds.

Finfish populations

Some finfish species are naturally attracted to artificial structures, and many are especially attracted to artificial food sources. Aquaculture feeds consist of fish meal and fish oil both of which are known finfish attractants (e.g. Machias et al 2005). It was considered that the combination of food sources and artificial shelters/habitats may attract finfish and alter the behaviour of certain finfish species, across a range of trophic levels. The following behavioural responses were considered likely:

- attraction to or avoidance of the farming area
- increased/decreased visitation rates
- increased duration of visits
- increased/decreased abundance and
- altered feeding behaviours

Finfish fisheries

The proposal may impact fish habitats for non-target species inhabiting sandy areas beneath and adjacent to the proposed sea-cages. However, any impacts are likely to be highly localised and typically restricted to within 110 m of the sea-cages (Section 7.3.2).

The proposal is also unlikely to significantly impact the habitats of target finfish species landed within the MWADZ, i.e. baldchin groper, snapper, West Australian dhufish, spangled emperor, coral trout and other demersal scalefish species. The area proposed for the MWADZ and the potential zone affected by inorganic and organic nutrient dispersal (Section 7.3), represents a very small component of the distribution of these species. As such, the proposed aquaculture activities are unlikely to have significant impact on finfish recruitment patterns and/or the spawning stock of finfish species.

Sea-cages are likely to aggregate some species of finfish and may potentially attract predatory fish including sharks and large pelagic species to the area. This may result in increased numbers of predatory fishes in the vicinity of cages that may be attractive to recreational and commercial fishers (e.g. mackerel, tuna etc.). However, it was considered unlikely that the proposal will lead to significant changes in the abundance and distribution of finfish species within the broader proposal area.

8.2.4 Major findings and recommendations

Invertebrate populations

The area expected to be affected by a decline in abundance of the target invertebrates is negligible relative to the natural range of the species considered (much less than 1 %).

Invertebrate fisheries

The MWADZ proposal is unlikely to cause significant adverse impact to habitats occupied by commercially targeted scallop species from the AIMWTMF. Any changes, if they occur at all, are expected to be localised and constrained within the footprint of the sea-cages.

The presence of physical aquaculture infrastructure requires a relatively small portion of the current fishing ground within the AIMWTMF. The physical presence of aquaculture infrastructure including fish cages, anchors and feeding systems will prevent fishing in the area where the cage clusters are located. Moreover, commercial fishers are likely to avoid areas within the MWADZ, given risks of entanglement.

Finfish populations

The review highlighted the need to reduce, wherever possible, the sources finfish attractants. The following mitigation and management measures were identified:

- removal of dead and moribund stock on a daily basis
- moderate stocking levels
- containment of all post-harvest blood water
- use of a high quality pellet feed
- controlled feeding regimes and
- compliance with the industry benchmark of less than 1% feed wastage

Finfish fisheries

The physical presence of aquaculture infrastructure including fish cages, anchors and feeding systems will prevent fishing in the area where the cage clusters are located. However, under the proposed management policy, the MWADZ will be non-exclusive, meaning commercial and recreational fishers will be permitted to fish the zone under the extent to which they are currently permitted, noting that the current extent of commercial line fishing in the proposal area is relatively minor.

8.3 Marine mammals and turtles

8.3.1 Approach

The potential for adverse interactions between the proposed MWADZ and regionally significant marine mammal and turtle populations was investigated via a comprehensive desktop assessment. Section 8.3 provides a summary of the assessment focussing on the species considered most at risk, the potential adverse effects of sea-cage aquaculture and the potential mitigation strategies that maybe used to reduce the risks to manageable levels. Text included in this section is excerpted from BMT Oceanica (2015b). For the full assessment, refer to Appendix A.

8.3.2 Potential adverse interactions

Thirty-one cetacean and two pinniped species may occur in or near the MWADZ. The species that are likely to be encountered include: the pygmy blue whale; humpback whale, Australian sea lion; Indo-Pacific bottlenose dolphin; and the common bottlenose dolphin. Species with a low likelihood of occurring include: the blue whale; southern right whale; Bryde's whale; killer whale; and the dugong.

Several aspects of the proposal have the potential to impart adverse effects to marine mammals and turtles, including: physical presence of the aquaculture sea-cages, vessel movements and artificial light. The physical presence of sea-cages may change natural feeding behaviours, cause serious injury or change the distribution and migration patterns. Vessel collisions may result in injury, harm or behavioural disturbance to marine fauna, and increased artificial light levels may disrupt or disorient marine turtles (BMT Oceanica 2015b).

The potential for impacts to marine mammals and turtles will be monitored and managed under the EMMP for this proposal, which is published separately (BMT Oceanica 2015a).

8.3.3 Possible behavioural responses

Presence of sea-cages

The physical presence of sea-cages invariably attracts large marine predators, which visit the cages in search of food. Food sources include either the accumulations of wild finfish beneath and around the sea-cages (which provide refuge for certain fish-species), or the aquaculture stock inside the sea-cages. Pinnipeds (fur seals and sea lions) in particular are capable of developing complex predation behaviour, ranging from damaging nets and cages to entering enclosed structures and feeding on the fish inside (Kemper et al. 2003). Once the behaviour is established in individuals, attempts to predate on fish within aquaculture sea-cages may occur all year round with seasonal or daily patterns, potentially resulting in serious injury or mortality to (Vilata et al. 2010). Seals and sea lions have been entangled in the cage nets, anchor lines and anti-predator nets that are designed as a protective barrier around the sea-cages. Entanglements generally result where sea-cages employ larger mesh sizes (>15 cm), have unrepaired holes, open bottom nets and/or loose or baggy nets (Kemper et al. 2003).

It has been determined that pinniped visitation is up to 10 times higher at fish farms that are located within 30 km of significant 'haul-out' sites (where sea lions congregate on land). At Port Lincoln, South Australia, for example, tuna sea-cages were located within 25 km to the second-largest, Australian sea lion breeding colony at Dangerous Reef, directly influencing the high level of pinniped predation observed (Kemper et al. 2003). Since the MWADZ is less than 10 km from the Australian sea lion haul-out site on the Easter Group of Islands, individuals from this population may be attracted to the proposed sea-cages. Recent population viability analyses revealed that all WA Australian sea lion populations are extremely vulnerable to additional mortality pressure, the impacts of which may lead to population declines, reduced survivorship and increased extinction risk for the species (Campbell 2005). Habitat degradation and interactions with aquaculture operations were identified as significant factors contributing to the lack of recovery for the species (DSEWPaC 2013a, b). Therefore, any threat of incidental mortality, including potentially negative impacts from aquaculture operations, may significantly affect Australian sea lions populations at the Abrolhos Islands.

Cetaceans also have a history of adverse interactions with sea-farms. In the Mediterranean Sea, coastal marine fish farms experienced a year-round presence of common bottlenose dolphins that were likely foraging opportunistically at or around the fish cages (Lopez & Shirai 2007). Entanglements have occurred, especially when the anti-predator nets are loose, and employ large mesh sizes (>15 cm). Furthermore, a recent Mediterranean study concluded that productive waters around aquaculture sea-cages attracted bottlenose dolphins and altered their foraging strategies, while they fed on discarded fish from the cages (Piroddi et al. 2011). In Australia, non-fatal and fatal entanglements in anti-predator nets with large mesh sizes (>15 cm) have been documented across several dolphin species, including common, bottlenose and dusky dolphins (Kemper et al. 2003). From these documented cases, the proposed MWADZ may have impacts on bottlenose dolphins, including indirect changes to their natural foraging behaviours and directly, via serious injury or mortality due to entanglement in anti-predator nets.

Adverse interactions between whales and aquaculture sea-cages have also been recorded. A humpback whale became entrapped within a sea-cage in Port Lincoln, and an unidentified whale is documented to have collided with a salmon cage in Tasmania (Pemberton et al. 1991, Kemper et al. 2003). Between 1982 and 2010, five humpback whales have become entangled in WA aquaculture gear for abalone, pearl and mussel (Groom & Coughran 2012). Humpback whales are common in the Abrolhos region (DSEWPaC 2013a), and there is therefore an elevated risk of adverse interactions with the MWADZ.

Additionally, the presence of sea-cages has the potential to adversely impact the marine environment through nutrient enrichment, which is a management concern for marine fauna, particularly marine turtles and dugongs (DSEWPaC 2012b). Inputs of inorganic nutrients, primarily dissolved inorganic nitrogen, are rapidly assimilated by phytoplankton. Under ideal conditions, inputs of nutrients may lead to excessive phytoplankton growth, resulting in extensive algal blooms (see Section 4.4.1); though, for this proposal, the risk of algal blooms is considered low (Section 7.4.6). Algal blooms are associated with reduced growth, development and reproduction in turtles (DSEWPaC 2012b).

Vessel movements

The proposed MWADZ will employ a range of vessels for operations, including maintenance, feeding and harvesting. Vessel presence and movements may directly (i.e. injuries and mortalities from collisions) and indirectly (i.e. behavioural disturbance from noise) impact marine mammals and turtles. The likelihood of a serious injury or mortality for a large whale from a vessel strike decreases when vessels travel at speeds less than 15 knots (Vanderlaan & Taggart 2007). Although dolphins are known to avoid moving vessels, large whales and turtles may not respond to approaching vessels depending on their activity at the time of collision. Behavioural disturbance may be indicated by various reactions, including (but not limited to) avoidance, swimming speed changes, quick dives, breathing changes and aggression (DEH 2006). Vessel collisions may incidentally injure or kill dugongs while feeding in shallow inshore waters, and dugongs are known to habituate to vessel traffic and disturbance, thereby increasing the likelihood for collisions and injuries (DSEWPaC 2012b). Management measures to reduce the likelihood of adverse impacts from vessel movements may include restrictions for approach distance and speed limits, as per the Australian National Guidelines for Whale and Dolphin Watching 2005 (DEH 2006).

Artificial lights

For safety, navigation and operational reasons, the proposed sea-cages may require lighting at night. Artificial lighting may cause adverse environmental impacts to marine fauna that are sensitive to light (such as marine turtles) by disrupting their natural behaviour through disorientation, attraction or avoidance (EPA 2010). Adult female turtles are known to avoid nesting at beaches illuminated with artificial light, and hatchlings depend on natural light to navigate to the open sea and maybe misguided by artificial light.

8.3.4 Major findings and recommendations

Sea-cage aquaculture has the potential to adversely impact marine mammal and turtle populations via a number of cause-effect-response pathways. Experiences elsewhere have shown that risks are exacerbated by farm practices and the choice of infrastructure. For example, incidents of visitation were heightened where excessive wastes (fish carcasses) were present in the water, and incidents of entanglement occurred in predator nets with mesh sizes greater than 15 cm. Other operational aspects that may increase the potential for adverse interactions included use of high intensity artificial light, excessive noise and vessel speeds greater than 15 knots.

Efforts to reduce interactions with Australian sea lions and bottlenose dolphins may include controlled feeding regimes, prompt removal of dead fish, tensioning nets and employing anti-predator nets with mesh sizes less than 15 cm in diameter (Schotte & Pemberton (2002) recommend mesh sizes of ~6 cm diameter). The most successful mitigation strategy requires physically excluding the fish stocks in the cages and during any movements or transfers (Robinson et al. 2008). Examples of the types of management measures to be implemented are provided in Table 8.2. All management options would most effectively be employed during routine operations, and/or incorporated to the aquaculture infrastructure. Compliance with the recommended approaches is likely to be assessed via an audit of operation records, including records of interactions with marine mammals and turtles.

Table 8.2 Summary of project aspects, potential environmental impacts and possible management measures for interactions with marine mammals and turtles

Project Aspect	Potential Environmental Impact	Possible Management Measures
Aquaculture cage	Feeding behaviour change Serious injury or mortality Habitat change	Anti-predator nets (mesh size <15 cm) Constant maintenance and monitoring Controlled feeding regimes to minimise waste and prompt removal of dead stock Use of semi-rigid or well tensioned net material Adequate distance from known fauna habitats High walled sea-cages to prevent pinniped access
Aquaculture activities	The availability of supplementary food (stock feed) may change feeding behaviour Noise associated with the installation of cages may cause behavioural disturbances	Controlled feeding regimes – to minimise feed waste Prompt removal of dead stock Noise levels at all times will be within Environment Protection (Noise) Regulations thresholds and it is preferential to install the cages outside of humpback whale southern migratory months (given humpback whales are the only “likely” migratory cetacean)
Vessel movements	Serious injury or mortality Behavioural disturbance	Do not approach within 100 m of a whale and 50 m of a dolphin Do not approach calves or pods with calves Move at slow speed (<15 knots) Avoid sudden/repeated changes in direction Avoid sudden/excessive noise Allow fauna to move in against the shore

Project Aspect	Potential Environmental Impact	Possible Management Measures
Lighting disturbance	Behavioural disturbance through: • disorientation • attraction • avoidance of important habitats	Reduce intensity of artificial light Use long-wavelength lights
Environmental quality	Toxicity Regional eutrophication	Water quality monitoring Sediment quality monitoring

8.4 Seabirds

8.4.1 Approach

The Abrolhos are one of the most significant seabird breeding locations in the eastern Indian Ocean (Section 3.7). Section 8.4 provides a summary of a desktop impact assessment applied to Abrolhos seabird populations. Text included in this section is excerpted from Halfmoon biosciences (2015), the full content of which is included in Appendix D.

The suggested approach to managing seabird interactions is outlined further in the EMMP for this proposal, which is published separately (BMT Oceanica, 2015).

8.4.2 Potential adverse interactions

Interactions which can have a detrimental impact upon seabirds can occur at the island breeding colony or whilst foraging at sea. Direct disturbance to colonies from human visitation can include trampling or exposure of nests, disorientation of nestlings, enhanced predation or kleptoparasitism and interruption to breeding or feeding behaviours. Adverse interactions while foraging may arise from attraction to, or avoidance of, vessels and marine infrastructure or disturbance to prey aggregations or associated predators and exposure to contaminants. Direct interactions with finfish farming operations could include:

- supplementary feeding from stock predation, fish food, waste material or food scraps
- collisions with sea cages, other structures or vessels moored at night
- attraction and disorientation due to lighting on service vessels, pens or navigation markers
- entanglement in cage mesh, predator nets or protective bird netting
- attraction of prey to vessel or sea cages due to “FAD” effects.
- attraction to the fish stock
- use of vessel or sea cages as roosting sites

The location of the Pelsaert Group aquaculture zone is 2 km from Stick Island. There is a mixed colony of little shearwaters and white-faced storm petrels on Stick Island (Surman and Nicholson 2009), and many wedge-tailed shearwaters use Middle Channel as a flight path back to their colonies on Pelsaert, Middle and Gun Islands from their foraging grounds. All these petrel species return to their colonies at night. The presence of a semi-permanently moored vessel could potentially impact upon individuals of these species through:

- collision
- light attraction
- disorientation

Collision rates will be greatly increased by unmasked, bright lights. These impacts may result in either injury or death. Also, birds found on the vessel decks invariably regurgitate meals meant to be delivered to young at the nest, thereby depriving those nestlings of a single feed.

At certain times of year, fledgling shearwaters and storm petrels depart nesting grounds and head to sea in the darkness of pre-dawn. These young inexperienced birds orientate to light on the horizon and are particularly vulnerable to being attracted to lighting, becoming disorientated. The food for the juvenile stock raised in the cages will be pelletised, which will have negligible attractiveness to pursuit-diving seabirds such as pied cormorants and wedge-tailed shearwaters. However, pied cormorants may be attracted to the cages to feed upon the juvenile stock and in doing so may attempt to reach fish through the mesh. This may present an entanglement issues for this species.

8.4.3 Possible behavioural responses

The Figures below outline cause-effect-response pathways for six key groups of seabirds that have been identified as being potentially impacted from fin fish aquaculture at the Abrolhos. These are:

- pied cormorants
- silver gulls
- pacific gulls
- wedge-tailed shearwaters
- neritic terns
- pelagic foraging terns and noddies

Of these, pied cormorants, silver gulls and Pacific gulls were considered particularly at risk due to their propensity to increase with proximity to new anthropogenic food sources (Halfmoon biosciences (2015)).

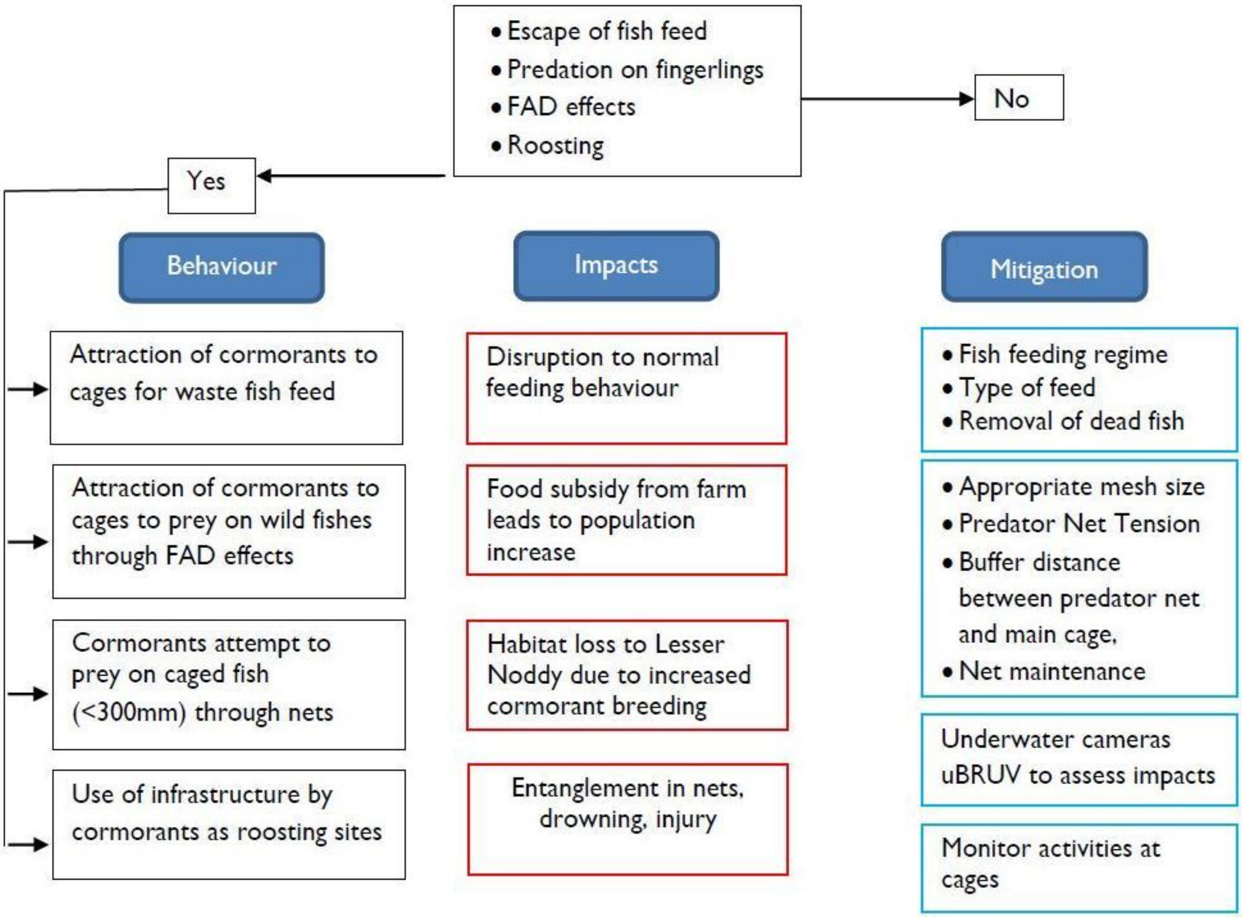


Figure 8-8 Potential impacts to cormorants and possible mitigation measures

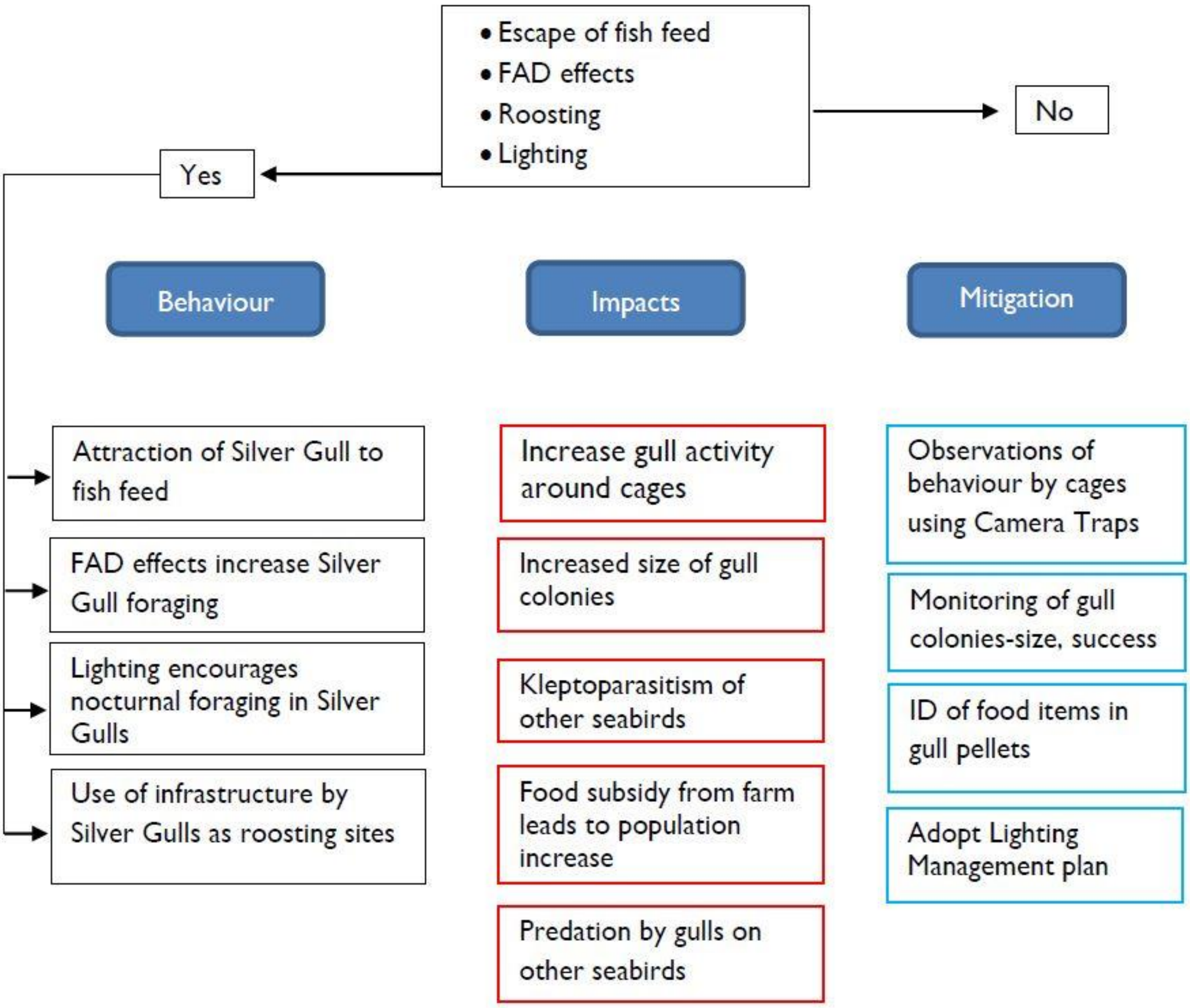


Figure 8-9 **Potential impacts to silver gulls and possible mitigation measures**

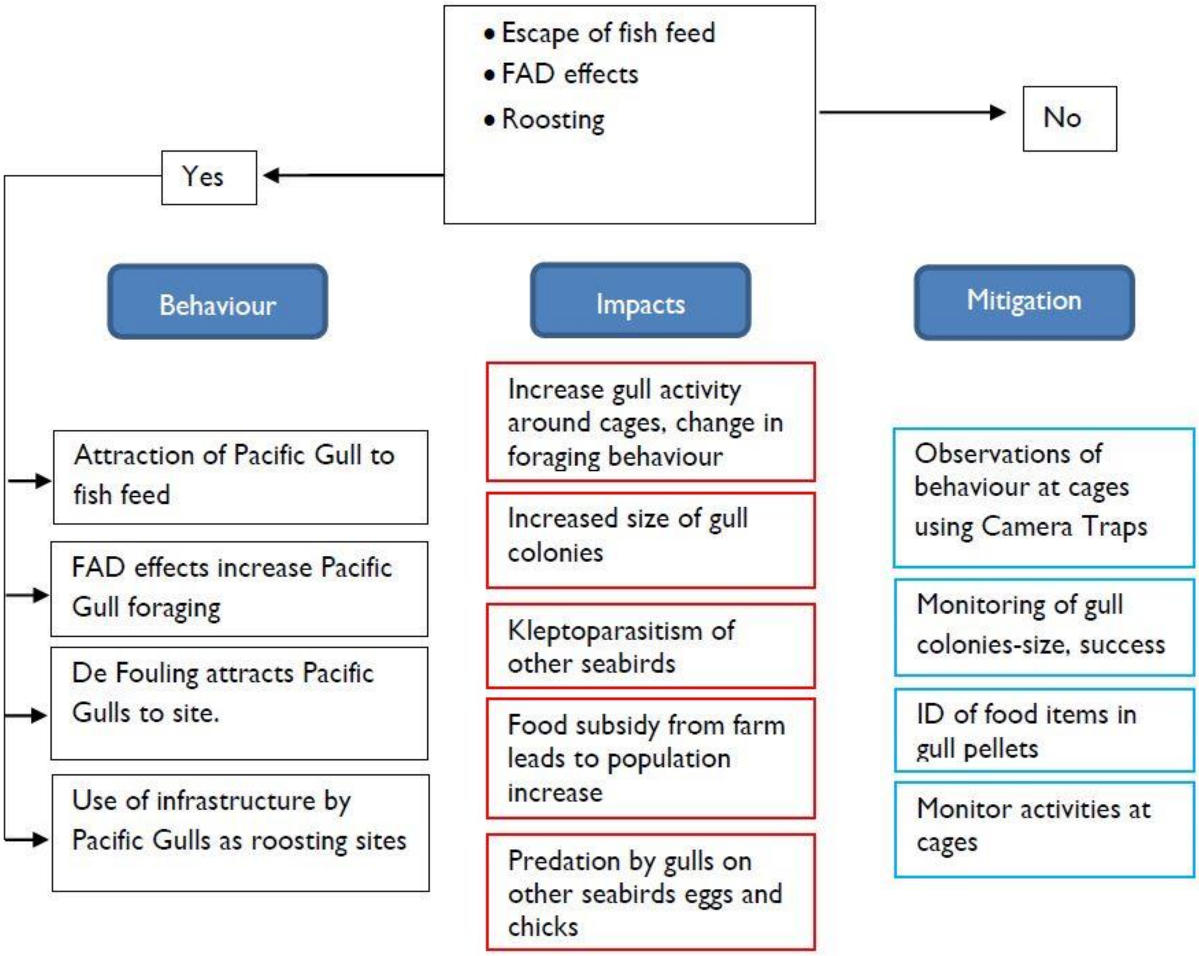


Figure 8-10 Potential impacts to Pacific gulls and possible mitigation measures

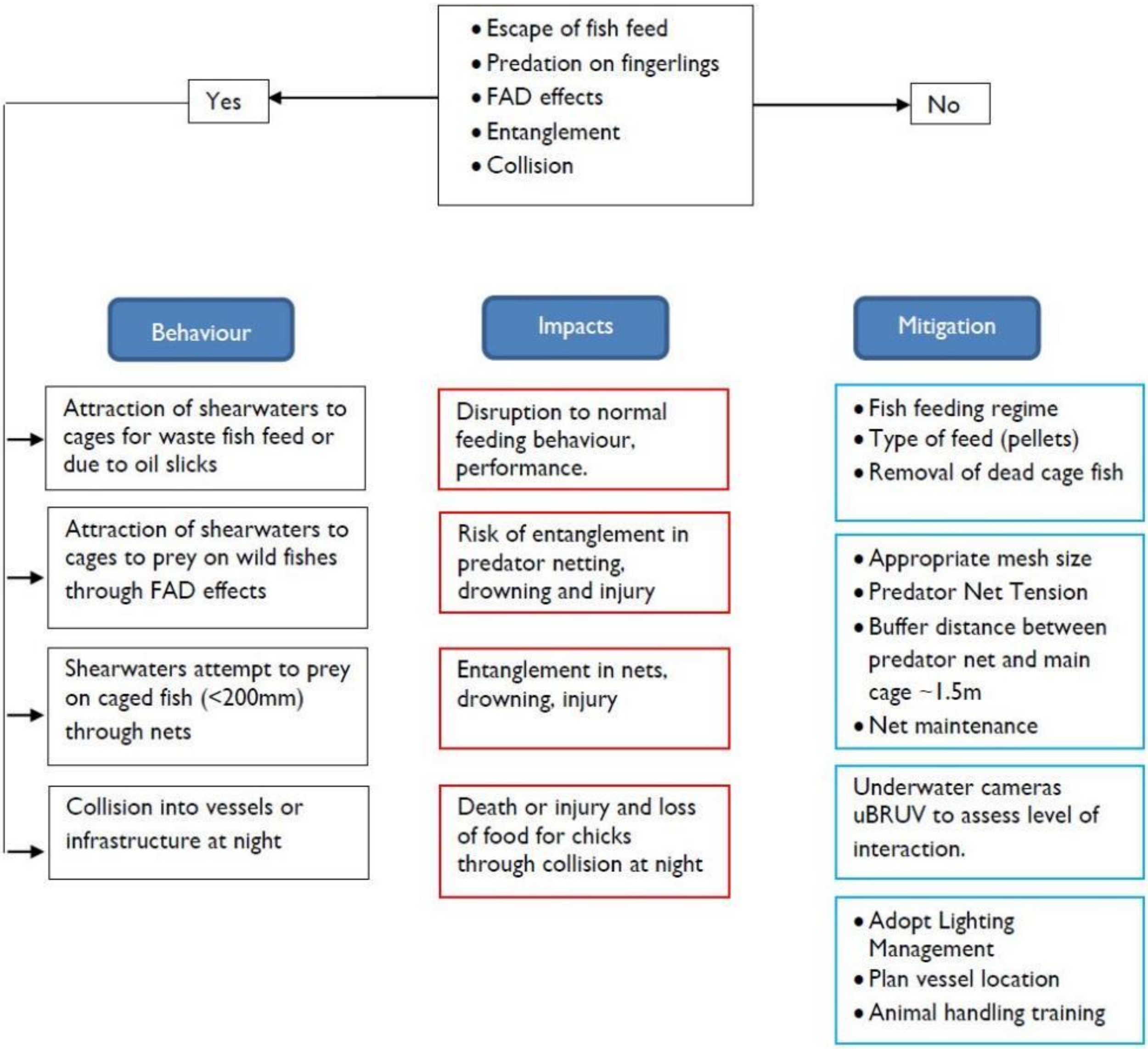


Figure 8-11 Potential impacts to wedge-tailed shearwaters and possible mitigation measures

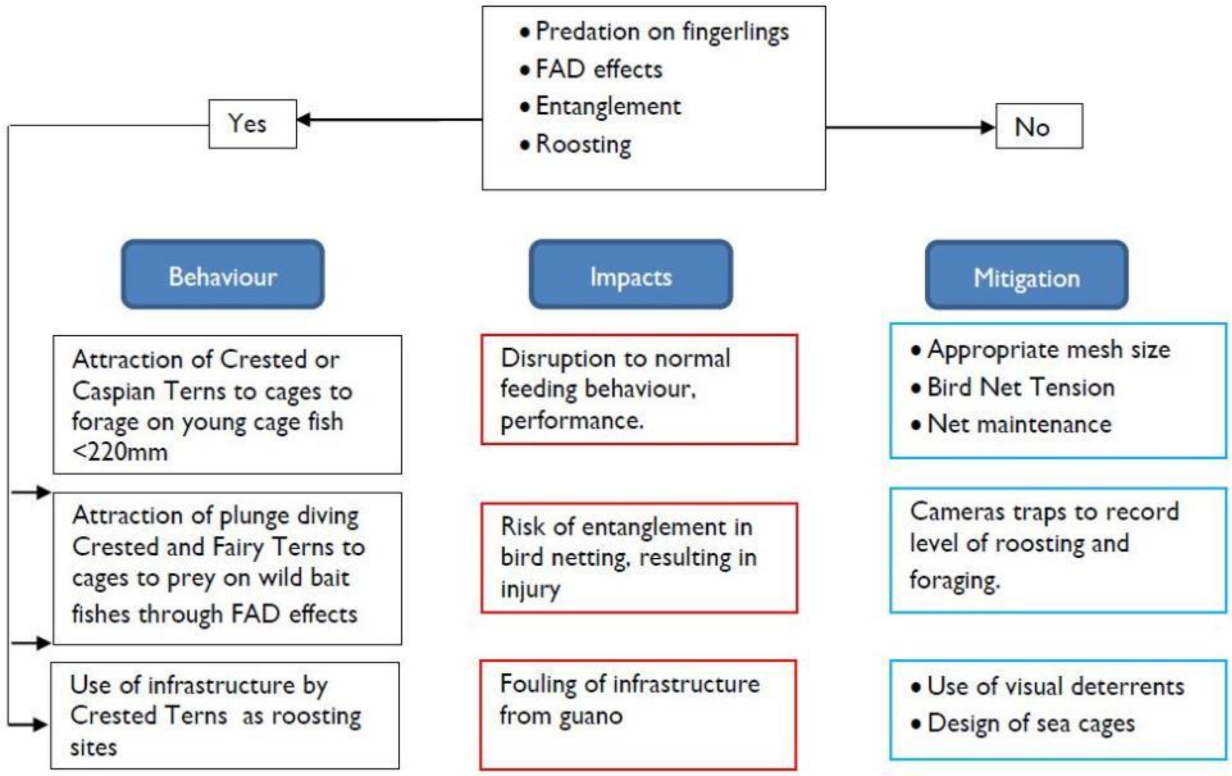


Figure 8-12 **Potential impacts to neritic terns and possible mitigation measures**

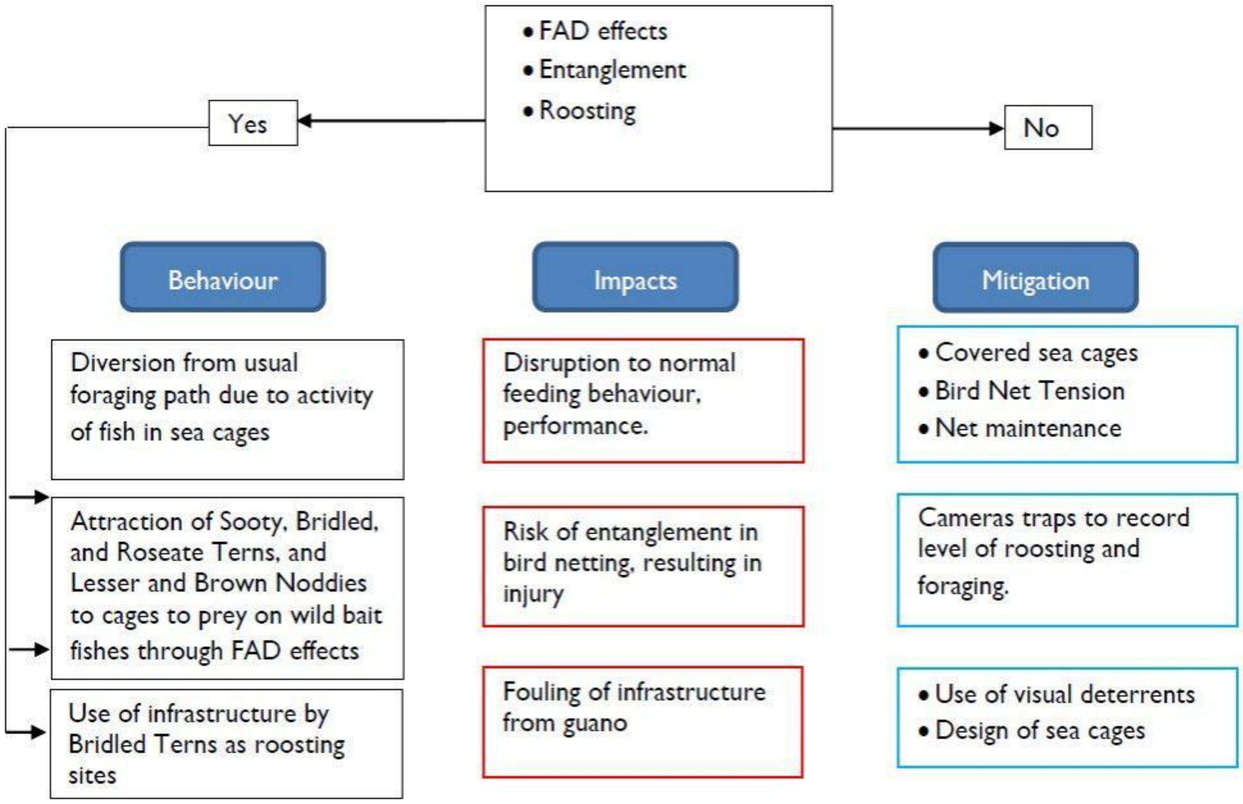


Figure 8-13 **Potential impacts to pelagic foraging terns and noddies and possible mitigation measures**

8.4.4 Risk and mitigation assessment

The potential adverse interactions (risks) between seabirds and sea-cage fish-farming at the Abrolhos are identified together with the available 'best practice' mitigation measures in Table 8.3.

Table 8.3 Seabird interaction risk mitigation

Factor	Interaction	Potential Consequence	Available Mitigation Methods
1. Pen Location	Attraction: • Seabirds attracted to pens from colonies on the Houtman Abrolhos Islands. • Seabirds distracted from normal flight path by fish activity adjacent sea cages or within sea cages.	• Changes in seabird behaviour or energetics, changing reproductive performance or increasing mortality • Changes in seabird population sizes leading to increased interspecific competition, kleptoparisitism, predation of eggs and young and habitat alteration on the Houtman Abrolhos Islands. • Shifts in terrestrial ecosystems driven by changes in breeding seabird numbers.	• All locations are within foraging range of all seabird breeding species. Choice between proposed fish-farming zones on this scale is unlikely to reduce potential for interactions.
2. Fish - feed	Fish feed is available to foraging seabirds providing an energy / nutrient subsidy, this is less likely if pelletised feed is used. Species likely to exploit fish food are gulls and cormorants.	• Increasing populations of potential increaser species (Silver Gulls, Pacific Gulls and Pied Cormorants) leading to ecological changes (see 1 above). • Increase Pied Cormorant populations will reduce nesting habitat for Lesser Noddies on Wooded Island. • Increased gull populations may impact other nesting seabirds through predation and competition.	• Pellets preferred over whole fish. • Sub-surface, slow release feeders. • Current speeds not sufficient to allow lateral export of feed through meshes. • Complete pen coverage with bird mesh. • Submersible sea-cages
3.Cultured fish size	• Seabirds attracted to forage on farmed stock within their preferred prey size ranges. • Seabirds distracted by large schooling species associated with mixed species foraging aggregations.	• Increasing populations of both gulls and cormorants leading to ecological changes (see 2 above). • Loss of cultured stock. • Reduced foraging efficiency reducing reproductive performance. • Risk of entanglement in anti-predator netting.	• Complete pen coverage with bird mesh. • Submersible sea-cages. • Anti-predator nets with appropriate mesh size for seabirds (6cm) • Space between anti predator net and sea cage ~1.5m.
4. Sea-pen diameter	Interactions with aerial-snatch predators (e.g. Sea-Eagles & Ospreys) will increase with pen diameter.	• Loss of farmed stock, and redistribution or increased abundance of marine raptors.	• Complete pen coverage with bird mesh. • Limit diameter of sea-cages. • Submersible sea-cages
5. Raft characteristics	Some seabirds (e.g. Bridled Terns, gulls) preferentially perch on flotsam or floating objects and may utilise sea-cages as roosts.	• Faeces from birds may reduce water quality, transfer pathogens / parasites to stock. • Collisions with structures or entanglement with nets. • Fouling of gear. • Negative interactions from staff towards native fauna	• Complete pen coverage with bird mesh. • Design of railings, floats, net-rings to reduce perching. • Alternative artificial rafts. • Submersible sea-cages • Bird Deterrents (Visual, audio, physical)
6. FAD effects	Attraction of larval fish and crustaceans, bait fishes and predatory fishes due to FAD effects of superstructures.	• Seabirds may concentrate around fish farms increasing potentially adverse interactions (see 1 above). • Increased foraging opportunities for some species (increaser species). • Increased risk of entanglement from foraging seabirds	• FAD effects are likely to increase with distance from reefs. • Alternative artificial rafts or reefs. • Mesh sizes.
7. Fish oil slicks	Oily residues from stock and feed will form slicks which draw-in forage fishes (enhancing FAD effect) and seabirds (particularly olfactory foragers such as shearwaters and storm-petrels).	• Seabirds may concentrate around fish farms increasing potentially adverse interactions (see 1 above). • Increased foraging opportunities for some species (increaser species). • Increased risk of entanglement from foraging seabirds, particularly diving species.	• Reduce oil content /production of feeds. • Remove dead fish from cage
8. Superstructure and predator nets	Structures including netting above and below the water surface may entrap or entangle foraging or roosting seabirds.	• Increased mortality particularly among pursuit diving species, e.g. cormorants and shearwaters. • Potential entanglement from Osprey and White-breasted Sea Eagles.	• Appropriate mesh sizes, visibility and net tension. • Regular net checks and maintenance • Camera trap monitoring • Remote Underwater Video (BRUV) monitoring
9. Lighting	• Many seabirds fly at night and are disorientated by bright navigation or vessel flood-lights. • Lights may also attract zooplankton further increasing the FAD effect of sea-cages allowing gulls to feed at night	• Increased seabird mortality from collisions with super structure of cages and moored vessels. • Enhanced prey aggregation around fish-farms may increase adverse interactions with seabirds. • Enhanced food supply for increaser species, Silver Gulls are known to forage under lights at night.	• Development of lighting management plan • Design of light horizon and wavelength. • Reduction in use of lighting. • Seasonal lighting reduction policies.
10.Moored Vessels	• Accommodation and farm vessels on site increase collision and disorientation risks to seabirds. • Moored vessels provide roosts for seabirds • Vessel wastes may attract increaser species. • Increased boating traffic may deter natural foraging behaviour.	• Increased seabird mortality from collisions (see 9 above). • Loss of food for seabird young from adults regurgitating after collision or disorientation on vessel. • Enhanced food supply for increaser species, Silver Gulls are known to forage under lights at night or on waste from vessels (food scraps, bait, and offal).	• Development of lighting management plan • Design of light horizon and wavelength. • Management plan for reducing impacts from collision • Training for bird handling and reporting • Reduction in use of lines or rigging across vessel • Mooring location outside of flight paths.
11.Marine Debris	Loss of lines, netting, plastics, floats or refuse from operations.	• Entanglement of marine fauna in portions of nets or lines lost from farm or over side of vessels (scuppers). • Ingestion of plastics from farm wastes, reduction in foraging efficiency and delivery of food to young.	• Waste management plan • Return of all waste to mainland • Maintenance of farm gear • Mesh over scuppers to prevent loss to sea.
12. Food Supplementation from de-fouling operations	Gulls that rely naturally on marine invertebrates may be attracted to operations removing encrustations	Food supplementation or entrapment	• Collection of biological material for disposal away from aquaculture operations or burial.

8.4.5 Major findings and recommendations

Studies of the potential adverse interactions between seabirds and aquaculture installations identified similar risk factors to those discussed in Halfmoon biosciences (2015). These include entanglement, habitat exclusion, disturbance from farm activities, increased prey availability, creation of roosting sites, implications to foraging success and spread of pathogens (Sagar 2008, Lloyd 2003, Comeau et al. 2009). However, additional findings are presented in Halfmoon biosciences (2015) including the potential for disruption to foraging patterns, decline in nesting habitat to vulnerable species and importantly changes in foraging behaviour and consequent predicted population changes in increaser gull species.

Key findings of the assessment outlined particularly the potential adverse effects of lighting and waste aquaculture feeds (Halfmoon biosciences 2015). Lights shining on the water-surface have the effect of attracting and concentrating plankton and other marine life suitable as feed for seabirds. This effect has resulted in increases in silver gull numbers in the offshore oil and gas industry, attracting the night-time visitation of seabirds to feed on the resulting prey aggregations. Bright lights directed towards the horizon may also attract and disorientate seabirds at night including shearwaters, storm-petrels and pelagic terns. Fledging shearwater chicks orientate to lights on the horizon and are common casualties at coastal towns, on ships and fishing boats. However, these effects were found to be easily mitigated through best-practice approaches to lighting management (Halfmoon biosciences 2015).

Under best-practice feed management, approximately 1% of uneaten feed is expected to enter the marine environment through the sides and bottom of the sea-cages. It is expected that waste feed will result in aggregations wild fish in the size ranges attractive to foraging pied cormorants (Halfmoon biosciences 2015). Investigations of the foraging ecology of 'high risk' increaser species, including pied cormorants, silver gulls and pacific gulls, indicate that all are reliant on naturally available prey types. Littoral zone invertebrates dominate the gull diets and benthic fishes dominate pied cormorant diets. While there is potential for pied cormorants, silver gulls and pacific gulls to increase through exploitation of food sources associated with the MWADZ, it is understood that best practices approaches to management (sea cage design, selection of netting and waste feed minimisation) are likely to reduce the potential for exploitation by these seabirds. For further context refer to Halfmoon biosciences (2015) in Appendix D.

9. Conclusions

Risks associated with the DoF proposal to establish a finfish aquaculture zone at the Abrolhos Islands were assessed based on a number of technical studies, including the development and execution of an integrated environmental model and multiple technical desktop assessments. The purpose of this document was to summarise the findings of the technical studies, and to provide advice on the likely cumulative impacts of sea-cage operations on the marine environment under a range of operational scenarios. Results have been evaluated in the context of the key environmental factors identified in the ESD (Table 1.1), and the findings of this document will feed into the broader PER for the MWADZ.

9.1 Baseline status of the proposed aquaculture zone

Results of the baseline studies indicate that the waters inside the project area are clean and well mixed. Maximum and minimum water temperatures were achieved in autumn (23.5°C) and winter (20.8°C), respectively. Salinity and dissolved oxygen levels were consistent through the water column with little evidence of stratification (Section 5.3.1). The water was highly oxygenated, achieving surface oxygen saturation levels between 98 and 99% and bottom oxygen saturation levels between 95 and 98% (Section 5.3.1). Light attenuation in the MWADZ was lower (0.04–0.19 per m) than that obtained in the KADZ (1.2–1.8 per m), results indicative of very clear water, with excellent light penetration.

Water currents are variable, ranging between 5.8 and 14.4 cm/s (Section 7.2). Concentrations of ammonium (2.7 µg/L) and chlorophyll-a (0.43 µg/L) were lower than those recorded in the Kimberley Aquaculture Development Zone (KADZ) (5.4 µg/L and 0.9 µg/L, respectively and compared well with those recorded in Perth's coastal waters, pointing to an overall oligotrophic (nutrient poor) environment. Nitrite + Nitrate levels (12.9 µg/L) were higher than those recorded in Perth's coastal waters (6.5 µg/L) and in the KADZ (8.7 µg/L). Concentrations of both inorganic nutrients and chlorophyll-a were seasonally variable, but higher in the cooler months.

The benthic environment consisted generally of a shallow (~15 cm thick) layer of sand overlying rocky substrate. Higher current speeds in the northern area (northern 13-14.5 cm/s compared to the south 8.7-11 cm/s) were reflected in the tendency toward larger sediment grain sizes in the northern reaches of the MWADZ (Section 5.4.1). Sediment conditions were variable, with seasonal fluctuations in ammonium, nitrogen and total organic carbon and generally higher values in warmer months. Infaunal assemblages were diverse (10 phyla; 129 families), with communities dominated by polychaetes (Section 5.4.4). Higher levels of infauna diversity and abundance were observed in the summer months.

Surveys indicated that the seafloor is a mosaic of habitats consisting of open sandy meadows and mixed biological assemblages. This mixture of substrates supports macroalgae, rhodoliths, sessile invertebrates and some corals; however, all of the available data suggest that their presence may be itinerant given the observed differences between surveys (Section 5.5). Northern MWADZ habitats were more diverse, with the northern area comprising 59% bare sand and 34% mixed assemblages. Small patches of reef were present near the north-east boundary of the MWADZ but only made up 8% of the total habitat. By contrast, the southern MWADZ comprised 96% bare sand and 5% mixed assemblage. Although ephemeral seagrass communities were have been observed historically in the MWADZ (Section 5.4.5), none were observed during the current assessment.

9.2 Suitability of the proposed aquaculture zone

Desktop assessments were undertaken to determine the likely impact of the proposal to marine mammals, seabirds and other significant fauna, including sharks and rays and other finfish. Several risks were identified, including the potential for the sea-cages to act as a physical impediment to animal movement and water flow, a source of entanglement/capture, an artificial source of food and as a significant artificial attractant and roosting area for seabirds.

These risks are not unique to the proposed MWADZ. Experience gained in Australia and in other parts of the world has resulted in significant advances in knowledge of aquaculture environmental management, including in the development of methods for both minimising risks and managing residual risks (Section 8). It was considered that where residual risks remained, these could be managed via the use of industry best-practice infrastructure and management strategies. Examples of these included use of high-walled sea-cages (to limit access of pinnipeds), use of nets to exclude seabirds, and implementation of modern fish-feeding methods to both limit wastage and impede opportunistic feeding by sea-birds. The suggested approach to management is outlined further in the EMMP for this proposal, which is published separately (BMT Oceanica 2015a).

Sea-cage aquaculture may under some circumstances lead to smothering or serious damage to benthic habitats including benthic primary producing habitats (BPPHs). The potential for impacts to BPPHs was assessed in the context of EAG 3 (see approach in Section 4.5.1). The assessment was undertaken against Category C in the Cumulative Loss Guidelines (EAG 3) which stipulates allowable losses of no more than 2% within an agreed local assessment unit (LAU). The assessment found that the proposal was unlikely to yield significant cumulative losses and the total cumulative loss would be restricted to less than 1%, which is below the 2% Category C benchmark. The findings of the assessment are in keeping with the overall results of the EIA, which predicted that the most severe impacts are restricted to small areas (Section 7.3).

The effect of sea-cages was also examined in the context of the local and regional hydrodynamics. Sea-cages invariably impart some resistance to flows, acting to slow or deflect waters in the vicinity of the cages. Sea-cages have the effect of increasing current speeds around and immediately beneath the cages. Where the cages are 'tall', and placed in shallow water, this can have the effect of scouring the underlying marine sediments. Hydrodynamic modelling undertaken in this study showed that the proposed cages were placed in sufficient water depths to avoid scouring of the benthos. Modelling indicated that water currents were slowed inside the cages, and slightly elevated (relative to background) beneath the cages. However, none of these effects were predicted to result in ecological consequences.

The results of the integrated modelling provided insights into the likely benthic footprints of the sea-cages under a range of scenarios (Table 4.16). Modelling was based on the assumption that wastes from sea-cages would exhibit different settling velocities. It was also assumed that the particles exhibited 'adhesive' properties (partly due to its mucus content), which reduced their resuspension potential relative to inorganic particles (Nowell et al. 1981; Masalo et al. 2008).

Risks associated with key water column contaminants, dissolved inorganic nitrogen (DIN) and suspended particles were examined after one year of production. Suspended particles were examined in the context of smothering and interruption to filter feeding processes, and DIN in the context of algal growth potential, nutrient enrichment and shading. Risks associated with organic waste inputs were examined in the context of sediment organic enrichment and changes to sediment chemistry. The time taken for sediments to achieve chemical remediation was determined following two, three and five years of finfish production.

Concentrations of DIN down-current of the sea-cages increased with increasing biomass and increasing stocking density. However, the plumes dissipated rapidly, with concentrations returning to levels consistent with a high level of ecological protection inside the southern MWADZ boundary, and within 2.3 km of the northern MWADZ boundary. Despite large inputs of DIN to the system, none of the scenarios resulted in significant changes to the chlorophyll-a concentrations in the broader project area. Similar results were obtained with respect to light and water column dissolved oxygen levels. The extent of light reduction (or shading) is largely associated with the extent of particles in the water, a proportion of which is phytoplankton. Although the proposal presents conditions under which phytoplankton may be stimulated, thus also increasing light attenuation, none of the modelled scenarios resulted in discernible chlorophyll-a concentrations and sub-surface light conditions were not affected (Section 7.4.4).

Deposition of organic material resulted in rapid changes to concentrations of oxygen and hydrogen sulphide in sediments beneath the sea-cages (Section 7.3.2). Results suggested that the ZoHI would occupy 82-177 ha (S2-S1) to 139-177 ha (S6-S5) after 5 years production (Section 7.3), but less after 3 (2-1 ha to 95-105 ha) and 2 years (0-0.2 ha to 88-91 ha) production. By reducing the length of the production period from 5 to 3 years, the area occupied by the ZoHI reduced by close to a 100% in S2, 45% in S4 and 31% in S6.

Reductions in both the standing biomass and the length of production also reduced the extent of the ZoHI, as measured along the maximum radius down-current from the cage clusters. After 5 years continuous production, the ZoHI, extended to a maximum of 110 m and 70 m under S6 and S5, but less than that under other scenarios, and shorter production periods: in S4 for example, distances reduced to 60 m and 15 m after 3 and 2 years production respectively, and for S3, the distance reduced to 10 m after 3 years production. After 2 years production, the ZoHI in S3 did not breach the cage cluster perimeter.

Increasing the stocking density, while maintaining the standing biomass (i.e. stocking density S4 > stocking density S3; standing biomass S4 = standing biomass S3), had the effect of reducing the total area occupied by the ZoHI across the zone. This effect was particularly strong after 5 years production, but less so after 3 and 2 years production. While the spatial extent of the ZoHI was reduced under these scenarios, the effect was to increase the intensity of impacts beneath the sea-cages, thus extending the time required for sediment (chemical) remediation during fallowing. Notwithstanding this prediction, the model indicated that large standing biomasses (up to 5000 t per sea-cage cluster) are achievable, while constraining the benthic impacts to relatively small areas. This is also reflected in the literature, with most detectable impacts to the sea-floor being restricted to within 10 and 100 m of the sea-cage perimeter (Carroll et al. 2003; Crawford 2003, Borja et al 2009).

The ZoHI is the area where impacts on benthic habitats are predicted to be irreversible, as per EAG 7. The term irreversible is defined as ‘lacking a capacity to return or recover to a state resembling that prior to being impacted within a timeframe of five years or less’. Despite the use of the term irreversible, it is noted that sea-cages are not permanent structures and can be moved to facilitate benthic rehabilitation. Recovery times in the ZoHI and ZoMI ranged between 1 and 7+ years, depending on the scenario, duration of production and the distance from the sea-cages. Immediately under the sea-cages, sediments required greater than 7 years to achieve full recovery, irrespective of the standing biomass modelled. However, this reduced to 6 and 5-6 after 3 and 2 years production respectively.

In addition to contributing organic wastes to the seafloor, aquaculture may contribute pharmaceuticals to the marine environment. Antibiotics are used as needed to treat bacterial disease occurring in farmed fish and are generally administered in feed. Calculations have shown that 70% to 80% of drugs used administered in fish farms end up in the environment, and drug concentrations with antibacterial properties have been detected in sediments beneath sea-cages (Samuelsen et al. 1992). Antibiotics may impart pressure on the environment by reducing or changing numbers of sediment bacteria, which in turn may affect biochemical and/or broader ecological processes. The persistent use of antibiotics has also been shown to lead to bacterial resistance (Anderson and Levin 1999). In the treatment of farmed salmon in Tasmania, oxytetracycline is the most common antibiotic used, accounting for more than 70% of total antibiotic use during 2006–2008 (Parsons 2012). A strong seasonal component to the use of antibiotics has been noted in Tasmania, with the greatest requirement in the summer months when water temperatures are elevated and pathogens most virulent. Oxytetracycline has been found to persist in marine sediments beneath sea cages for up to twelve weeks, with a half-life of ten weeks (Jacobsen and Berglind 1988). However, traces of the drug may be present for up to two years after treatment (Lalumera et al. 2004). It is also relatively persistent to anoxic conditions which are common under sea-cages (Jacobsen and Berglind 1988). Because antibiotics are administered in feeds, the spatial extent of potential impacts is likely reflected in the settlement patterns of organic wastes. Modelling predicted that the majority of wastes[4] in the MWADZ would be deposited to the seafloor within 60 m of the sea-cages. If antibiotics are required, it would be administered for short periods of time. The strongest effects of antibiotics could last for up to 10 weeks but are likely to be constrained to relatively small areas.

Suspended particles were examined in the context of smothering and interruption to filter feeding processes, and DIN in the context of algal growth potential, nutrient enrichment and shading. While none of the triggers for filter feeding processes were exceeded, some effects of smothering were detected (S4-S6), but where they occurred, were spatially constrained to areas immediately under the sea-cages. The very low density of (at least a significant portion) of fish faecal waste was reflected in the tendency for the smallest particles to disperse great distances beyond the sea-cages (several km over a 12 month period). These particles which contributed to the ZoI, were not predicted to reach the sediments in high enough volumes to exceed the environmental criteria. Areas classified as the ZoI could be expected to maintain normal chemical (oxygen and sulphide) signature, with no resulting changes in infaunal diversity.

In summary, results presented here indicate that the impacts of the proposal can be constrained within small areas of the MWADZ, with no adverse effects to regional environmental quality. Risks associated with significant marine fauna were considered manageable via the implementation of industry best-practice methods and use of appropriate infrastructure. Findings demonstrated the general suitability of the project area given its:

- water depth, which in turn contributes to a very large volume
- average current speeds, which are at the lower limit of ideal
- lack of extensive or permanent BPPHs
- location on historic trawling grounds and
- size, allowing ample scope for fallowing and associated recovery of benthic habitats

All of the modelling scenarios tested were based on full scale production, with between 15 000 and 30 000 t of standing biomass in the water at any one time (for up to five years). A conservative approach was adopted to ensure the outputs of modelling were equivalent to 'most likely worst case' outcomes, as required by the ESD (Table 2.1). As such, the impacts predicted in this document are more extensive than might be expected on average, but are within the upper range of impacts reported in the literature (i.e. Brooks et al. 2004).

---

[4] *Based on the Zone of High Impact after three years production*

9.3 Interim production limits

This assessment simulated the effects of standing biomasses in the range 15 000 t to 30 000 t of finfish, for periods of between one year (water quality) and five years (sediments). Despite using a conservative approach, none of the simulations were predicted to result in detrimental changes in water quality, and only scenarios S4–S6 were predicted to impart severe impacts (ZoHI) to sediments greater than 70 m beyond the immediate vicinity of the sea-cages.

The constraining factor, therefore, is whether the scale of impacts to sediment is environmentally acceptable, and whether they can be controlled via targeted management strategies (such as fallowing) and through the use of appropriately classified areas of ecological protection (EPA 2015). It is also considered that even when calibrated appropriately, environmental models are subject to many sources of compounding error. Although no adverse effects to the regional environment were predicted at the upper range of the scenarios tested (i.e. 30 000 t), it is recommended that 24 000 t standing biomass is set as an interim limit, pending further validation of the particle dispersion and sediment diagenesis models.

Baseline field data on sediment characteristics and water quality collected during operations will provide suitable information with which to validate the models, and thus fine-tune their precision. This in turn may be used to adjust the allowable future production limits, according to the results of the modelling outputs.

9.4 Recommendations

Results presented within this report are equivalent to the 'most-likely worst-case' outcomes as per the requirements of the ESD. The tested scenarios were designed to be (a) sufficient to support a viable finfish aquaculture industry and (b) within the critical assimilative capacity of the marine environment, based on an understanding of systems with similar flushing regimes and similar nutrient inputs (see Section 7.4.6). As such, it is recommended that the mid-range limit 24 000 t standing biomass is set as an interim limit, pending further validation post-commencement of operational monitoring. It is further recommended that this limit is validated in the future in the context of additional metocean assessments, including the effect of severe storms, and the frequency of benthic ‘resetting’ events.

10. References

Abdo DA, Bellchambers LM, Evans SN (2012) Turning up the heat: Increasing temperature and coral bleaching at the high latitude coral reefs of the Houtman Abrolhos Islands. PLoS One 7:e43878

Alongi DM, McKinnon AD, Brinkman R, Trott LA, Undu MC, Muawanah, Rachmansyah (2009) The fate of organic matter derived from small-scale fish cage aquaculture in coastal water of Sulawesi and Sumatra, Indonesia. Aquaculture 295:60–75

Anderson MJ, Gorley RN, Clarke KR (2008) PERMANOVA+ for PRIMER: Guide to software and statistical methods. PRIMER-E Pty Ltd, Plymouth, UK

ANZECC & ARMCANZ (2000) National Water Quality Management Strategy: Paper No 4 - Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Volume 1 - The Guidelines (Chapters 3 – Aquatic Ecosystems). Australian and New Zealand Environment and Conservation Council / Agriculture and Resource Management Council of Australia and New Zealand, October 2000.

Austen MC, Warwick RM, Rosando MC (1989) Meiobenthiic and macrobenthic community structure along a putative pollution gradient in southern Portugal. Marine Pollution Bulletin 20: 398–405

Babcock RC, Willis BL, Simpson CJ (1994) Mass spawning of corals on a high latitude coral reef. Coral Reefs 13:161–169

Baden SP, Loo L-O, Pihl L, Rosenberg R (1990) Effect of eutrophication on benthic communities including fish: Swedish West Coast. Ambio 19: 113–122

Bannister A, Kemper CM, Warnecke RM (1996) The action plan for Australian cetaceans. Australian Nature Conservancy Agency, Canberra, ACT, September 1996

Bellwood DR, Hughes TP, Folke C, Nystrom M (2004) Confronting the coral reef crisis. Nature 429: 827–833Benedetti et al. 2010

Benetti DD, Benetti GI, Rivera JA, Sardenberg B, O' Hanlon B (2010) Site selection criteria for open ocean aquaculture. Marine Technology Society Journal 44: 22-35

Berner R (1980) Early diagenesis: A theoretical approach. New Jersey, Princeton University Press

Beveridge M (2004) Cage aquaculture. Third Edition. U.K.: Fishing News Books, Blackwell Sci. Ltd. 376 pp

BMT Oceanica (2015a) Mid-West Aquaculture Zone Environmental Monitoring and Management Plan. Prepared for the Department of Fisheries by BMT Oceanica Pty Ltd, Report No. 1164_001/1_RevE, September 2015

BMT Oceanica (2015b) Potential impact of the Mid-west Aquaculture Development Zone on local marine mammals and turtles. Technical memorandum prepared for the Department of Fisheries by BMT Oceanica Pty Ltd, September 2015

BMT Oceanica (2015c) Perth Long-term ocean outlet monitoring program - 2015 annual summer water quality survey. Report prepared for Water Corporation by BMT Oceanica Pty Ltd, June 2015, Report No. 1120_006/2_Rev0

Brearley A (1997) Seagrasses and isopod borers from the Wallabi Islands, Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 1. Western Australian Museum, Perth, Western Australia, p 63–74

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

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

Borja A, Rodriguez GJ, Black K, Bodoy A, Emblow C, Fernades TF, Forte J, Karakassis I, Muxika I, Nickell TD, Papageorgiou N, Pranovi F, Sevastou K, Tomassetti P, Angel D (2009) Assessing the suitability of a range of benthic indices in the evaluation of environmental impact of fin and shellfish aquaculture located in sites across Europe. Aquaculture 293: 231-240

Boudreau BP (1996) A method-of-lines code for carbon and nutrient diagenesis in aquatic sediments. Computers & Geosciences, 22(5): 479-496

Boudreau BP (1997) Diagenetic models and their implementation. Heidelberg, Springer

Brigolin DR, Pastres TD, Nickell CJ, Cromey DR, Aguilera and P Regnier (2009) Modelling the impact of aquaculture on early diagenetic processes in sea loch sediments. Marine Ecology-Progress Series 388: 63-80.

Campbell R (2005) Historical distribution and abundance of the Australian sea lion (*Neophoca cinerea*) on the west coast of Western Australia. Fisheries Research Report no. 148. Department of Fisheries, Perth, Western Australia

Carroll ML, Cochrane S, Fieler R, Velvin R, White P (2003) Organic enrichment of sediments from salmon farming in Norway: Environmental factors, management practices, and monitoring techniques. Aquaculture 226: 165–180

Chabanne D, Finn H, Salgado-Kent C, Bejder L (2012) Identification of a resident community of bottlenose dolphins (*Tursiops truncatus*) in the Swan Canning Riverpark, Western Australia, using behavioural information. Pacific Conservation Biology 18:247–262

Chen YS, Beveridge MCM, Telfer TC (1999) Settling rate characteristics and nutrient content of the faeces of Atlantic salmon, *Salmo salar* L., and the implications for modelling of solid waste dispersion, Aquaculture Research, 30:395-398

Chevron (2012) Gorgon Gas Development and Jansz Feed Gas Pipeline: Coastal and Marine Baseline State and Environmental Impact Report. Chevron Australia, Document No G1-NT-REPX0001838, Perth, Western Australia, November 2012

Clarke KR (1993) Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18:117–143

Clarke KR, Green RH (1988) Statistical design and analysis for a "biological effects" study. Marine Ecology Progress Series 46: 213–226

Clarke KR, Somerfield PJ, Chapman MG (2006) On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray-Curtis coefficient for denuded assemblages. Journal of Experimental Marine Biology and Ecology 330:55–80

Comeau, LA, St-Onge P, Pernet F and Lanteigne L (2009) Deterring coastal birds from roosting on oyster culture gear in eastern New Brunswick, Canada. Aquaculture Engineering 40: 87-94

Cornejo P, Hito HH, Gutierrez MH, Olivares G (2014). Numerical studies on the hydrodynamic effects of a salmon farm in an idealized environment. Aquaculture 430: 195-206

Couture R-M, Shafei B, Van Cappellen P, Tessier A, Gobeil C and Tessier A (2010) "Non-steady state modelling of arsenic diagenesis in lake sediments. Environmental Science & Technology 44(1): 197-203

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

Cresswell GR (1991) The Leeuwin Current - Observations and Recent Models, J Royal Soc Western Australia, 74, 1-14

Cromey CJ, Black KD, Edwards A, Jack IA (1998) Modelling the deposition and biological effects of organic carbon from marine sewage discharges. Estuarine, Coastal and Shelf Science 47: 295–308

Crossland CJ, Hatcher BG, Atkinson MJ, Smith SV (1984) Dissolved nutrients of a high-latitude coral reef, Houtman Abrolhos Islands, Western Australia. Marine Ecology Progress Series 14:159–163

Dauer DM, Rodi AJJ, Ranasinghe JA (1992) Effects of low dissolved oxygen events on the macrobenthos of the lower Chesapeake Bay. Estuaries 15: 384–391

deBruyn AMH and Gobas APC (2004) Modelling the diagenetic fate of persistent organic pollutants in organically enriched sediments. Ecological Modelling 179: 405-416

DEH (2006) Australian National Guidelines for Whale and Dolphin Watching 2005. Department of the Environment and Heritage, Canberra, Australian Capital Territory

DEWHA (2008) The North-west Marine Bioregional Plan – Bioregional profile. A description of the ecosystems, conservation values and uses of the North-west Marine Region. Department of the Environment Water Heritage and the Arts, Canberra, Australian Capital Territory

DHI (2013) Kimberley Aquaculture Zone - Environmental Field Studies and Numerical Modelling in Support of an EIS. Final report prepared for the Western Australian Govt. Dept. of Fisheries, May 2013

Dimitriadis C and Koutsoubas D (2011) Functional diversity and species turnover of benthic invertebrates along a local environmental gradient induced by an aquaculture unit: the contribution of species dispersal ability and rarity. Hydrobiologia 670: 307-315

DoF (2015a) Threat identification, hazard pathway analysis and assessment of a key risk presented by the establishment of the Mid West Aquaculture Development Zone, to the sustainability of threatened, endangered and protected fish populations at the Abrolhos Islands, Western Australia, Version 4, August 2015

DoF (2015b) Threat identification, hazard pathway analysis and assessment of the key risks presented by the establishment of the Mid West Aquaculture Development Zone to invertebrate and finfish species and fisheries at the Abrolhos Islands, Western Australia, Version 2, July 2015

DoE (2014a) Protected Matters Search Tool. Australian Government Department of the Environment, Canberra, Australian Capital Territory. Available at <http://www.environment.gov.au/topics/about-us/legislation/environment-protection-and-biodiversity-conservation-act-1999/protected> [Accessed 6 November 2014]

DoE (2014b) Species profile and threats database: Department of the Environment, Canberra, ACT. Available at <http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=68447> [Accessed 6 November 2014]

DoE (2015) Species profile and threats database: Department of the Environment, Canberra, ACT. Available at <http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=68447> [Accessed 8 June 2015]

Double MC, Gales N, Jenner KCS, Jenner M-N (2010) Satellite tracking of south-bound female humpback whales in the Kimberley region of Western Australia. Final report to the Australian Marine Mammal Centre, Tasmania, September 2010

Double MC, Jenner KCS, Jenner M-N, Ball I, Laverick S, Gales N (2012) Satellite tracking of pygmy blue whales (*Balaenoptera musculus brevicauda*) off Western Australia. Final report to the Australian Marine Mammal Centre, Tasmania, May 2012

DPIPWE (2011) Section 40 report in relation to the draft amendment no.3 to the D'Entrecasteaux Channel Marine Farming Development Plan February 2002. Department of Primary Industries, Parks, Water and Environment, Tasmania

DSEWPaC (2012a) Species Group Report Card – Cetaceans. Australian Government Department of Sustainability, Environment, Water, Population and Communities, Canberra, Australian Capital Territory.

DSEWPaC (2012b) Species Group Report Card - Marine Reptiles. Department of Sustainability, Environment, Water, Population and Communities, Canberra, Australian Capital Territory

DSEWPAC (2013) Issues paper for the Australian sea lion (*Neophoca cinerea*). Department of Sustainability, Environment, Water, Population and Communities, Australian Commonwealth Government, Canberra, Australian Capital Territory

DSEWPaC (2013a) Recovery plan for the Australian sea lion (*Neophoca cinerea*). Department of Sustainability, Environment, Water, Population and Communities, Australian Commonwealth Government, Canberra, Australian Capital Territory

Dzwilewski PT and Fenton G (2003) Shock Wave/Sound Propagation Modelling Results for Calculating Marine Protected Species Impact Zones During Explosive Removal of Offshore Structures. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2003-059. 39 pp

Eleftheriou A, Moore DC, Basford DJ, Robertson MR (1982) Underwater experiments on the effect of sewage sludge on a marine ecosystem. Netherlands Journal of Sea Research 16: 465–473EPA 2011 – EAG 7

EPA (2010) Environmental Assessment Guidelines for Protecting Marine Turtles from Light Impacts. No 5. Environmental Protection Authority, Perth, Western Australia, November 2010

EPA (2011) Environmental Assessment Guideline for Marine Dredging Proposals. Environmental Protection Authority, Report No EAG 7, Perth, Western Australia, December 2009

EPA (2013) Environmental Scoping Document for the Mid-west Aquaculture Development Zone. Prepared by the Environmental Protection Authority, July 2013

EPA (2015) Environmental quality criteria reference document for Cockburn Sound (2015), Perth, Western Australia

Erseus C (1997) Marine Tubificidae (Oligochaeta) from the Montebello and Houtman Abrolhos Islands, Western Australia, with descriptions of twenty-three new species. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 2. Western Australian Museum, Perth, Western Australia, p 389–458

Felsing M, Telfer T, Glencross B (2005) ^{15}N-enrichment of an aquaculture diet and tracing of cage culture waste in an estuarine environment. J. App. Icthyology 22: 419-426

Feng M, Majewski LJ, Fandry CB & Waite AM (2007) Characteristics of two counter-rotating eddies in the Leeuwin Current system off the Western Australian coast. Deep Sea Research Part II: Topical Studies in Oceanography, 54(8): 961-980

Fernandes M and Tanner J (2008) Modelling of nitrogen loads from the farming of yellowtail kingfish *Seriola lalandi* (Valenciennes, 1833). Aquaculture research 39: 1328-1338

Finn H (2005) Conservation biology of bottlenose dolphins (*Tursiops* sp.) in Perth metropolitan waters. PhD Thesis, Murdoch University, Perth, Western Australia

Findlay RH, Watling L, Mayer LM (1995) Environmental impact of salmon net-pen culture on marine benthic communities in Maine: a case study. Estuaries 18: 145– 179

Forbes TL, Lopez G R (1990) The effect of food concentration, body size and environmental oxygen tension on the growth of the deposit-feeding polychaete, *Capitella* species 1. Limnology and Oceanography 35: 1535–1544

Forbes TL, Forbes VE, Depledge MH (1994) Individual physiological responses to environmental hypoxia and organic enrichment: Implications for early soft-bottom community succession. Journal of Marine Research 52: 1081–1100

Fossing H, P Berg B, Thamdrup S, Rysgaard H, Sorensen and K Nielsen (2004) A model set-up for an oxygen and nutrient flux model for Aarhus Bay (Denmark)

Froelich P, Klinkhammer G, Bender M, Luedtke N, Heath G, Cullen D (1979) Early oxidation of organic matter in pelagic sediments of the eastern equatorial Atlantic suboxic diagenesis. Geochimica et Cosmochimica Acta 43(7): 1075-1090

Gaston GR, Edds KA (1994) Long-term study of benthic communities on the continental shelf off Cameron, Lousiana: a review of brine effects and hypoxia. Gulf Res Reports. 9: 57–64

Glover EA, Taylor JD (1997) Diversity and distribution of subtidal molluscs from the outer continental shelf, Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 2. Western Australian Museum, Perth, Western Australia, p 281–305

Goldsworthy SD, Page B, Kennedy C, Welz K, Shaughnessy PD (2011) Australians sea lion population monitoring at Seal Bay and the Seal Slide, Kangaroo Island: 2010 breeding season. Report to the Department of the Environment and Natural Resources. SARDI Aquatic Sciences Publication Number F2011/000216-1. SARDI Research Report Series No: 556, pp 36

Gray JS (1992) Eutrophication in the sea. In: Marine Eutrophication and Population Dynamics. Colombo G, Ferrari I, Ceccherelli VU, Rossi R. Olsen and Olsen, Fredensbor (eds), Denmark. pp. 3–16

Groom CJ, Coughran DK (2012) Entanglements of baleen whales off the coast of Western Australia between 1982 and 2010: patterns of occurrence, outcomes and management responses. Pacific Conservation Biology 18:203–214

Gross JE (2003) Developing conceptual models for monitoring programs. National Parks Service Inventory and Monitoring Program, Ft Collins

Halfmoon biosciences (2015) Impact Assessment of aquaculture on seabird communities of the Abrolhos Islands, to support the Mid-West Aquaculture Development Zone proposal. Report prepared by Halfmoon Biosciences for the Department of Fisheries, August 2015, 93pp

Halide H, Stigebrandt A, Rehbein M, McKinnon AD (2009) Developing a decision support system for sustainable cage aquaculture. Environ Model Softw 24:694-702

Hamer DJ, Ward TM, Shaughnessy PD, Clark SR (2011) Assessing the effectiveness of the Great Australian Bight Marine Park in protecting the endangered Australian sea lion Neophoca cinerea from bycatch mortality in shark gillnets. Endangered Species Research 14:203–216

Hargrave BT, Holmer M, Newcombe CP (2008) Towards a classification of organic enrichment in marine sediments based on biogeochemical indicators. Marine Pollution Bulletin 56:810–824

Hargrave BT (2010) Empirical relationships describing benthic impacts of salmon aquaculture. Aquaculture Environment Interactions 1:33–46

Harriott VJ (1998) Growth of the staghorn coral *Acropora formosa* at Houtman Abrolhos, Western Australia. Marine Biology 132:319–325

Hasegawa D, Sheng J, Greenberg DA, Thompson KR (2011) Far-field effects of tidal energy extraction in the Minas Passage on tidal circulation in the Bay of Fundy and Gulf of Maine using a nested grid coastal circulation model. Ocean Dynamics 61:1845–1868

Hatcher BG, Rimmer DW (1985) The role of grazing in controlling community structure on a high latitude coral reef: Measurements of grazing intensity. Proceedings of the Fifth International Coral Reef Symposium 6:229–236

Hatcher BG, Kirkman H, Wood WF (1987) Growth of the kelp *Ecklonia radiata* near the northern limit of its range in Western Australia. Marine Biology 95:63–73

Herzfeld M, Parslow J, Sakov P, Andrewartha J (2006) Biochemical modelling of Australia's northwest shelf. Technical Report No. 8

Heyward A, Fromont J, Schonberg CHL, Colquhoun J, Radford B, Gomez O (2010) The sponge gardens of Ningaloo Reef, Western Australia. The Open Marine Biology Journal 4:3–11

Hughes TP, Graham NAJ, Jackson JBC, Mumby PJ, Steneck RS (2010) Rising to the challenge of sustaining coral reef resilience. Trends Ecol Evol 25: 633–642

Hutchins JB (1997) Checklist of fishes of the Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 1. Western Australian Museum, Perth, Western Australia, p 239–253

Hutchings PA (1997) The Terebellidae (F. Polychaeta) from the Wallabi Group, Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 2. Western Australian Museum, Perth, Western Australia, p 459–501

Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Botsford LW, Bourque BJ, Bradbury RH, Cooke R, Erlandson J, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ, Warner RR (2001) Historical overfishing and the recent collapse of coastal ecosystems. Science 293: 629–637

Jenner KCS, Jenner M-N, McCabe KA (2001) Geographical and temporal movements of humpback whales in Western Australian waters. APPEA Journal 2001:749–765

Jenner C, Jenner MN (2004) Photo-ID resights of blue whales in the Perth Canyon. Unpublished data in possession of authors

Josefson AB, Jensen JN (1992) Effects of hypoxia on soft-sediment macrobenthos in southern Kattegat, Denmark. In: Marine Eutrophication and Population Dynamics. Colombo G, Ferrari I, Ceccherelli VU, Rossi R (eds). Olsen and Olsen, Fredensborg, Denmark. pp. 21–28

Karakassis I, Hatziyanni E, Tsapakis M, Plaiti W (1999) Benthic recovery following cessation of fish farming: A series of successes and catastrophes. Marine Ecology Progress Series 184:205–218

Kemper CM, Pemberton D, Cawthorn M, Heinrich S, Mann J, Wursig B, Shaughnessy P, Gales R (2003) Aquaculture and marine mammals: co-existence or conflict? In Gales N, Hindell M, Kirkwood R (eds) Marine Mammals: Fisheries, Tourism and Management Issues. CSIRO Publishing, Victoria, Australia, p. 208–225

Keogh MJ and Mapstone BD (1997) Designing environmental monitoring for pulp mills in Australia. Wat. Sci. Tech. 35: 397-404

Limpus CJ (2007) A Biological Review of Australian Marine Turtles: 5. Flatback Turtle *Natator depressus* (Garman). Environmental Protection Agency, Brisbane, Queensland, November 2007

Lloyd BD (2003) Potential effects of mussel farming on New Zealand’s marine mammals and seabirds: a discussion paper. Department of Conservation, Wellington

Lopez BD, Shirai JAB (2007) Bottlenose dolphin (*Tursiops truncatus*) presence and incidental capture in a marine fish farm on the north-eastern coast of Sardinia (Italy). Journal of the Marine Biological Association of the UK 87:113–117

Littler MM, Littler DS (1984) Models of tropical reef biogenesis: the contribution of algae. Round FE, Chapman DJ (Eds) Progress in phycological research. Biopress, Bristol, pp 323–364

Macleod CKA, Mitchell IM, Crawford CM, Connell RD (2002) Evaluation of sediment recovery after removal of finfish cages from marine farm lease no. 76 (Gunpowder Jetty), North West Bay. TAFI Technical Report No. 13. January 2002. Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Hobart, Australia

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

Madsen PT, Wahlberg M, Tougaard J, Lucke K and P.L. Tyack (2006) Wind turbine underwater noise and marine mammals: Implications of current knowledge and data needs. Mar. Ecol. Prog. Ser. 309: 279–295

Mahnken CVW (1993) Benthic faunal recovery and succession after removal of a marine fish farm. PhD thesis, University of Washington, Seattle 290 pp

Marsh H, Penrose H, Eros C, Hughes J (2002) Dugong Status Report and Action Plans for Countries and Territories, United Nations Environment Programme, Nairobi

Marsh LM (1994) Echinoderms of the Houtman Abrolhos Islands, Western Australia and their relationship to the Leeuwin Current. In: David B, Guille A, Feral J-P, Roux M (eds) Echinoderms through time. Balkema, Rotterdam, p 55–61

Masalo I, Guadayol O, Peters F, Oca J (2008) Analysis of sedimentation and resuspension processes of aquaculture biosolids using an oscillating grid. Aquacultural Engineering 38: 135-144

Masini (2012). Strategic Approach to Offsets - Dredging Science Initiative.

Massoudieh AF, Bombardelli T, Ginn and F Bombardelli (2010) A biogeochemical model of contaminant fate and transport in river waters and sediments. Journal of Contaminant Hydrology 112(1-4): 103-117

Mazzola A, Mirto S, La Rosa T, Fabiano M, Danovaro R (2000) Fish-farming effects on benthic community structure in coastal sediments: Analysis of meiofaunal recovery. ICES Journal of Marine Science 57: 1454–1461

McCauley RD, Jenner C (2010) Migratory patterns and estimated population size of pygmy blue whales (*Balaenoptera musculus brevicauda*) traversing the Western Australian coast based on passive acoustics. Unpublished Paper (SC/62/SH26) presented to the International Whaling Committee Scientific Committee, Morocco, June 2010

McCauley RD, Jenner C, Bannister JL, Burton CLK, Cato DH, Duncan A (2001) Blue whale calling in the Rottnest Trench – 2000, Western Australia Report R2001-6. Centre for Marine Science & Technology, Perth, Western Australia

McLeod and Forbes (2004) Guide to the assessment of sediment condition at marine finfish farms in Tasmania

Moccia R, Bevan D, Reid G (2007) Composition of Fecal Waste from Commercial Trout Farms in Ontario: Macro and Micro Nutrient Analyses and Recommendations for Recycling. Final Report Submitted to the Ontario Sustainable Aquaculture Working Group Environment Canada, 22pp

Moore JAY, Bellchambers LM, Depczynski MR, Evans RD, Evans SN, Field SN, Friedman KJ, Gilmour JP, Holmes TH, Middlebrook R, Radford BT, Ridgway T, Shedrawi G, Taylor H, Thomson DP, Wilson SK (2012) Unprecedented mass bleaching and loss of coral across 12˚ of latitude in Western Australia in 2010-11. PLoS One 7:e51807

Moran D, Pether S, Lee PS (2009) Growth, feed conversion and faecal discharge of yellowtail kingfish (*Seriola lalandi*) fed three commercial diets. New Zealand Journal of Marine and Freshwater Research, 2009, Vol. 43: 917-927

Morrisey DJ, Gibbs MM, Pickmere SE, Cole RG (2000) Predicting impacts and recovery of marine-farm sites in Stewart Island, New Zealand, from the Findlay–Watling model. Aquaculture 185 (3– 4): 257–271

Nardi K, Jones GP, Moran MJ, Cheng YW (2004) Contrasting effects of marine protected areas on the abundance of two exploited reef fishes at the sub-tropical Houtman Abrolhos Islands, Western Australia. Environmental Conservation 31:160–168

National Research Council (2005) Marine Mammal Populations and Ocean Noise: Determining when noise causes biologically significant effects. Washington, DC: The National Academies Press. 126 pp

NOAA (2015) Coral Reef Watch. National Oceanic and Atmospheric Administration, Washington DC, USA. Available from <http://coralreefwatch.noaa.gov/satellite/index.html> [Accessed 14 April 2015]

Nowel ARM, Jumars PA, Eckman JE (1981) Effects of biological activity on the entrainment of marine sediments. Mar. Geol. 42: 133–153

Oceanica (2010). Oakajee Port – Marine Mammal Baseline Investigation 2008–2009. Prepared for Oakajee Port and Rail by Oceanica Consulting Pty Ltd, Report No 503_010/2, Perth, Western Australia, September 2010

Oceanica (2013) Kimberley Aquaculture Zone Strategic Assessment - Cause-effect pathways and impact thresholds. Report prepared for DHI by Oceanica Consulting Pty Ltd, Report No. 961_008/1

Olesiuk PF, JW Lawson and EA Trippel (2012). Pathways of effects of noise associated with aquaculture on natural marine ecosystems in Canada. DFO Can. Sci. Advis. Sec. Res. Doc. 2010/025. vi + 64 p

Orsini J-P, Shaughnessy PD, Newsome D (2006) Impacts of human visitors on Australian sea lions (*Neophoca cinera*) at Carnac Island, Western Australia: Implications for tourism management. Tourism in Marine Environments 3:101–115

Paraska DW, MR Hipsey and SU Salmon (2014) Sediment diagenesis models: Review of approaches, challenges and opportunities. Environmental Modelling & Software 61(0): 297-325

Parsons K (2012) State of the D’Entrecasteaux Channel and the lower Huon Estuary 2012. Report prepared by Ecomarine Consulting for the D’Entrecasteaux Channel Project, 222pp

Parsons, K. E. (2012). State of the D’Entrecasteaux Channel and the lower Huon Estuary 2012. Report for the D’Entrecasteaux Channel Project, prepared by Ecomarine Consulting, pp 195

Pattiaratchi C, Woo M (2009) The mean state of the Leeuwin Current system between North West Cape and Cape Leeuwin. Journal of the Royal Society of Western Australia 92:221-241

Pearce AF (1997) The Leeuwin Current and the Houtman Abrolhos Islands. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 1. Western Australian Museum, Perth, Western Australia, p 11–46

Pearce AF, Feng M (2013) The rise and fall of the “marine heat wave” off Western Australia during the summer of 2010/2011. Journal of Marine Systems 111–112:139–156

Pearce A & Pattiaratchi C 1999 The Capes Current: a summer countercurrent flowing past Cape Leeuwin and Cape Naturaliste, Western Australia. Continental Shelf Research 19: 401–420

Pearce A F, Rossbach M, Tait M & Brown R (1999) Sea temperature variability off Western Australia 1990 to 1994. Fisheries WA Research Report 111, 45 pp

Pearson TH, Rosenberg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology An Annual Review 16: 229–311

Pemberton D, Brothers N, Copson G (1991) Predators on marine fish farms in Tasmania. Papers and Proceedings of the Royal Society of Tasmania 125:33–35

Peña S, Katsev, Oguz and Gilbert (2010) Modeling dissolved oxygen dynamics and hypoxia. Biogeosciences 7

Pendoley KL (1997) Sea Turtles and Management of Marine Seismic Programs Western Australia. Petroleum Exploration Society of Australia Journal 25:8–15

Pendoley KL (2005) Sea turtles and the environmental management of industrial activities in north-west Western Australia. PhD Thesis, Murdoch University: Perth.

PIANC (2010) Dredging and Port Construction around Coral Reefs. PIANC and United Nations Environment Program, Report No 108

Piroddi C, Bedarzi G, Christensen V (2011) Marine open cage aquaculture in the eastern Mediterranean Sea: a new trophic resource for bottlenose dolphins. Marine Ecology Progress Series 440:255–266

Phillips JC, Huisman JM (2009) Influence of the Leeuwin Current on the marine flora of the Houtman Abrolhos. Journal of the Royal Society of Western Australia 92:139-146

Rasher DB, Engel S, Bonito V, Fraser GJ, Montoya JP, Hay ME (2012) Effects of herbivory, nutrients, and reef protection on algal proliferation and coral growth on a tropical reef. Oecologia 169: 187–198

Richardson WJ, CR Greene Jr, CI Malme and DH Thomson (1995) Marine Mammals and Noise. New York: Academic Press. 576 pp

Robinson S, Terauds A, Gales R, Greenwood M (2008) Mitigating fur seal interactions: relocation from Tasmanian aquaculture farms. Aquatic Conservation: Marine Freshwater Ecosystems 18:1180–1188

Ross GJB, Burbidge AA, Brothers N, Canty P, Dann P, Fuller PJ, Kerry KR, Norman FI, Menkhorst PW, Pemberton D, Shaughnessy G, Shaughnessy PD, Smith GC, Stokes T, Tranter J (1995) The status of Australia's seabirds. In ‘State of the marine environment report of Australia Technical Annex: 1’. (eds L. Zann and P. Kailola) pp 167–182

Rossi V, Feng M, Pattiaratchi C, Roughan M & Waite AM (2013) On the factors influencing the development of sporadic upwelling in the Leeuwin Current system. Journal of Geophysical Research: Oceans, 118(7): 3608-3621

Sagar P (2008) Assessment of the potential impacts on waders and seabirds of finfish marinefarming in the Firth of Thames. Environment Waikato. 30pp.

Schaffner LC, Jonsson P, Diaz RJ, Rosenberg R, Gapcynski P (1992) Benthic communities and bioturbation history of estuarine and coastal systems: effects of hypoxia and anoxia. In: Marine Coastal Eutrophication (eds) Vollenweider RA, Marchetti R. Viviani R, Elsevier, Amsterdam, London, New York, Tokyo pp. 1001–1016

Shaughnessy PD, Goldsworthy SD, Hamer DJ, Page B, McIntosh RR (2011) Australian sea lions *Neophoca cinerea* at colonies in South Australia: distribution and abundance, 2004 to 2008. Endangered Species Research 13:87–98

Schotte R, Pemberton D (2002) Development of a stock protection system for flexible oceanic pens containing finfish. Fisheries Research and Development Project 99/361. Fisheries Research and Development Project 99/361. Fisheries Research and Development Corporation, Hobart pp 85

Smith SV (1981) The Houtman Abrolhos Islands: Carbon metabolism of coral reefs at high latitude. Limnology and Oceanography 26:612–621

Southall BL, AE Bowles WT, Ellison JJ, Finneran RL, Gentry CR, Greene Jr, D Kastak DR, Ketten JH, Miller PE, Nachtigall WJ, Richardson JA, Thomas and PL Tyack (2007) Marine mammal noise exposure criteria: Initial scientific recommendations. Aquat. Mamm. 33: 411-522

Storr GM, Johnstone RE and Griffin P (1986) Birds of the Houtman Abrolhos, Western Australia. Rec. W.A. Mus. 24: 1–42

Sukumaran A (1997) Circulation and flushing characteristics of the Easter Group Lagoon, Houtman Abrolhos Islands. BSc(Hons) Thesis, Department of Environmental Engineering, University of Western Australia, Perth.

Surman CA (2004) The Potential Impacts of Tuna Ranching on Seabirds of the Recherche Archipelago and surrounds. Unpublished report prepared for Indian Ocean Fresh by Halfmoon Biosciences. 7pp

Surman CA and Nicholson LW (2009) A survey of the breeding seabirds and migratory shorebirds of the Houtman Abrolhos, Western Australia. Corella, 33(4):89-98

Tanner JE, Clark TD, Fernandez M and Fitzgibbon Q (2007) Innovative solutions for aquaculture: spatial impacts and carrying capacity - further developing, refining and validating existing models of environmental effects of finfish farming. South Australian Research and Development Institute – Aquatic Sciences, Adelaide, Australia

Tanner JE and Fernandes M (2010) Environmental effects of yellowtail kingfish aquaculture in South Australia. Aquaculture Environment Interactions. 1:155-165

Van Cappellen P and Wang FY (1995) Metal cycling in surface sediments. Metal contaminated aquatic sediments. H. Allen. Chelsea, Ann Arbor Press: 21 - 61

Van Cappellen P and YF Wang (1996) Cycling of iron and manganese in surface sediments: A general theory for the coupled transport and reaction of carbon, oxygen, nitrogen, sulfur, iron, and manganese. American Journal of Science 296(3): 197-243

Vanerlaan ASM, Taggart CT (2007) Vessel collisions with whales: the probability of lethal injury based on vessel speed. Marine Mammal Science 23(1): 144–156

Veron JEN, March LM (1988) Hermatypic corals of Western Australia: Records and annotated species list. Records of the Western Australian Museum, Supplement 29:1–136

Vilata J, Oliva D, Sepulveda M (2010) The predation of farmed salmon by South American sea lions (*Otaria flavescens*) in southern Chile. ICES Journal of Marine Science 67: 475–482

Underwood AJ (1997) Experiments in Ecology: Their Logical Design and Interpretation Using Analysis of Variance. Cambridge Univ. Press, Cambridge

Underwood AJ, Chapman MG (2003) Power, precaution, Type II error and sampling design in assessment of environmental impacts. Journal of Experimental Marine Biology and Ecology 296: 49-70

Waite AM, Thompson PA, Pesant S, Feng M, Beckley LE, Domingues CM & Twomey L (2007) The Leeuwin Current and its eddies: An introductory overview. Deep Sea Research Part II: Topical Studies in Oceanography, 54(8), 789-796.

Watson DL, Harvey ES, Kendrick GA, Nardi K, Anderson MJ (2007) Protection from fishing alters the species composition of fish assemblages in a temperate-tropical transition zone. Marine Biology 152:1197–1206

Watson DL, Anderson MJ, Kendrick GA, Nardi K, Harvey ES (2009) Effects of protection from fishing on the lengths of targeted and non-targeted fish species at the Houtman Abrolhos Islands, Western Australia. Marine Ecology Progress Series 384:241–249

Watson JE (1997) The hydroid fauna of the Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 2. Western Australian Museum, Perth, Western Australia, p 503–546

Webster FJ, Dibden CJ, Weir KE, Chubb CF (2002) Towards an assessment of the natural and human use impacts on the marine environment of the Abrolhos Islands. Vol 1: Summary of existing information and current levels of human use. Fisheries Research Report 134, Department of Fisheries, Western Australia, 124p.

Wells FE (1997) Introduction to the marine environment of the Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the Houtman Abrolhos Islands, Western Australia. Volume 1. Western Australian Museum, Perth, Western Australia, p 1–10

Wells FE, Bryce CW (1997) A preliminary checklist of the marine macromolluscs of the Houtman Abrolhos Islands, Western Australia. In: Wells FE (ed) The marine flora and fauna of the

Houtman Abrolhos Islands, Western Australia. Volume 2. Western Australian Museum, Perth, Western Australia, p 362–383

Wilson BR, Marsh LM (1979) Coral reef communities at the Houtman Abrolhos, Western Australia, in a zone of biogeographic overlap. Proceedings of the International Symposium on Marine Biogeography in the Southern Hemisphere. New Zealand Department of Scientific and Industrial Research Information Series 137:259–278

Womersley HBS (1981) Biogeography of Australasian marine macroalgae. In: Clayton MN, King RJ (eds) Marine botany: An Australasian perspective. Longman Cheshire, Melbourne, p 293–307

Woo M and Pattiaratchi C (2008) Hydrography and water masses off the western Australian coast. Deep Sea Research Part I: Oceanographic Research Papers, 55(9), 1090-1104.

Wu Y, Chaffey J, Law B, Greenberg DA, Drozdowski A, Page F, Haigh S (2014) A three-dimensional hydrodynamic model for aquaculture: a case study in the Bay of Fundy. Aquaculture Environmental Interactions 5: 235-248

Yelverton JT (1981) Underwater explosion damage risk criteria for fish, birds, and mammals. Manuscript presented at 102nd Meet. Acoust. Soc. Am., Miami Beach, FL. Dec. 1981

Yelverton JT, DR Richmond ER, Fletcher and RK Jones (1973) Safe distances from underwater explosions for mammals and birds. DNA 3114T. Rep. from Lovelace Foundation for Medical Educ. and Res., Albuquerque, NM for Defense Nuclear Agency, Washington, DC. 67 pp. NTIS AD-766952

Appendix A

Potential for Impact to Marine Mammals and Turtles

Appendix B

Potential for Impact to Endangered and Protected Finfish

Appendix C

Potential for Impact to Invertebrates and Finfish Species & Fisheries

Appendix D

Potential for Impact to Seabirds

Appendix E

Peer Review of Modelling Processes and Interpretation

Appendix F

Development and Calibration of the Hydrodynamic and Wave Models

Appendix G

Development of the Sediment Diagenesis Model

BMT Oceanica Pty Ltd
PO Box 462 Wembley
WA 6913 Australia
Tel: +61 8 6272 0000
Fax: +61 8 6272 0099
bmtoceanica@bmtoceanica.com.au
www.bmtoceanica.com.au