Managing ecosystem interactions across differing environments: building flexibility and risk assurance into environmental management strategies

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Abbreviations

Technical terminology

ADCP	Acoustic doppler current profiler
AUV	Autonomous underwater vehicles
BACI	Before, after, control, impact
BGC	Biogeochemical
BRUV	Baited remote underwater video camera
CI	Inner control
CO	Outer control
CONNIE	CONNectivity InterfacE
CPCe	Coral point count with Excel extension
DIC	Dissolved inorganic carbon
DO	Dissolved oxygen
FG	Functional group
FIA	Flow injection analysis
GLM	Generalised linear model
H’	Shannon-Weaver diversity index
LOI	Loss on ignition
LVM	Latent variable model
MARE	Mean absolute relative error
MFDP	Marine farming development plan
OLS	Ordinary least squares
ORP	Oxidation reduction potential
PCO	Principal coordinate analysis
RMSE	Root mean square error
ROV	Remotely operated vehicle
RVA	Rapid visual assessment
TSS	Total suspended solids
UVC	Underwater visual census
$\delta^{13}C$	Carbon stable isotope
$\delta^{15}N$	Nitrogen stable isotope

Location Identification

Leases

SB1	Storm Bay 1
TR	Trumpeter
EL	East of Lippies

THC	Table Head Central
FR	Franklin
GR	Gordon River
SH	Strahan

Reef Monitoring Sites

ACTA	Actaeons
BBN	Bull Bay North
BBS	Bull Bay South
BIGH	Southerly Bight
CQE	Cape Queen Elizabeth
CQN	Cape Queen North
DPT	Dennes Point
ESPE	Esperence Point
FR	Franklin site
GR	Gordon River site
LADY	Lady Bay
LIPP	Lippies Point
LOMA	Lomas Point
OT1	One Tree South 1
OT2	One Tree South 2
OTN	One Tree North
PART	Partridge Island
PENG	Penguin Point
REDC	Redcliffs
SCOT	Scott Point
SIST	Sisters Bay
STIS	Southport Island
STPT	Southport
TBM	Trumpeter Bay Mid
TBN	Trumpeter Bay North
TBS	Trumpeter Bay South
TOWE	Tower Bay
VBS	Variety Bay South
VBY	Variety Bay
ZUID	Zuidpool Rock

1 Executive Summary

What the report is about

The overarching aim of this research was to provide an improved understanding of the environmental interactions of salmon farming and to provide recommendations to both government and industry on monitoring and management strategies that are appropriate to the level of risk associated with these interactions.

Background

The criteria for monitoring and assessment of sediment impacts and recovery associated with intensive salmon farming were established more than 15 years ago. However, changes in farming practices, innovations in technology, and expansion of the industry into new areas highlighted the need to review the underlying principles to ensure management and monitoring strategies remain best practice, and that farming operations continue to be sustainable in all regions. Consequently, a program of research was established to assess current monitoring and management strategies, provide an understanding of regional and operational variability in local scale (sediment) response, and define both common and regionally specific local scale response principles.

In addition, it was noted there was increasing concern in the community about the potential for broader scale interactions with reef systems and it was recommended this should be a feature of any resultant research plan seeking to inform monitoring and management. It was also recognised that modelling capabilities have increased markedly since the original research was conducted, and so an important component of this study was to evaluate how currently available modelling tools could support monitoring.

Approach

The resultant program of research undertook targeted surveys assessing the magnitude and extent of the response to organic enrichment in sediments and in adjacent reef systems across a range of new farming regions (i.e. Southern Channel, Storm Bay, Macquarie Harbour) and compared those results with previous data and assessment criteria to identify regional or operational differences. This information was then used to evaluate management and monitoring criteria under contemporary farming contexts. Farm-based modelling approaches (DEPOMOD/ NEW DEPOMOD) and more recent model emulator tools (i.e. EMS and CONNIE) were tested against empirical data to evaluate their performance in predicting the deposition of particulate wastes and the dispersion of dissolved nutrients. The value of these models as management tools for predicting and informing both the likelihood and level of risk associated with distribution of farm wastes to different locations and habitats was assessed.

Aims/Objectives

Objective 1: Establish the key sediment responses and associated assessment criteria for all areas in which farming currently occurs, building on our existing understanding to identify both generic and regionally specific performance criteria.

Objective 2: Improve our understanding of how sediments process farm waste across different levels of production and in different growing areas. This will help to ensure that sediment assessment criteria, used for monitoring and management, are appropriate and optimised for each growing region.

Objective 3: Evaluate the potential for interactions between local reef systems and salmon farming, determine the risk of direct impacts, recommend monitoring and assessment approaches appropriate to the level of risk, and identify mitigation strategies where relevant.

Objective 4: Assess the performance of modelling tools that have been developed specifically to support management decision making in predicting the dispersion of farm waste (dissolved and particulate) and consider how these might best be applied in the Tasmanian context (generally and regionally).

In line with these objectives the report is divided into 3 sections:

- Local Scale Monitoring (Objective 1 & 2)
- Reef Interactions (Objective 3)
- Modelling (Objective 4)

The key findings of each section are summarised below.

Local Scale Monitoring

Maintaining sustainable performance of Tasmanian marine farms requires an understanding of how farming in new areas and changes in farming practices might change environmental interactions. This component of the study addressed the need to validate local scale monitoring approaches in Macquarie Harbour and in two new growing areas in southern Tasmania (Storm Bay and the Southern Channel) to ensure monitoring and the indicators of environmental impact remain fit for purpose.

Methods

For all study leases, replicated samples were collected along transects which started at a stocked cage and extended outwards to 1 km. At each of the Storm Bay and Southern Channel leases, there were three transects with sites at 0 m, 35 m, 100 m, 200 m, 500 m and 1000 m from the cage. In Macquarie Harbour, there were two transects at each lease, with sites at 0 m, 50 m, 100 m, 250 m and 500 m from the cage. In addition, outer control sites were sampled (at least 1000 m from the cage) at all leases. Closer (inner) control sites (100-300 m from cages) that were 35m from the lease boundary were also included at the Storm Bay and Southern Channel leases to assess performance at the regulatory compliance distance. At each site, a full suite of physico- chemical and biological parameters were evaluated together with ROV visual assessments and sediment process measurements.

Key findings

Comparison of the biotic and abiotic factors in Macquarie Harbour and the two new growing areas in southern Tasmania showed the sediments in Macquarie Harbour were inherently depauperate (low faunal abundance, species richness and diversity) whilst Storm Bay and Southern Channel sites supported diverse and species rich communities. Differences in sediment grain size, a factor that influences the macrofauna community composition, were also apparent with Macquarie Harbour sediments being much finer than those of the other two study regions. Measurements of sediment redox levels indicated Macquarie Harbour sediments were highly reducing (i.e. inherently low in oxygen/ anaerobic) for much of the system whereas the two southern study regions were generally oxidising (i.e. aerobic). This is not surprising given Macquarie Harbour's highly stratified water column, deep central basin and shallow entrance to the ocean that has been shown to result in reduced mixing of bottom waters and naturally low dissolved oxygen conditions (Ross et al. 2021). In contrast, the sediment conditions at the other two study regions reflect the open nature of those systems and the increased levels of water exchange, which result in higher oxygen concentrations in the bottom waters and sediments. The carbon and nitrogen signatures of the sediments also highlighted the different background sources of organic material in each of the regions. In Macquarie Harbour the isotopic signature of the sediments indicated a far greater contribution of terrestrial and freshwater inputs to the organic matter pool. In addition, there was a clear change in the signal along the harbour, with the terrestrial signal increasing with distance from the harbour entrance. Isotopic signals from the two southern study regions were more consistently marine.

The differing oxygen concentrations also affect the pathways that break down organic matter and process nitrogen. The results indicate the microbial transformation of ammonia to nitrate (nitrification) occurs predominately in the water column and the transformation of nitrate to nitrogen gas (denitrification) in the sediments in Macquarie Harbour, whereas in the south, both processes appear to occur primarily in the sediments.

Given the clear biotic (faunal) and abiotic (biogeochemical) differences in conditions between the sites it is not surprising the overall response to marine farming in Macquarie Harbour was quite different from that in the two southern study regions. The biological (faunal) response to enrichment at the southern sites was more similar, but there were local differences that reflected the prevailing communities and more subtle changes in environmental conditions. The responses to farming inputs at the southern sites was also broadly consistent with those previously outlined for organic enrichment in Tasmania (Macleod and Forbes 2004) and elsewhere (Pearson and Rosenberg 1976) with the composition of taxa indicating the key stages of degradation and recovery. Recovery of the sediments at cage sites, following a period of fallowing, was associated with an increase in certain taxa; these same species were typically most abundant at the sites 35 m from the cage, indicating spatial recovery as well as temporal recovery. Further from the cage, the community composition was consistent with that generally associated with lower levels of enrichment and improved environmental conditions. However, there were regional differences in the community response, and all three southern leases had a larger footprint than might have been suggested based on the earlier studies by Macleod and Forbes (2004) and others. The community composition and process rates suggested the influence of farming, albeit at much lower levels, was evident and measurable out to distances of 200 m at the leases in the south and 500 m in Macquarie Harbour. The more expansive footprint in the south is likely a result of changes in the prevailing environmental conditions (deeper sites and stronger, more dispersive currents) at the newer leases as well as changes in farming practices (bigger cages).

In Macquarie Harbour, the change in the benthic community with distance from the cages was quite different to that observed in other farming locations around Tasmania. This is largely due to its unique environmental conditions and benthic ecology; however, the significance of the observed changes and the overall pattern of response was consistent with elsewhere in so much as the community and changes defined different levels of organic enrichment and as such could be used to monitor and predict change. The highly enriched cage sites were dominated by opportunistic species (i.e. species which can rapidly respond to changes in the organic loading of the sediment). Interestingly, the community response pattern in Macquarie Harbour varied much more between leases and surveys than in the southern sites. This appears to be not only driven by variation in differences in farm management such as feed inputs and farm history but also changes in local environmental conditions such as bottom water oxygen concentration. Given the greater complexity of the prevailing environmental conditions in Macquarie Harbour, the predictability of management response is more complex. However, understanding the biological response to sediment conditions, and knowing this will follow a predictable pattern, does provide a basis upon which to generate reliable indicators of sediment health for management.

Like macrofaunal community structure, the direct measurement of sediment mineralisation rates also proved to be a highly sensitive measure of organic matter enrichment in all study regions, and therefore could be a powerful tool should a more detailed assessment of the spatial footprint of a farm be required. Redox and sulphide also remained useful indicators of sediment health, but redox potential appeared to be less variable across the range of conditions assessed and as such would be a more reliable measure of change than sulphide concentrations. Sulphide remained an informative measure of relative change across sites and therefore is likely to be more useful to describe spatial patterns of enrichment than temporal change. In Macquarie Harbour, the results showed both redox and sulphide need to be interpreted carefully considering the broader background environmental conditions at the time of the survey. Of the other sediment parameters typically used in monitoring programs, the $d^{15}N$ isotopic signature of sediments is emerging as a reasonably sensitive indicator of enrichment levels, particularly in the more marine locations where the contrast with the background (marine) signature has increased due to changes in feed composition with greater terrestrial inputs.

A key aim of the local scale study was to better understand the recovery responses of the sediments to farm management strategies such as fallowing. Where the surveys captured fallowing periods at the southern leases the responses were largely consistent with expectations. Sediment chemistry and function responded relatively rapidly to fallowing, and there was typically a decrease in the abundance of opportunistic species and a concomitant increase in the abundance of the less tolerant taxa close to the cages. The response to fallowing in Macquarie Harbour was more complex due to the influence of variable bottom water dissolved oxygen concentrations. Recovery appeared slower at sites further into the harbour where bottom water oxygen levels were lower and less affected by oceanic recharge, and at leases that had a longer history of farming or where production levels were more intense. This highlights the complex interplay between two key factors that govern the responsiveness of sediments and macrofaunal communities to fallowing: farm production levels and oxygen supply.

Importantly, the results of this study further validated the visual assessment methods of sediment health developed by Macleod and Forbes (2004) in new growing regions. When site-specific criteria and weightings were employed, the visual health scores could differentiate impact levels in all study regions. Using the same footage as currently required for monitoring but evaluating the full suite of criteria provides a more holistic and informative measure of sediment health at very little extra cost.

Implications for relevant stakeholders/ Recommendations

The guidelines provided by Macleod and Forbes (2004) remain a useful basis to inform management on the level of impact and recovery. However, given differences in environmental conditions and responses between some sites and regions, notably in Macquarie Harbour, there needs to be a level of site and region specificity to the guidelines and their interpretation. One of the most significant findings was the validation of the visual assessment methodology. The method could clearly discern different impact levels in all the study regions, and we recommend that it be considered as more holistic measure of sediment health to aid interpretation of benthic compliance and regulation. Of the other metrics used for environmental monitoring, our findings confirm evaluation of macrobenthic community structure is the most sensitive, reliable, and informative measure of sediment conditions, and as such should remain an essential element of both baseline assessments and regulatory monitoring. Measures of sediment redox and sulphide also remain fit for purpose for monitoring but provide more location dependent measures of the enrichment footprint.

Considering the inherent variability in environmental conditions and response to enrichment in both space (within and between regions) and time (surveys) this study reinforces the importance of establishing robust baseline conditions and ongoing monitoring of reference conditions. As such, we recommend environmental standards should focus on change relative to baseline and reference conditions rather than a suite of standardised/ fixed parameters and that reference locations should be further from the farms (i.e. >500 m in the south and >1000 m in Macquarie Harbour) than previously considered given the larger footprints evident from this study. We also suggest t further consideration be given to the level of change that defines ‘unacceptable’ when measured at compliance locations 35 m from the lease boundary.

The larger benthic footprints shown in the study highlight the importance of lease size and the layout (e.g. proximity of cage grids to each other and lease boundaries) for environmental management. We suggest maintaining separation between cage grids, and from cage grids to the lease boundary will help minimise the overlap of enrichment footprint, facilitating faster recovery and reduce effects beyond the lease boundary.

Fallowing remains an important management practice to facilitate the recovery of sediments. Our findings reinforce the importance of both the period and spatial scale of fallowing in promoting sediment health and assimilative capacity, particularly in locations/situations where enrichment footprints are larger and overlapping and/or where environmental conditions influence recovery rates (e.g. bottom water oxygen).

Reef Interactions

This component of the research sought to evaluate the main ways in which salmon farming might affect local reef systems, identify potential indicators of change in reef systems because of those interactions and recommend appropriate monitoring approaches.

Methods

This section of the project had four major components; a global review on the effects of organic enrichment on reefs more generally and the potential for interactions with salmon farming, a review on established methods for monitoring reef ecosystem, the implementation of established methods and the development of novel methods targeted for the detection of organic enrichment on reef systems.

The review on the effects of organic enrichment highlighted pathways of interaction that need to be considered as part of a monitoring program. These potential indicators of change in reef systems in Tasmania were assessed through both established methods such as biodiversity surveys identified through the methods review, as well as the development of new methods that were fit-for-purpose for monitoring for the effects of organic enrichment. Biodiversity on reefs throughout the Storm Bay and D’Entrecasteaux Channel regions were assessed using the approach developed by Edgar and Barrett. The results of that assessment were used to inform the development and subsequent trial of a novel rapid visual assessment (RVA) technique which focused on ecosystem function rather than species diversity. Towed video was also tested as an alternative, fully remote, approach to assess key species distribution and abundance, and the findings compared with that of both the Edgar - Barrett approach and the RVA technique.

This study also examined the potential to use change in specific indicators and colonisation patterns on settlement plates as a proxy measure of adverse impacts. Abalone settlement plates, cryptic species assemblages and epiphyte plates were all evaluated to determine their effectiveness as potential indicators.

Key findings

The review highlighted there is the potential for dissolved and particulate nutrient inputs from salmon farming to interact with reef ecosystems in several ways, both directly and indirectly. Outcomes vary depending on environmental conditions; the level of wave exposure and water movement will influence not only the basic ecology but can also affect the response to farm derived particulate and dissolved inputs. Understanding how reef systems respond to inputs from salmon farming requires a detailed understanding of the local environment, the broader regional and global pressures, and the inherent characteristics of the reef community itself. These factors need to be considered carefully in the design and implementation of any monitoring programs and in the interpretation of the resultant data.

The review of monitoring approaches for rocky reefs identified two methods that could be applied in Tasmania: the visual underwater census of Edgar-Barrett or the modified Reef Life Survey version of this. Both provide a detailed characterisation of reef ecosystems and can establish reliable baselines with which to assess background variability and compare and review change over time. However, the need for a more direct assessment of parameters that respond to nutrient enrichment and can be applied across a greater spatial area was also identified. As such we developed the RVA technique designed to detect a loss of resilience in reef ecosystems in response to organic enrichment.

The Edgar-Barrett surveys identified a set of key species that clearly defined the reef communities within the northern Bruny Island and southern Channel regions. These species determined the community composition and underpinned ecosystem function. Consequently, assessment of canopy cover, sub-canopy and substrate were identified as key components for inclusion in the RVA technique. Epiphytic, filamentous and nuisance algal species known to be indicative of organic enrichment were also included in the RVA's.

In the southern Channel region the RVA method was able to detect a broadscale enrichment gradient, reflecting the long-term combination of urban, agricultural and industrial inputs into this region, showing the potential of this approach as a simple and dependable tool to monitor temperate reefs. Further work is needed to i) examine the performance of the RVA surveys over a longer time series, ii) assess the responsiveness of the RVA technique to acute impact/ enrichment gradients and iii) determine if the RVA method can be used to evaluate recovery following the removal of an enrichment source. This work is already underway.

Towed video enabled a substantial increase in spatial coverage, but the number of parameters able to be evaluated and the accuracy with which they could be assessed was markedly reduced. Canopy cover was the only variable that could be determined with any level of certainty, but only relatively large changes in cover could be detected due to the categorical nature of the video scoring. At this stage, the method is not deemed reliable enough to detect low to moderate levels of organic enrichment or meaningful changes in ecosystem resilience. However, it may still be useful to confirm, or perhaps spatially extrapolate, the findings of either the RVA or Edgar-Barrett approaches. For example, if either the RVA or Edgar-Barrett surveys indicated canopy loss at an assessment site, towed video could be used to determine whether this was localised or more regional in nature. The value of other remote assessment techniques (i.e. ROV or AUV) to validate or extend the findings of more detailed diver-based approaches needs to be investigated further.

The assessment of specific indicators, on abalone settlement plates, cryptic species assemblages and epiphyte plates were found not to be useful as they were too inherently variable to reliably detect change either between sites or over time.

Although this study was primarily focused on developing methods to monitor the impacts of salmon farming on reef ecosystems, the results also provided important insights into how reef ecosystems interact with organic enrichment from farming. Proximity to the source of enrichment is well established as a key determinant of the likely impact on reef ecosystems (Oh et al. 2015). In this study there was no evidence of any direct effects of organic enrichment on reefs more than 1.5 km from fish farms. A functional response to organic enrichment was detected at sites closer than 1.5km to fish farms in the Southern D'Entrecasteaux Channel, but it was not possible to say for certain that this was due to fish farms as these sites were exposed to other sources of nutrients (both natural and anthropogenic). Robust baseline data and a long time series are clearly important prerequisites if there is an intent to better understand causal relationships and attribution of change on reef ecosystems.

Implications for relevant stakeholders/ Recommendations

Overall, this study highlights the value of both biodiversity and targeted functional RVA approaches as complementary tools for monitoring reef ecosystems and for assessing the potential impacts from organic enrichment. We would propose the adoption of an adaptive, stratified monitoring program that combines regular but less frequent (e.g. every 5-7 years) Edgar-Barrett biodiversity surveys with more frequent (biannual) RVA's to target specific measures of reef function. Surveys should be undertaken using both methods before farming commences to establish baseline conditions. Given the inherent variability in the system, baseline as well as ongoing surveys should be conducted at multiple sites at varying distances from the farm(s). Distance from source is often a good proxy for determining the level of exposure to nutrients and has long been used to inform location of monitoring sites; however, modelling tools that consider not just distance but water movement (e.g. hydrodynamics) to predict the potential impact of nutrients could also be used to inform reef sampling design. Modelling can also play a key role in the interpretation of reef assessment data and in informing possible cause and effect relationships.

An important outcome of the research is the identification of a range of condition parameters for ongoing monitoring of temperate reef ecosystems that reflect different stages of organic enrichment. In Table 1, we have listed the key parameters proposed for the RVA technique and provide examples of the types of thresholds that might be developed to support management. Whilst these thresholds provide an excellent start point, we acknowledge that they need to be informed and refined by a longer time series of data, and that further evaluation based on clearly defined impact gradients (i.e. change with distance or time from a measured source) would improve their reliability. This additional validation - akin to the process used to develop thresholds for soft sediments - will ensure we fully understand how the proposed parameters respond to changing levels of organic input, both as farming intensity increases and decreases.

For a monitoring program to be effective it must be able detect a “meaningful” change in reef condition. We have recommended an approach that can provide both an early warning of impact, based on the response of a suite of selected enrichment indicators, and a more detailed understanding of the nature and significance of potential impacts through a broader biodiversity assessment (Table 2). Determining causality is not trivial and will ultimately rely on a weight of evidence approach that includes the results from all of the proposed monitoring tools. An additional understanding of the potential inputs (loads) to the system, i.e. robust baseline data, ongoing time series data, production/ source data, modelling along with information from associated monitoring programs and reference stations (e.g. BEMP water quality, IMOS oceanographic products) may be required if the intent is to understand the source and fate of dissolved and particulate materials (from multiple sources, both natural and anthropogenic, such as rivers, fish farms, WWTPs).

Table 1 Suggested Rapid Visual Assessment (RVA) parameters and some indicative response criteria for further evaluation.

RVA Observation Parameter	Indicative Threshold Levels/ Response Criteria and Actions
Total canopy cover (including breakdown of species)	CRITERIA • Significant decline in canopy cover OR decline in canopy cover of key species – triggers additional survey/immediate review • Decline in total canopy cover relative to long-term mean (3-5yrs) noted for review at next survey • OR decline in total canopy cover in consecutive surveys ACTION Review overall response in context of production and broader environmental information. Response to criteria should be refined as more data becomes available with a view to establishing more site-specific targets and triggers.
Epiphytic & filamentous algal cover	CRITERIA • Significant increase in cover of both epiphytic and filamentous algae– – triggers additional survey/ targeted assessment. • Increase in cover of epiphytic algae and/ or filamentous algae relative to long-term seasonal mean (3-5yrs)– noted for review at next survey. ACTION Review overall response in context of production and broader environmental information. Response to criteria should be refined as more data becomes available with a view to establishing more site-specific targets and triggers.
Opportunistic red algae (e.g *Asparagopsis armata* in southern Tasmania) Opportunistic green algae (e.g. *Chaetomorpha billiardierii, Ulva* spp.)	CRITERIA • Significant increase in cover of opportunistic species – triggers additional survey/ targeted assessment. • Increase in cover of opportunistic species– noted for review at next survey.

	ACTION Review overall response in context of production and broader environmental information. Response to criteria should be refined as more data becomes available with a view to establishing more site specific targets and triggers.
Additional Parameters Worth Consideration	**These were considered important secondary response criteria which could inform assessments and provide supplemental understanding, and as such are worth monitoring.**
Sub-canopy green cover	Significant increase.
Sub-canopy red cover	Potential increase due to higher sedimentation in water column. Overall increase in red+green:brown algae ratio expected in enhanced nutrient conditions.
Pink encrusting algal cover	Potential decline and replacement by turfing or opportunistic algae if canopy is lost.
Red encrusting algal cover	Could decline as per pink encrusting or increase due to changes in predation pressure or light conditions.

Table 2 Summary of proposed monitoring criteria and application in adaptive management framework.

Proposed Monitoring Approach	Purpose	When	Adaptive Management Response/ Assessment Criteria	Exceedance of Investigative Threshold Levels
1. Edgar-Barrett biodiversity surveys	Comprehensive biodiversity assessment of reef system. Assessment of baseline conditions for assessments of regional differences and system wide change	Prior to commencement of farming and every 5-7 years	Review and comparison of each 5-7 yr dataset to assess long-term change in ecosystem condition.	Initiate exploratory studies comparisons indicate a significant change in key indicators (e.g. biodiversity and key species).
3. RVA surveys	To detect functional change, identify early warning indicators of change	Prior to commencement of farming and biannually	Regular review of condition parameters in relation to proximity and other drivers. Ongoing refinement and validation of monitoring criteria and thresholds	Review of data in the context of other available information to establish significance of observations. This would likely include the use of other available tools/evidence to determine causation, such as feed inputs and the used of CONNIE dispersion modelling described in chapter 7.3, comparison with patterns observed in other reef monitoring programs, and/or the investigation of local vs regional water quality parameters. If both RVA and Edgar-Barrett surveys suggest canopy loss may be widespread, a towed video assessment would be required to examine the spatial extent of the impact. Implement mitigation/ remediation plan in consultation with stakeholders.
4. Towed videos	To determine the spatial extent of any change/ impact observed using RVA/ Edgar-Barrett approaches.	As required.	Characterise spatial extent and scale of any observed change and inform management strategies.	Implement additional investigation and/or mitigation/ remediation plan in consultation with stakeholders

Modelling

Modelling environmental interactions with coastal salmon aquaculture in Tasmania has developed significantly over the past 15 years. This development spans the more complex high-level system wide biogeochemical models that require expertise in physical oceanography and/or biogeochemical modelling to implement and run to the simpler modelling tools designed to support aquaculture management decisions more readily. The latter require less expertise and are far less costly to implement and run. This component of the study sought to assess the performance of the modelling tools developed to predict the dispersion of dissolved and particulate farm waste and consider how these might best be applied in the Tasmanian context.

Methods

The commercially available model, DEPOMOD, is used globally for lease scale depositional modelling to determine the benthic footprint resulting from solid wastes exiting fish pens, and in recent years has been used regularly to support planning in Tasmania. The developers of DEPOMOD have recently released a new version, NewDEPOMOD, which offers improved modelling of resuspension and sediment transport processes. Neither the original nor new version of DEPOMOD has been validated for Tasmanian sites. The first stage of this project sought to compare the predicted depositional footprint of solid waste generated by each of these models at selected sites in each of the study regions (i.e. Lower Huon/ Channel, Storm Bay, and Macquarie Harbour) with empirical data. This comparison not only looked at the effectiveness of the models for predicting deposition but also extended the analysis to determine how well the models predicted benthic condition. DEPOMOD and NewDEPOMOD predict the direct deposition of particulate waste to the sediments (i.e. where the model is run with resuspension turned off) and the subsequent spread of the deposited material across the seabed (i.e. when the model is run with resuspension turned on). Sediment traps were deployed to provide an empirical measure of direct deposition (total load to the sediments) for model validation. This information was compared with a suite of condition indicators commonly used in benthic monitoring (e.g. sulphide, faunal abundance, and diversity) to assess the benthic response which encompasses the effects of both direct deposition and resuspension.

Simple dispersion models can provide valuable management information and insights on how farm wastes, debris or even pathogens might move around a body of water. The second modelling element of this study sought to validate the simplified nutrient dispersion tools currently available to managers in Tasmania and evaluate how those models could be employed to support management decisions regarding salmon farming interactions. CONNIE is a particle tracking tool developed by CSIRO available via a web interface. We also compared the predictions from CONNIE with the outputs of two hydrodynamic models, SETas and DHD. SETas was applied at the Storm Bay 1 lease whilst DHD was applied at Lippies and CONNIE was tested at both locations. The outputs of CONNIE were validated against empirical observations of nutrient levels collected in the proximity of the salmon farms

Key findings

Both versions of DEPOMOD performed well at predicting total sediment deposition (loads) at the Storm Bay 1 lease, with NewDEPOMOD performing well at the Franklin lease in Macquarie Harbour. NewDEPOMOD was designed with an updated sediment transport module that more realistically captures the flow of waste over dramatic shifts in bathymetry as seen at Franklin. At the Lippies lease in the Southern Channel neither of the two models accurately predicted total sediment deposition. We suggest this is because Lippies is a more dispersive site owing to the much higher bottom currents and deeper bathymetry. However, challenges with the sediment trap deployment may have also compromised our assessment at this site.

There were also differences between the study locations in how well the models predicted the overall benthic footprint (i.e. the area affected by both deposition and re-suspension), and this too would seem to be largely due to differences in bathymetry and current speeds. At the Storm Bay 1 lease where the seabed was quite level and bottom current speeds were relatively low, the predictions from both DEPOMOD and NewDEPOMOD were similar and closely resembled the measured benthic footprint. However, at Lippies where current speeds were higher and at Table Head in Macquarie Harbour where the bathymetry is steep, DEPOMOD and NewDEPOMOD predicted quite different footprints. NewDEPOMOD provided a more accurate assessment due to its improved ability to adjust for complex bathymetry and account for sediment resuspension at dispersive sites. Another improvement in NewDEPOMOD is the ability to incorporate a spatially variable current field. Although we suggest this is unnecessary at sites like Storm Bay 1 where the currents are reasonably consistent across the lease, it will likely improve predictions at a lease like Franklin where currents are variable in both space and time due to changes in bathymetry and the influence of river flows. In summary, we would recommend NewDEPOMOD as the most accurate approach for depositional modelling but would suggest further testing and development at dispersive sites and ongoing validation against empirical data, regardless of the choice of model.

The predicted spatial dispersion patterns were very similar for CONNIE and DHD / SETas. This is not surprising given CONNIE is underpinned by the output from the hydrodynamic model (STORM) which is almost identical to SETas but has a slightly higher resolution. However, the DHD /SETas models consistently predicted higher nutrient concentrations compared with the CONNIE model. This may reflect that SETas/DHD are not specifically designed to model outputs at this this scale with their coarser grid resolution. When assessed against empirical observations, the modelled dissolved nutrient (in this case ammonia) concentrations using CONNIE showed strong correlation with observations. Storm Bay was not as well correlated as the other sites, but it was the site at which the technique was first tested; the approach was improved over the latter two surveys at Lippies. The increased spatial replication in subsequent empirical sampling provided a more accurate representation of concentrations within the lease. Consequently, if the Storm Bay validation was repeated, using the improved sampling techniques, it is likely to show greater correlation. It is proposed dispersal modelling can potentially be used in a similar manner to depositional modelling to inform both site selection and location of environmental monitoring sites, and that water quality data currently collected across various monitoring programs could be used to calibrate and validate the modelling.

Implications for relevant stakeholders/ Recommendations

NewDEPOMOD is the most accurate approach for depositional modelling, but there were challenges with its application at more dispersive sites whereby the spatial extent of the footprint was often underestimated. This will be improved though calibration of resuspension at more dispersive sites and the concomitant adjustment to the default parameter values. Modelled deposition was also a reasonably good predictor of benthic change. However, the relationship between sediment load and the benthic response varied between locations i.e. there was no evidence of a threshold deposition level could be applied universally to infer benthic change. Consequently, validation with empirical data is recommended for application at new locations.

CONNIE was assessed as being more accurate than hydrodynamic modelling alone in reproducing near scale nutrient concentrations around salmon farm leases. CONNIE offers a relatively low-cost method to assess dispersal of dissolved waste, potential for interaction, and management at distances of 1-2 km, which is a scale not typically well resolved using existing BGC models. Establishing the relationship between dissolved nutrient concentration levels and responses in the near-field ecology would further increase the value of this approach as an ecological management tool. We propose that dispersal modelling can potentially be used in a similar manner to depositional modelling to inform both site selection and location of environmental monitoring sites. Water quality and habitat condition data collected through existing monitoring programs can be used to further calibrate and validate model predictions and ecological responses.

2 Acknowledgments

We first and foremost wish to acknowledge the considerable contribution of the project steering committee in the guidance, management, and review of this project (Matt Barrenger, Avril Brown, Julian Harrington, Josh Fielding, Jen Fry, Stephen Gallagher, Dean Lisson, Adam Main, Dianne Maynard, Depha Miedecke, David Morehead, Mark Nikolai, Dom O'Brien, Steve Percival, Grant Pullen, Sean Riley, Linda Sams, John Sansom, Graham Woods).

This research would not have been possible without the help of many others, especially the CSIRO Coastal Environmental Modelling Team, and in particular Bec Gorton and Scott Condie. In addition, we would like to specifically acknowledge the support provided by the team at Aquenal, especially Joe Valentine, Sam Gray and Jeremy Lane whose expertise and input to the reef studies helped significantly in the development of the sampling approach and played a key role in the outcomes.

We would also like to thank the staff at DPIPWE Marine Farming Branch and EPA for their support of this research, their expert insights and help in compiling additional farm information (particularly John Adams, Raymond Bannister, Damien Blackwell, Eric Brain, Greg Dowson, Ian Dutton, Wes Ford, Stephen Gallagher, David Horner, Dianne Maynard, Graham Woods).

The Tasmanian salmon farming industry are acknowledged for their support of the project in general, and for the provision of data, video and farm production information specifically. However, we would particularly like to thank the farm staff in each of the regions who facilitated our sampling on site, and on many occasions went above and beyond to help us when technical difficulties arose.

Most of all, the project team would like to acknowledge and thank the broader technical support team at IMAS who assisted in this project without whom it just wouldn't have been possible. In particular, Andrew Pender who played a key role in the co-ordination of the significant field and logistical support required for this study. Adam Davey and Jimmy Hortle who sorted the many, many samples - without their taxonomic expertise the local scale study would not have been possible. Jason Beard, Olivia Johnson, Sam Kruimink, Rebecca Chabot, and Benjamin Quigley all deserve a special mention, but there were many more staff, students and volunteers who contributed to this study, and we thank each and every one.

In conclusion, we would just like to acknowledge the informal contribution of the broader community to the focus of this work. We have always been mindful of the considerable public interest in this work and acknowledge that this has had an influence on the team and informed and guided some aspects of this project, particularly the interpretation of the findings as we sought to provide the understanding required. We sincerely hope that the findings will positively inform the public debate and the management of salmon farming in Tasmania.

3 Introduction

The salmon industry in Tasmania is currently in a phase of expansion, with plans to increase to a $1 billion a year industry by 2030 (DPIPWE 2017). To achieve this, the industry needs to consider a suite of alternate production approaches including improvements in farming practices, innovations in technology and expansion of the industry into new areas. Maintaining high environmental performance is a priority for both the aquaculture industry and its regulators. This requires that regulators and industry understand and have the potential to appropriately manage any variances in environmental interactions associated with differing farming approaches. The current standards for management and regulation of sediment impacts and recovery were established based on research conducted over 10 years ago (Macleod et al., 2004), and industry operations have moved on with both improved technologies and new farming locations. To help ensure that management and monitoring strategies remain best practice, and that farming operations continue to be efficient and sustainable, assessment of the sediment impact and recovery dynamics in these newly developed farming environments and under different farming practices/technologies was required. In addition, identifying how understanding farm impact and associated nutrient dynamics might be used to inform local, medium, and broad-scale interactions will provide an important basis for establishing an effective strategy for system-wide management and interactions with other users of the waterbodies.

It is clear from discussions with various resource users (i.e. fish farmers and commercial and recreational fishers) that the perception of potential risks differs between Macquarie Harbour and the southern farming regions. In Macquarie Harbour a critical issue is whether the current on-farm monitoring and local scale impact indicators are “fit for purpose” i.e. do they support sustainable management by providing an accurate understanding of sediment conditions.

In the new farming areas in the southern regions (Lower Channel/ Storm Bay), establishing the effectiveness of the local scale monitoring is still important but a key concern in these regions is whether there may be adverse effects on reef health because of increased aquaculture activities (i.e. off-site interactions). Therefore a key element of this study will be to provide a better assessment of the potential risk to reef systems from sediment deposition and nutrient dispersion from fish farms directly.

Ultimately, the aim of this project is to provide robust monitoring and management advice and strategies that consider the nuances of local farming environments. Providing “fit for purpose” management advice to government and industry that allows regionally structured optimisation of farm management, along with risk appropriate management and monitoring criteria in different locations. Providing an understanding of the ecological significance of any differences in observed effects across regions and how that information might be used to inform and improve site specific and regional modelling and management approaches. This will require a portfolio of research focussed on risk assessment, indicator development and benchmarking.

RISK ASSESSMENT Providing an understanding of regional variability in sediment processes/ recovery and benthic/ pelagic coupling associated with existing and newly developed farming operations and the resultant implications for any assumptions used in both monitoring and in predictive (biogeochemical and depositional) modelling.

INDICATOR DEVELOPMENT/ BENCHMARKING Comparing environmental monitoring and management approaches in existing and newly developed farming areas based on previous impact/ recovery understanding and associated monitoring and regulation criteria to establish those response principles that are common to all areas and those that are regionally specific (with reference to visual assessments and biological and biogeochemical measures).

This study is designed to target the different research priorities and concerns in each new farming region (Lower Huon/ Channel, Storm Bay, Mac Harbour). In Macquarie Harbour the emphasis will be on validating local scale monitoring approaches (on-site focus), and whilst this will also be an important element in the Southern regions a key element of the research in the south will be defining cost-effective and risk appropriate approaches for assessment of reef health (off-site interactions). An important component of this study will be to use empirical monitoring data to calibrate and validate sediment deposition and nutrient dispersion models. These models provide an important predictive tool for determining risk to the ecology of both soft sediment and reef habitats in new farming regions.

The original objectives of the project were defined as follows[1]:

1. Establish key recovery response principles and benthic condition criteria for all areas in which farming currently occurs - *building on existing understanding to identify both generic and regionally specific performance criteria*

2. Improve our understanding of sediment process interactions and recovery responses, in order to ensure that monitoring and management strategies are optimised for each growing region - *a key objective will be relating the findings to the most important ecological and resource interactions of salmon farming in each region*.

3. To evaluate the potential for interactions between local reef systems and salmon farming - *determining the main risk factors, recommending risk appropriate monitoring and assessment approaches and identifying risk mitigation strategies where relevant*.

4. To improve our understanding of how local scale (site based) environmental condition data, can integrate with local scale modelling to improve management outcomes - *a key goal will be identifying how local scale understanding of sediment processes and benthic pelagic interactions can inform and be informed by regional modelling and management approaches*.

The research undertaken in this study can be broken down into three key elements with the objectives addressed within each component.

Local Scale Monitoring

We will evaluate benthic environmental conditions in each new farming region. Sediment traps will also be used to measured sediment deposition rates. Samples will be collected for detailed assessment of sediment biogeochemistry, benthic ecology and nutrient flux dynamics at different distances from the source of enrichment (cage) in each region and the results compared with established performance metrics; this includes validation of visual assessment techniques in new regions. Nutrient fluxes will be analysed against more readily measurable parameters to investigate the potential for more cost-effective indicators of key nutrient cycling processes. The results will be used to establish whether the current biological indicators are “fit for purpose”, i.e. with the aim of identifying the most effective approach for management. The findings from this component of the study will address objectives 1, 2 and 4.

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[1] Please note, over the course of the study the emphasis of the research changed slightly, with a greater emphasis on defining regional variability in the scale and magnitude of the benthic response and the reliability of different measures of sediment condition. We also simplified the language in some of the other objectives, see the refined objectives below (section 4)

Reef Interactions

This component will involve targeted evaluation of potential indicators of change in reef health associated with nutrient and sediment inputs from fish farming and is a direct response to the concerns highlighted by the abalone, rock lobster and recreational fishing communities. This investigation will be undertaken at selected reef systems in the lower Channel and Storm Bay region, with specific sites selected based on a range of factors including but not limited to proximity to farming, representative reef communities, importance to fishing communities, previous research, and outcomes of specific hydrodynamic and depositional modelling.

Nutrient and sediment exposure on reefs will be characterised using a combination of modelling, targeted nutrient sampling and sediment traps. A range of measurements will look at the effects of enhanced nutrients on macroalgal communities, via a range of population and community level visual assessments (e.g. reef life survey design using divers/video and photo quadrats) of change.

An important component of this work will be to put potential reef changes in the context of broader system level changes due to other ecosystem drivers as well as changes in farming-derived nutrient inputs. Existing broad scale reef monitoring programs such as the long term MPA and Reef Life Survey provide an ideal basis for these assessments. Another important element of the reef work will be to contrast the utility and cost effectiveness of differing reef monitoring techniques (from more traditional to novel) and to make recommendations for future monitoring. This component of the study explicitly addresses objective 3 and provides additional insight into objectives 1 and 2.

Modelling

Benthic depositional modelling can provide an understanding of the nature and extent of the impact footprint associated with farming activities. We will look to relate this modelled output to the benthic community data and different stages of impact and recovery. This can then be used to calibrate farm-based sediment depositional (incl. resuspension) modelling for different farming regions and thereby provide a more robust decision support tool to assist planning and ongoing management.

Dispersion modelling will be used to link the reef assessment information to the local scale studies, specifically looking to identify the exposure of reef systems to nutrients and sediments from fish farms. Sediment trap measurements will be important for model calibration and validation. Ultimately, the deposition and dispersion models will provide an important predictive tool for determining risk to the ecology of soft sediment and reef habitats in new farming regions. This element addresses objective 4 but will also inform our understanding of sediment process interactions (objective 2) and the potential for interactions between local reef systems and salmon farming (objective 3).

Overall, the project research and outputs will be developed to align with monitoring and management measures currently in place and to connect with, and build on, broader ecosystem-based management and research. The proposed study will build on existing research and management understanding, seeking to inform and improve practices rather than replace current approaches.

The aim of this project is to improve understanding of how salmonid farming operations interact with their surrounding environment and other fisheries, and where there is the potential for negative interactions. Consequently, this project will provide benefits not just to the salmon industry in Tasmania but to all the associated users of the coastal ecosystems in which salmon farming is currently active. This research will improve our understanding of nutrient cycling and the potential for local scale eutrophication in and around salmon farms on a regional basis. This knowledge will help refine existing management strategies to ensure that salmon farming is managed sustainably in all regions, promoting best practice management, which is important to reassure other users (e.g. coastal fisheries, shellfish aquaculture and recreational fishers) that they can co-exists with salmon farming in the coastal zone.

Whilst the final report is laid out in a slightly different format to that envisaged when the project was first proposed, the results and findings still address the original objectives. The revised format has been necessary due to some important changes in the project focus, and additional research arising because of new priorities for management of salmon farming over the course of the project. The deterioration in environmental conditions in Macquarie Harbour occurred early in the project, and consequently, in consultation with the steering committee, we modified the research plan to incorporate additional sampling and support for a companion project (FRDC 2016/067: Understanding oxygen dynamics and the importance for benthic recovery in Macquarie Harbour) designed specifically to improve understanding of benthic conditions, sediment process interactions and recovery response (objectives 1 and 2) for Macquarie Harbour. In this final report we have taken account of the findings of the additional research to better inform decision making across all regions.

The revised format now has 3 main sections purposefully focused on the critical areas of management interest with respect to the interactions of salmon farming: i) local scale response and monitoring, ii) reef response and monitoring and iii) how to apply current modelling tools most effectively. Each section has a synthesis chapter at the end that summaries the results, discusses the implications for management and includes recommendations. The Local Scale Monitoring section provides an understanding of how benthic communities respond to farming inputs, how current monitoring approaches might be improved, and when regionally specific metrics and criteria are required (Objectives 1 & 2). The Reef Interactions section considers the potential for interactions between local reef systems and salmon farming and provides recommendations on optimised monitoring strategies and condition assessment criteria (Objective 3). The section on Modelling uses the empirical data and assessments to calibrate and validate existing models used in management for predicting sediment deposition and nutrient dispersion from salmon farms (Objective 4). Collectively this provides the information necessary to optimise prediction, monitoring and management strategies for each growing region.

4 Objectives

The original objectives of the project were defined in the proposal as follows:

1: Establish key recovery response principles and benthic condition criteria for all areas in which farming currently occurs - *building on existing understanding to identify both generic and regionally specific performance criteria*

2: Improve our understanding of sediment process interactions and recovery responses, in order to ensure that monitoring and management strategies are optimised for each growing region - *a key objective will be relating the findings to the most important ecological and resource interactions of salmon farming in each region*.

3: To evaluate the potential for interactions between local reef systems and salmon farming - *determining the main risk factors, recommending risk appropriate monitoring and assessment approaches and identifying risk mitigation strategies where relevant*.

4: To improve our understanding of how local scale (site based) environmental condition data, can integrate with local scale modelling to improve management outcomes - *a key goal will be identifying how local scale understanding of sediment processes and benthic pelagic interactions can inform and be informed by regional modelling and management approaches*.

However, over the course of the study the emphasis of the research changed slightly, including a greater emphasis on defining regional variability in the scale and magnitude of the benthic response and the reliability of different measures of sediment condition. We also simplified the language in some of the other objectives, as such the following more accurately represent the final focus of the research.

1: Establish the key sediment responses and associated assessment criteria for all areas in which farming currently occurs, building on our existing understanding to identify both generic and regionally specific performance criteria.

2: Improve our understanding of how sediments process farm waste across different levels of production and in different growing areas. This will help to ensure that sediment assessment criteria, used for monitoring and management, are appropriate and optimised for each growing region.

3: Evaluate the potential for interactions between local reef systems and salmon farming, determine the risk of direct impacts, recommend monitoring and assessment approaches appropriate to the level of risk, and identify mitigation strategies where relevant.

4: Assess the performance of modelling tools that have been developed specifically to support management decision making in predicting the dispersion of farm waste (dissolved and particulate) and consider how these might best be applied in the Tasmanian context (generally and regionally).

5 Local Scale Monitoring

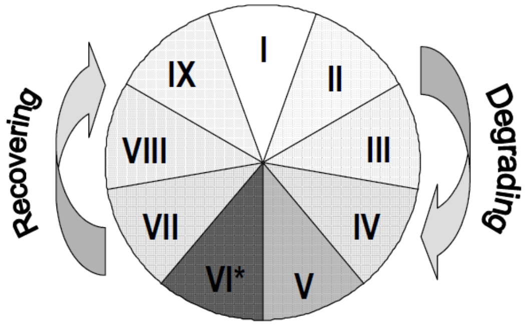
Assessing sediment responses in new growing areas to ensure that monitoring and the indicators of impact are fit-for-purpose and best practice

5.1 Introduction and Methods

5.1.1 Introduction

Marine farming development plan management controls stipulate “there must be no significant visual, physio-chemical or biological impacts at or extending 35 metres from the boundary of the lease area” The 35 m point from the boundary was based on both studies from Europe where particulate farm wastes were generally found to be concentrated within 35 m of the edge of the cage (Ye et al. 1990), and preliminary research conducted in Tasmania. Extensive research in Tasmania has since documented the extent of benthic affects associated with particulate farm waste (e.g., Ritz et al. 1989; McGhie et al. 2000; Crawford et al. 2002; Macleod et al. 2004; Edgar et al. 2005) and confirmed a distinct gradient of impact: from significant signs of enrichment immediately adjacent to cages to minor farm effects evident at sites 35 m from the lease boundary. This included an extensive review of the results of the biological, chemical, and physical data collected at the 35 m compliance points and control sites as part of the regulatory monitoring program from 1997-2003 (Woods et al. 2004). Based on the review and the demonstrated level of compliance, ongoing monitoring was designed to predominately use video evidence to detect unacceptable impacts. Significant visual impact/s is detected at any point 35 metres or more from the lease boundary may trigger an environmental survey to determine the extent of any breaches and whether the impact (s) are a result of marine farming operations.

Macleod and colleagues at IMAS (then TAFI) investigated the relationship between farm management practices, including fallowing and the level of impact and potential for recovery (Crawford et al. 2001; Macleod et al. 2002, 2004). The research found a clear relationship between farm management practices and level of impact, and identified nine distinct stages of sediment condition, encompassing both degradation and recovery stages, which farmers could use to classify sediment condition and inform management accordingly (Figure 5.1-1, Macleod et al. 2004). This research also highlighted regional differences in the key biotic and abiotic indicators of impact/recovery stages based on substrate type between the more exposed sandy and sheltered mud site used in the study (see Table 5.1-1). Although the field guide produced from this research (Macleod and Forbes 2004) was designed specifically to assist with farm monitoring and management, the findings were also used to inform regulatory monitoring and compliance requirements. Notably, the key features/indicators and thresholds associated with the benthic impacts were used to inform the development of environmental standards to allow for the assessment of unacceptable biological and physico-chemical impacts (i.e., nonvisual impacts) in Marine Farm Licence Conditions (now Environmental Licence Conditions)(Figure 5.1-2). This research also further validated and refined the use of visual assessment techniques. Using a suite of indices based on easily identifiable visual criteria which relate to different stages of impact, an objective scoring system of sediment condition was developed. Although the ROV footage collected during monitoring is generally only assessed against the compliance criteria that define a significant visual impact (e.g., presence of feed pellets or bacterial mats), the footage can be scored using the indices developed by Macleod and Forbes (2004) to provide a more detailed and sensitive assessment of sediment condition and the stage of impact/recovery.

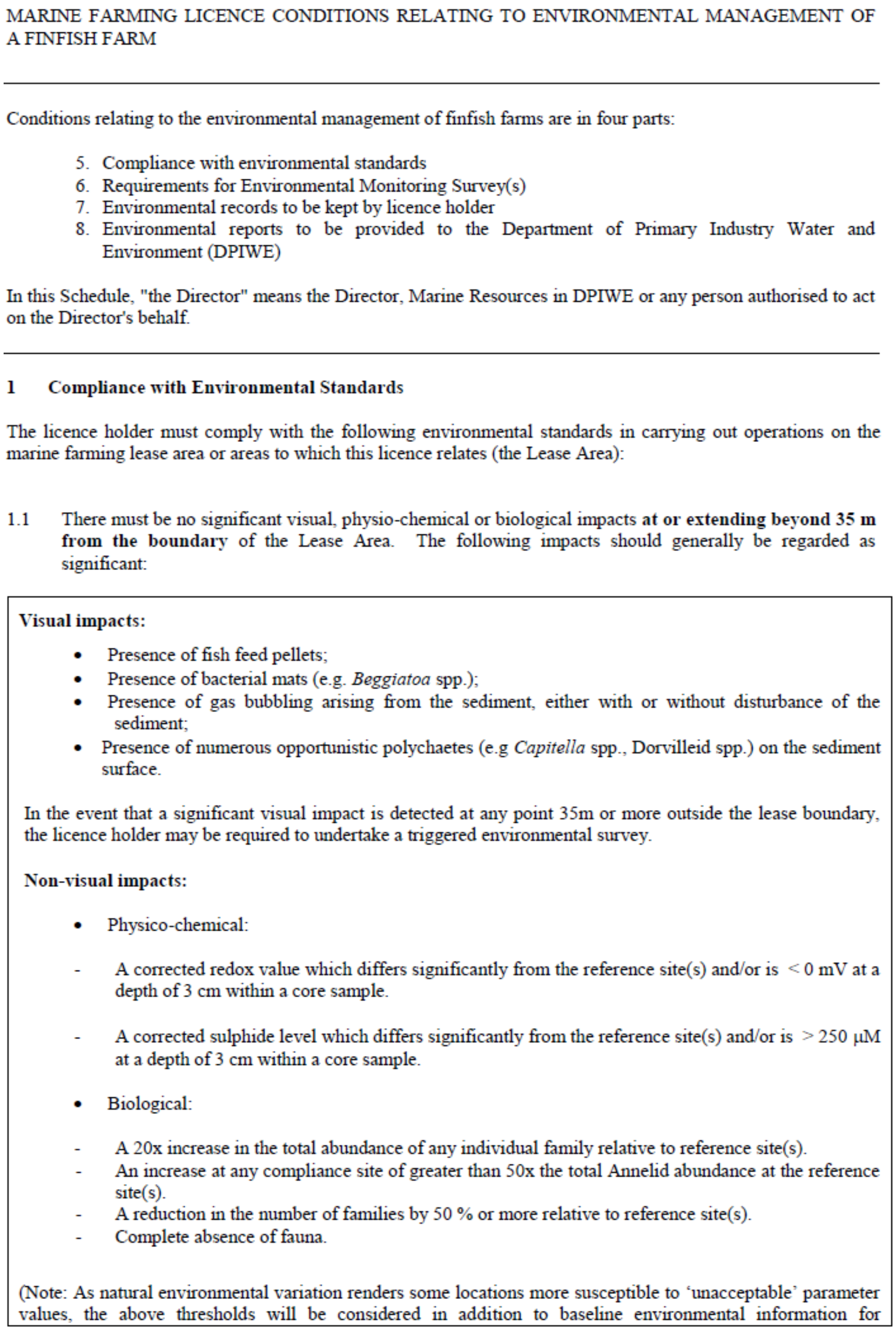


* Indicates conditions not observed in this study
Suggest stage IX is sufficiently recovered for restocking

STAGE – Category		STAGE – Description	
I	- Unimpacted	I	- No evidence of farm impact
II	- Minor Effects	II	- Slight infaunal & community change observed
III	- Moderate Effects	III	- Clear change in infauna & chemistry
IV	- Major Effects (1)	IV	- Major change in infauna & chemistry
V	- Major Effects (2)	V	- Bacterial mats evident, outgassing on disturbance
VI*	- Severe Effects	VI*	- Anoxic/ abiotic, spontaneous outgassing
VII	- Major Effects	VII	- Monospecific fauna, major chemistry effects
VIII	- Moderate Effects	VIII	- Fauna recovering, chemistry still clearly effected
IX	- Minor Effects	IX	- Largely recovered, although slight faunal/ chemical effects still apparent

Figure 5.1-1 Impact and recovery stages (from Macleod et al. 2004)

In the time since this research was conducted the industry has continued to grow and evolve. This includes the development of new production approaches (e.g., larger cages/pens, more efficient and automated feeding systems, increased smolt sizes) and expansion into new areas. Maintaining high environmental performance requires an understanding of how both farming in new areas/environments and changes in farming practices might change how farming interacts with the marine environment. To ensure that management remains best practice and farms continue to be efficient and sustainable, assessment of the local scale impacts and recovery dynamics is required in newly developed farming environments and under different farming technologies. The expansion of farming in Macquarie Harbour and the benthic response observed provided a very timely reminder for this need. The monitoring approach initially implemented in Macquarie Harbour was based on extensive understanding of sediment interactions with farming practices in the established farming regions of southern Tasmania. However, the results of a tactical FRDC project (2014- 038) demonstrated that the response of benthic communities in Macquarie Harbour differed from that observed in southern Tasmania, highlighting the need to validate local scale monitoring approaches. This study was designed to assess the local scale interactions in the two new growing areas and environments in southeast Tasmania (Storm Bay and the Southern Channel) as well as Macquarie Harbour, with the intent to ensure monitoring and the indicators of impact remain fit for purpose.



MARINE FARMING LICENCE CONDITIONS RELATING TO ENVIRONMENTAL MANAGEMENT OF A FINFISH FARM

Conditions relating to the environmental management of finfish farms are in four parts:

5. Compliance with environmental standards
6. Requirements for Environmental Monitoring Survey(s)
7. Environmental records to be kept by licence holder
8. Environmental reports to be provided to the Department of Primary Industry Water and Environment (DPIWE)

In this Schedule, "the Director" means the Director, Marine Resources in DPIWE or any person authorised to act on the Director's behalf.

1 Compliance with Environmental Standards

The licence holder must comply with the following environmental standards in carrying out operations on the marine farming lease area or areas to which this licence relates (the Lease Area):

1.1 There must be no significant visual, physio-chemical or biological impacts **at or extending beyond 35 m from the boundary** of the Lease Area. The following impacts should generally be regarded as significant:

Visual impacts:

- Presence of fish feed pellets;
- Presence of bacterial mats (e.g. *Beggiatoa* spp.);
- Presence of gas bubbling arising from the sediment, either with or without disturbance of the sediment;
- Presence of numerous opportunistic polychaetes (e.g *Capitella* spp., Dorvilleid spp.) on the sediment surface.

In the event that a significant visual impact is detected at any point 35m or more outside the lease boundary, the licence holder may be required to undertake a triggered environmental survey.

Non-visual impacts:

- Physico-chemical:

- A corrected redox value which differs significantly from the reference site(s) and/or is < 0 mV at a depth of 3 cm within a core sample.

- A corrected sulphide level which differs significantly from the reference site(s) and/or is > 250 µM at a depth of 3 cm within a core sample.

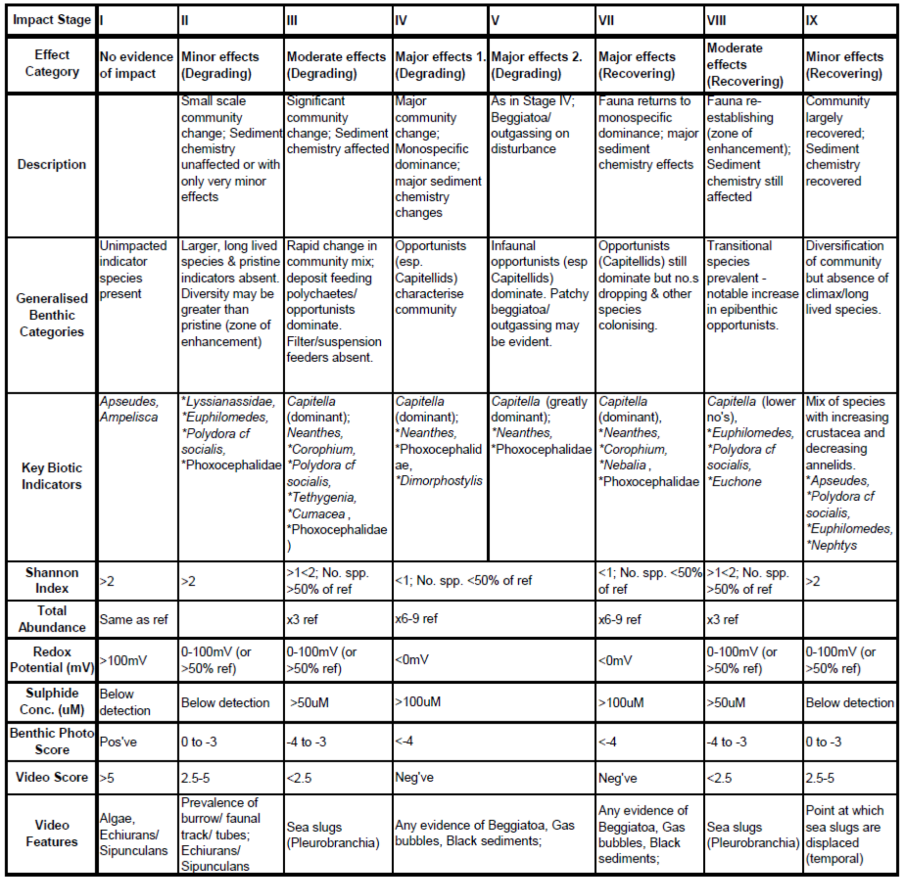
- Biological:

- A 20x increase in the total abundance of any individual family relative to reference site(s).
- An increase at any compliance site of greater than 50x the total Annelid abundance at the reference site(s).
- A reduction in the number of families by 50 % or more relative to reference site(s).
- Complete absence of fauna.

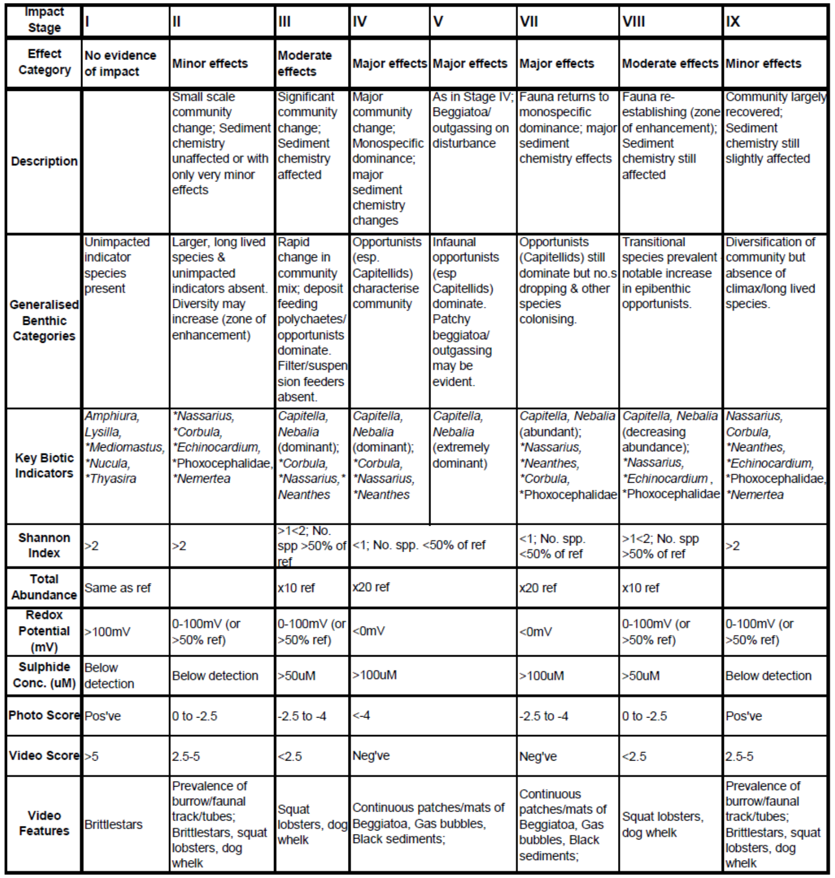
(Note: As natural environmental variation renders some locations more susceptible to 'unacceptable' parameter values, the above thresholds will be considered in addition to baseline environmental information for

Figure 5.1-2 Schedule 3 to Marine Farm Licence Conditions (from Woods et al. 2004).

Table 5.1-1 Summary of features characterising impact/recovery stages at the exposed/sand site (left) and sheltered/mud site (right) based on key features for each of the techniques deemed suitable for farm based assessment from Macleod et al. (2004b). NB. Key Biotic Indicators row: organisms identified with * are indicative in combination rather than individually.



Impact Stage	**I**	**II**	**III**	**IV**	**V**	**VII**	**VIII**	**IX**
Effect Category	**No evidence of impact**	**Minor effects (Degrading)**	**Moderate effects (Degrading)**	**Major effects 1. (Degrading)**	**Major effects 2. (Degrading)**	**Major effects (Recovering)**	**Moderate effects (Recovering)**	**Minor effects (Recovering)**
Description		Small scale community change; Sediment chemistry unaffected or with only very minor effects	Significant community change; Sediment chemistry affected	Major community change; Monospecific dominance; major sediment chemistry changes	As in Stage IV; Beggiatoa/ outgassing on disturbance	Fauna returns to monospecific dominance; major sediment chemistry effects	Fauna re-establishing (zone of enhancement); Sediment chemistry still affected	Community largely recovered; Sediment chemistry recovered
Generalised Benthic Categories	Unimpacted indicator species present	Larger, long lived species & pristine indicators absent. Diversity may be greater than pristine (zone of enhancement)	Rapid change in community mix; deposit feeding polychaetes/ opportunists dominate. Filter/suspension feeders absent.	Opportunists (esp. Capitellids) characterise community	Infaunal opportunists (esp Capitellids) dominate. Patchy beggiatoa/ outgassing may be evident.	Opportunists (Capitellids) still dominate but no.s dropping & other species colonising.	Transitional species prevalent - notable increase in epibenthic opportunists.	Diversification of community but absence of climax/long lived species.
Key Biotic Indicators	_Apseudes, Ampelisca_	*_Lyssianassidae_, *_Euphilomedes_, *_Polydora cf socialis_, *Phoxocephalidae	_Capitella_ (dominant); _Neanthes_, *_Corophium_, *_Polydora cf socialis_, *_Tethygenia_, *_Cumacea_ , *Phoxocephalidae)	_Capitella_ (dominant); *_Neanthes_, *Phoxocephalidae, *_Dimorphostylis_	_Capitella_ (greatly dominant); *_Neanthes_, *Phoxocephalidae	_Capitella_ (dominant), *_Neanthes_, *_Corophium_, *_Nebalia_ , *Phoxocephalidae	_Capitella_ (lower no's), *_Euphilomedes_, *_Polydora cf socialis_, *_Euchone_	Mix of species with increasing crustacea and decreasing annelids. *_Apseudes_, *_Polydora cf socialis_, *_Euphilomedes_, *_Nephtys_
Shannon Index	>2	>2	>1<2; No. spp. >50% of ref	<1; No. spp. <50% of ref		<1; No. spp. <50% of ref	>1<2; No. spp. >50% of ref	>2
Total Abundance	Same as ref		x3 ref	x6-9 ref		x6-9 ref	x3 ref	
Redox Potential (mV)	>100mV	0-100mV (or >50% ref)	0-100mV (or >50% ref)	<0mV		<0mV	0-100mV (or >50% ref)	0-100mV (or >50% ref)
Sulphide Conc. (uM)	Below detection	Below detection	>50uM	>100uM		>100uM	>50uM	Below detection
Benthic Photo Score	Pos've	0 to -3	-4 to -3	<-4		<-4	-4 to -3	0 to -3
Video Score	>5	2.5-5	<2.5	Neg've		Neg've	<2.5	2.5-5
Video Features	Algae, Echiurans/ Sipunculans	Prevalence of burrow/ faunal track/ tubes; Echiurans/ Sipunculans	Sea slugs (Pleurobranchia)	Any evidence of Beggiatoa, Gas bubbles, Black sediments;		Any evidence of Beggiatoa, Gas bubbles, Black sediments;	Sea slugs (Pleurobranchia)	Point at which sea slugs are displaced (temporal)



Impact Stage	**I**	**II**	**III**	**IV**	**V**	**VII**	**VIII**	**IX**
Effect Category	**No evidence of impact**	**Minor effects**	**Moderate effects**	**Major effects**	**Major effects**	**Major effects**	**Moderate effects**	**Minor effects**
Description		Small scale community change; Sediment chemistry unaffected or with only very minor effects	Significant community change; Sediment chemistry affected	Major community change; Monospecific dominance; major sediment chemistry changes	As in Stage IV; Beggiatoa/ outgassing on disturbance	Fauna returns to monospecific dominance; major sediment chemistry effects	Fauna re-establishing (zone of enhancement); Sediment chemistry still affected	Community largely recovered; Sediment chemistry still slightly affected
Generalised Benthic Categories	Unimpacted indicator species present	Larger, long lived species & unimpacted indicators absent. Diversity may increase (zone of enhancement)	Rapid change in community mix; deposit feeding polychaetes/ opportunists dominate. Filter/suspension feeders absent.	Opportunists (esp. Capitellids) characterise community	Infaunal opportunists (esp Capitellids) dominate. Patchy beggiatoa/ outgassing may be evident.	Opportunists (Capitellids) still dominate but no.s dropping & other species colonising.	Transitional species prevalent - notable increase in epibenthic opportunists.	Diversification of community but absence of climax/long lived species.
Key Biotic Indicators	_Amphiura, Lysilla_, *_Mediomastus_, *_Nucula_, *_Thyasira_	*_Nassarius_, *_Corbula_, *_Echinocardium_, *Phoxocephalidae, *_Nemertea_	_Capitella, Nebalia_ (dominant); *_Corbula_, *_Nassarius_,* _Neanthes_	_Capitella, Nebalia_ (dominant); *_Corbula_, *_Nassarius_, *_Neanthes_	_Capitella, Nebalia_ (extremely dominant)	_Capitella, Nebalia_ (abundant); *_Nassarius_, *_Neanthes_, *_Corbula_, *Phoxocephalidae	_Capitella, Nebalia_ (decreasing abundance); *_Nassarius_, *_Echinocardium_ , *Phoxocephalidae	_Nassarius, Corbula_, *_Neanthes_, *_Echinocardium_, *Phoxocephalidae, *_Nemertea_
Shannon Index	>2	>2	>1<2; No. spp >50% of ref	<1; No. spp. <50% of ref		<1; No. spp. <50% of ref	>1<2; No. spp >50% of ref	>2
Total Abundance	Same as ref		x10 ref	x20 ref		x20 ref	x10 ref	
Redox Potential (mV)	>100mV	0-100mV (or >50% ref)	0-100mV (or >50% ref)	<0mV		<0mV	0-100mV (or >50% ref)	0-100mV (or >50% ref)
Sulphide Conc. (uM)	Below detection	Below detection	>50uM	>100uM		>100uM	>50uM	Below detection
Photo Score	Pos've	0 to -2.5	-2.5 to -4	<-4		-2.5 to -4	0 to -2.5	Pos've
Video Score	>5	2.5-5	<2.5	Neg've		Neg've	<2.5	2.5-5
Video Features	Brittlestars	Prevalence of burrow/faunal track/tubes; Brittlestars, squat lobsters, dog whelk	Squat lobsters, dog whelk	Continuous patches/mats of Beggiatoa, Gas bubbles, Black sediments;		Continuous patches/mats of Beggiatoa, Gas bubbles, Black sediments;	Squat lobsters, dog whelk	Prevalence of burrow/faunal track/tubes; Brittlestars, squat lobsters, dog whelk

5.1.2 Methods

5.1.2.1 Site Information

Storm Bay

The Trumpeter Bay Marine Farm Lease (MF261), operated by Huon Aquaculture Pty Ltd, is in Storm Bay in southeast Tasmania, approximately 1.5km seaward of Trumpeter Bay. In 2014 the Storm Bay off Trumpeter Bay North Bruny Island Marine Farming Development Plan (MFDP) Area was amended to establish four new zones south of Trumpeter Bay (hereafter referred to as Storm Bay leases 1-4). A further amendment in 2018 has seen the approval of a new marine farming zone approximately 1.5 km east of Yellow Bluff (hereafter referred to as East of Yellow Bluff). Sampling in this study focused on the Trumpeter Bay and Storm Bay 1 leases (Figure 5.1-3).

These leases are moderately protected from the prevailing westerly winds and more exposed to southeast winds and seas which are generally uncommon, especially during summer periods. Water depth ranges between 25-30 m in the Trumpeter lease and 35−45 m at Storm Bay 1. In accordance with Schedule 3B, baseline environmental surveys were conducted prior to commencement of aquaculture operations on each of the study leases. In August 2014, a baseline survey of the north eastern part of the Trumpeter lease was conducted prior to the commencement of farming operations in November 2014[1] (Aquenal 2014). The baseline surveys for Storm Bay leases 1 and 2 were conducted in July 2015, with farming operations commencing on lease 1 in April 2016. A second survey occurred in April 2016 prior to Huon Aquaculture's planned expansion into the south-eastern area of the lease (Storm Bay sites 1-4 ; Aquenal 2016). The sedimentary environment at both study leases was found to be typical of sediments in deep (>20 m) and exposed locations; fine sands dominated, and silt and clay fractions were low (Aquenal 2015).

Southern Channel

The East of Lippies Marine Farm Lease (MF78) operated by Tassal Pty Ltd is in the southern D'Entrecasteaux Channel on the coast between Scott Point and Tower Bay. The lease was originally established in 2004 but its position and size were amended in 2014 and farming began in late 2016. The area is dominated by saline oceanic waters with some freshwater influences from the Huon River system flowing south out of the D'Entrecasteaux Channel. As the area is situated in the open channel, the effects of severe weather from the south west are not moderated by land. Water depths across the area are relatively deep and range from 35 to 50 m. The lease area is dominated by silty sand substrate and is approximately 1 km from the shoreline (Figure 5.1-4).

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[1] In April 2016 a baseline survey of the south-eastern section of the lease was conducted prior to commencement of farming on that part of the lease (Aquenal 2016). The sampling in this study was focused on the north-eastern section of the lease where farming commenced in November 2014.

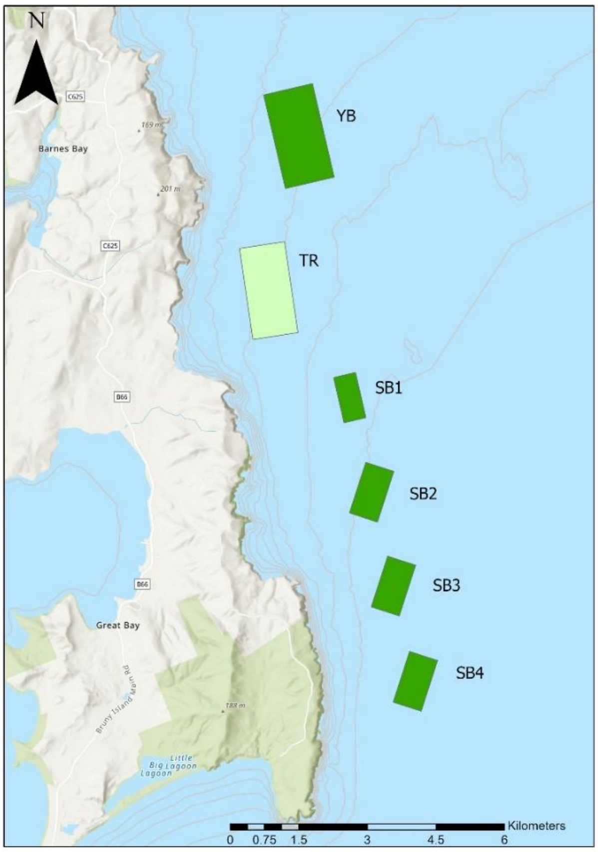

Figure 5.1-3 The location of the Trumpeter (TR) and Storm Bay (SB) leases 1-4.

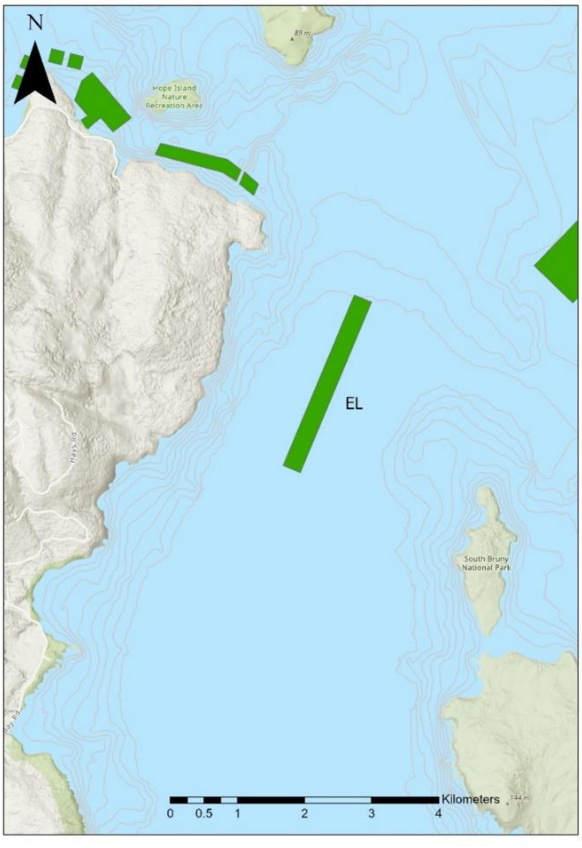

Figure 5.1-4 The location of the East Lippies (EL) lease in the southern channel.

Macquarie Harbour

The MFDP for Macquarie Harbour was approved in 1998, with production in the harbour growing to approximately 9,000 tonnes in 2011. In 2012 the Macquarie Harbour MFDP area was amended from 564 Ha to 926 Ha to facilitate industry expansion (Figure 5.1-5). Macquarie Harbour represents quite a different growing environment than elsewhere in the state. Located on the west coast of Tasmania the harbour has a shallow restricted entrance to the ocean which opens into a long deep basin with depths ranging from 0-50 m in the centre of the harbour. The water column is highly stratified due to significant freshwater flows from the Gordon and King Rivers, making the surface waters ideal for growing salmonids. However, the stratification and narrow entrance mean that water column exchange with the ocean is limited, particularly for bottom waters where dissolved oxygen concentrations are consistently low as a result. Productivity in the system is also considered to be low owing to the tannin rich freshwater layer limiting light penetration, and as such the sediments in the harbour are inherently depauperate of benthic invertebrates. However, changes in benthic condition in the harbour have been observed since mid-2013, with routine benthic monitoring showing an increase in the abundance of opportunistic polychaetes on the sediment surface in and around marine farming lease areas. To investigate this response, IMAS was commissioned to carry out targeted surveys at selected leases in early 2015 (Ross et al. 2016). The leases were chosen to represent operational activity, with each lease having a different farm history and location along the length of the harbour. For this study, surveys were

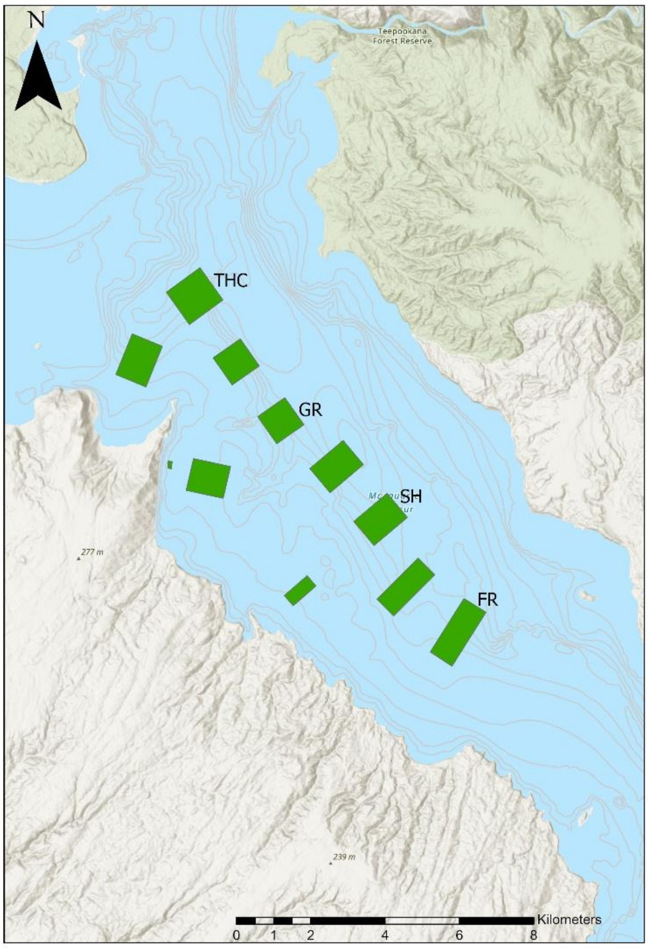

Figure 5.1-5 The location of the Table Head Central (THC), Gordon (GR), Strahan (SH) and Franklin (FR) leases in Macquarie Harbour

undertaken at the same four leases to enable us to access the broader dataset and support a more comprehensive interpretation. These leases were: MF133 Table Head Central (operated by Petuna) is the closest to the harbour entrance with depths ranging from 20-50 m; MH219 Gordon (operated by Tassal) and MH267 (operated by Huon) are situated mid harbour with depths ranging from 20-40 m and 35-42 m respectively, and MH266 Franklin (operated by Tassal), the lease furthest from the harbour mouth, has depths ranging from 25-36 m. MH219 has been farmed consistently since 2000, whereas the other 3 leases have only become operational since the amendment of the MFDP area was approved in 2012; farming commenced at lease MH133 and MMH267 in April 2013 and at MH266 in January 2014.

5.1.2.2 Survey Design

For all study leases, samples were collected along transects beginning at a stocked cage extending outwards from the lease. At the Storm Bay and Southern Channel leases, there were three transects with sites at 0 m (i.e. directly adjacent to the cage), 35 m, 100 m, 200 m, 500 m and 1000 m from the cage (Figure 5.1-8). In Macquarie Harbour[2], there were two transects at each lease, with sites at 0 m, 50 m, 100 m, 250 m, and 500 m from the cage. In addition, outer control sites were sampled (at least 1000 m from the cage) at all leases, and there were also inner control sites (100-300 m from cages) at the Storm Bay and Southern Channel leases. It is important to note that the sites along the transect were chosen to assess the effect of distance from the source of enrichment whilst minimising other sources of variability (e.g. changes in depth, sediment composition) whilst the control sites were included to compare the sediment conditions and benthic communities associated with farming against the broader ecology of the local area. The specifics of the sampling design for each of the regions are provided below.

Storm Bay

The Trumpeter (TR) and Storm Bay 1 (SB1) leases were surveyed in Winter 2016, Summer 2017, Winter 2017, and Summer 2018. The intention was to spread these surveys over different stages of the farming cycle (e.g. sampling at peak production and after fallowing) to see how the footprint changed and if recovery could be detected. We sampled every ~ 6 months to capture high and low points in production cycle for each lease (Figure 5.1-6). At the Trumpeter lease, the four surveys corresponded to late production (Winter 2016; 20 months after the fish were first introduced), end of the first production cycle/start of fallowing (Summer 2017; first month of the fallowing period), end of fallowing/start of the next production cycle (Winter 2017; 6 months after fish were harvested) and peak production (Summer 2018; 2 months following the production peak).

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[2] The survey and sampling design at Macquarie harbour was different to ensure consistency with a proceeding study (FRDC 2014-038).

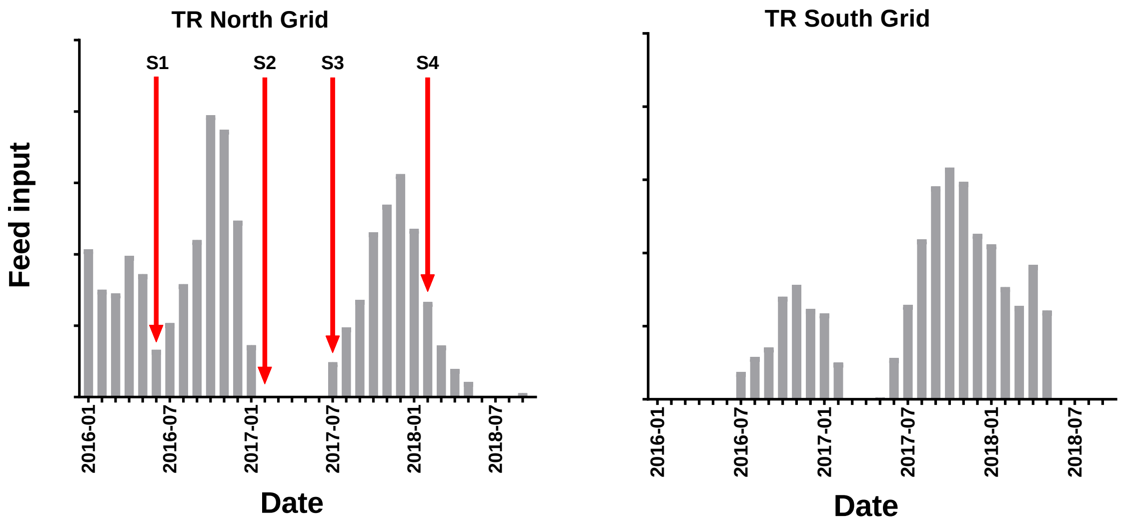

Figure 5.1-6 Monthly feed input into the north and south grid of the Trumpeter lease over the years 2016-2018, showing when surveys one to four (S1-S4) were completed in the current IMAS study. Note, all the transects were in the north grid where farming commenced earlier (Figure 5.1-8). To overcome the constraints of commercial in confidence considerations, feed information is provided in relative terms.

At the Storm Bay 1 lease, the transects were across two grids with different production cycles. On the northern grid the four surveys corresponded to the start of production (Winter 2016; 1 months after the fish were first introduced), mid production cycle (Summer 2017; 9 months into production), late production (14 months into production) and fallow (Summer 2018; 3 months after most fish were harvested). The southern grid was similar except that the first survey was 3 months into the production cycle and there was a 2-3-month fallow period between surveys 1 and 2 (Figure 5.1-7).

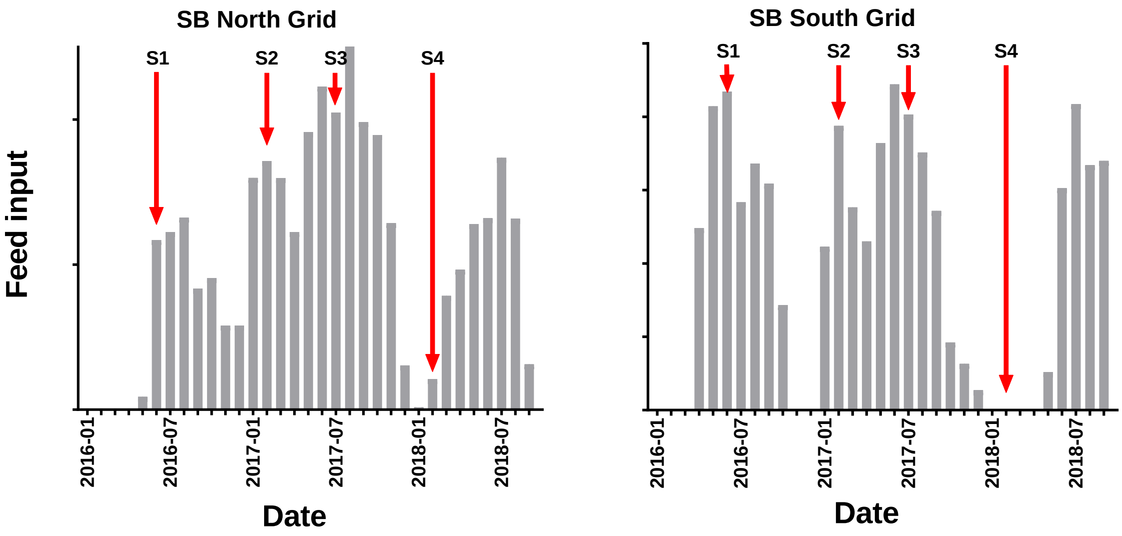

Figure 5.1-7 Monthly feed input into the north and south grid of the Storm Bay lease over the years 2016-2018, showing when surveys one to four (S1-S4) were completed in the current IMAS study. Note, the upper transect was on the north grid and the middle and lower transects on the south grid (Figure 5.1-8). To overcome the constraints of commercial in confidence considerations, feed information is provided in relative terms.

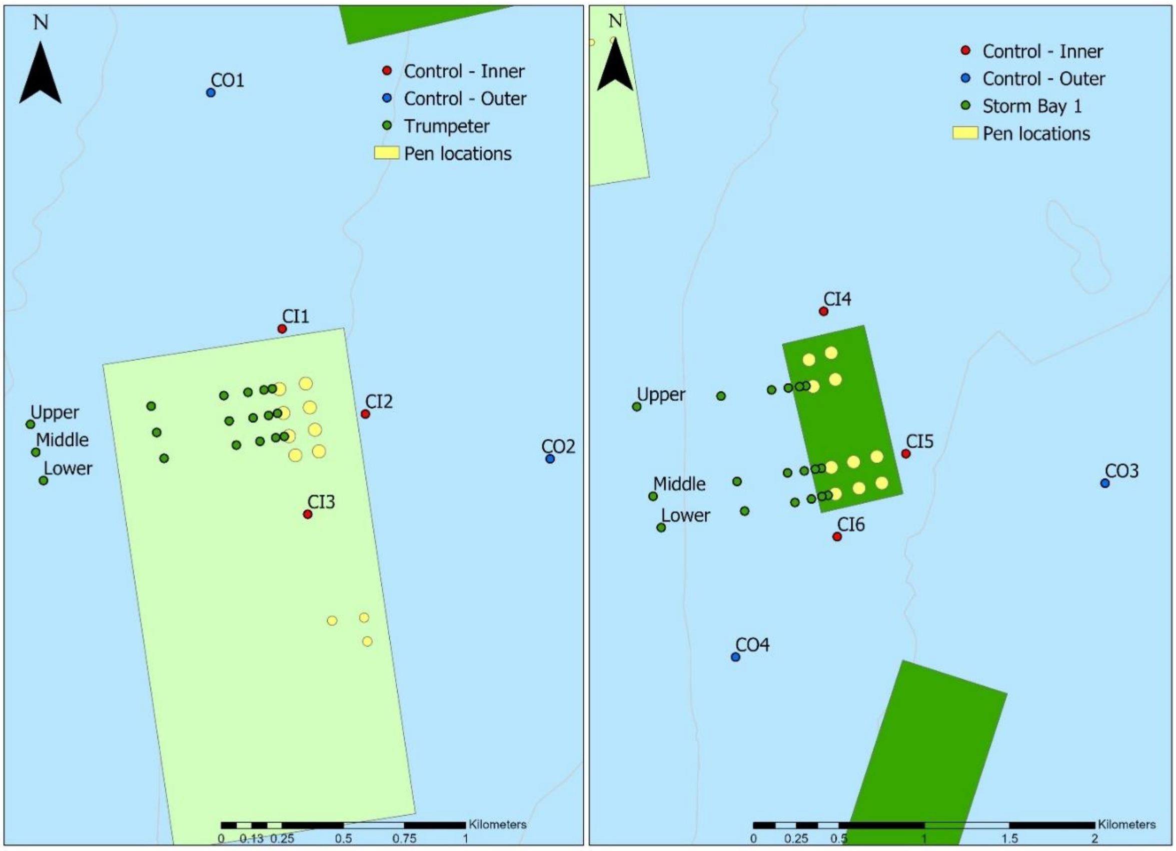

Figure 5.1-8 Map of the Trumpeter (left) and Storm Bay 1 (right) leases showing sampling site locations. Upper, Middle and Lower refer to each transect with sites at 0, 35, 100, 200, 500 and 1000 m from the cage. 'CI' are inner control and 'CO' the outer control outer sites.

Southern Channel

The Lippies lease was surveyed in Autumn 2017, Spring 2017 and Autumn 2018. The intention was to spread these over different stages on the farming cycle (e.g. at peak production and after fallowing) to see how the footprint changed and if recovery could be detected. We sampled every ~ 6 months to try to capture high and low points in production for the lease (Figure 5.1-9). As a result, the three surveys corresponded to mid-production (Autumn 2017; 7 months after the fish were first introduced), peak production (Spring 2017; 13 months after the fish were first introduced) and fallow (Autumn 2018; 2-3 months after most fish were harvested).

At the Lippies lease, the three transects (upper, middle, and lower) extended perpendicular outwards to the predominant direction of the lease (Figure 5.1-10). The transects were at the southern end of the lease where the farm was first stocked. Each transect consisted of six sampling sites at various distances from the cage, including one site directly adjacent to the cages (0 m), 35 m, 100 m, 200 m, 500 m, and 1000 m. During survey one the 1000 m site 1-ELM-6 was found to be positioned over patchy reef and gravel and was therefore replaced in future surveys by 1-ELM-6a, positioned 850 m from the cage and offshore away from the reef. Additionally, four inner control sites (approximately 100-300 m from the cages) and four outer control sites (approximately 900-1200 m from the cages) were sampled to provide a measure of natural variability (Figure 5.1-10). Sediment nutrient fluxes were only measured in surveys 2 and 3, and sampling was restricted to the upper and middle transect sites (see further details below).

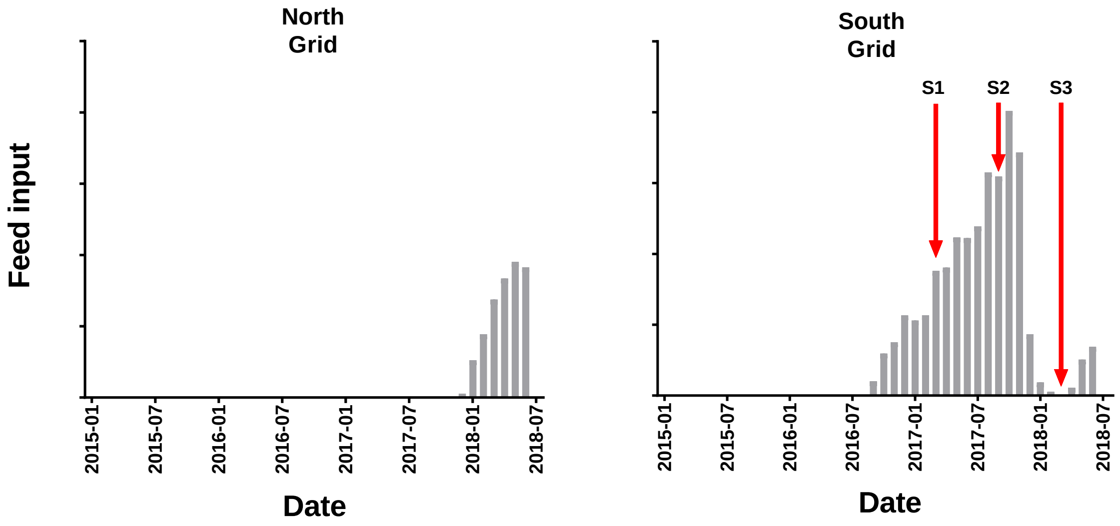

Figure 5.1-9 Monthly feed input into the north and south grid of the Lippies lease over the years 2015-2018, showing when surveys one to three (S1-S3) were completed in the current IMAS study. Note, all of the transects were on the south grid because farming commenced earlier in the study. To overcome the constraints of commercial in confidence considerations, feed information is provided in relative terms.

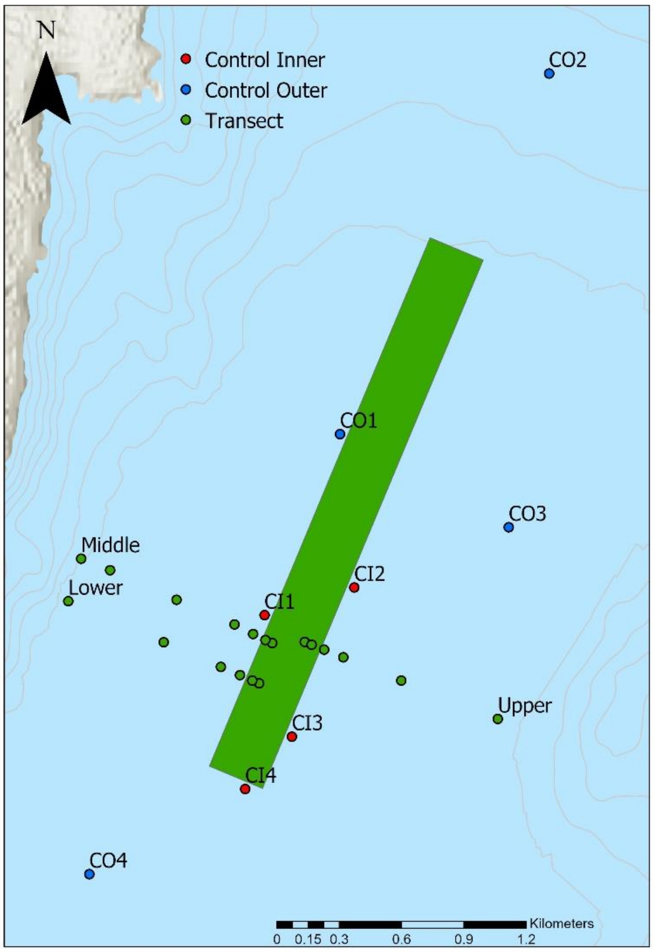

Figure 5.1-10 Map of the Lippies lease showing the sample locations.

Macquarie Harbour

The Table Head Central (THC), Gordon (GR), Strahan (SH) and Franklin (FR) leases were surveyed six times between January 2015 and October 2016 as part of FRDC 2015-024 but were continued under FRDC project 2016-067. Because the results of all the Macquarie Harbour surveys have been synthesised in reporting for FRDC 2016-67 (see Ross et al. 2021), the results section in this report provides a detailed assessment of the of the first 6 surveys in a format consistent with that used for the other regions. While the original intention was to spread these over different stages on the farming cycle (e.g. at peak production and after fallowing) in the same manner as the southern region locations to see how the footprint changed with production load, and if recovery could be detected, this was in fact difficult to achieve in Macquarie Harbour as the production cycles were so different both between and within the leases. Consequently, we sampled every ~ 3-4 months at this location to try to capture the high and low points in production for the lease (Figure 5.1-11).

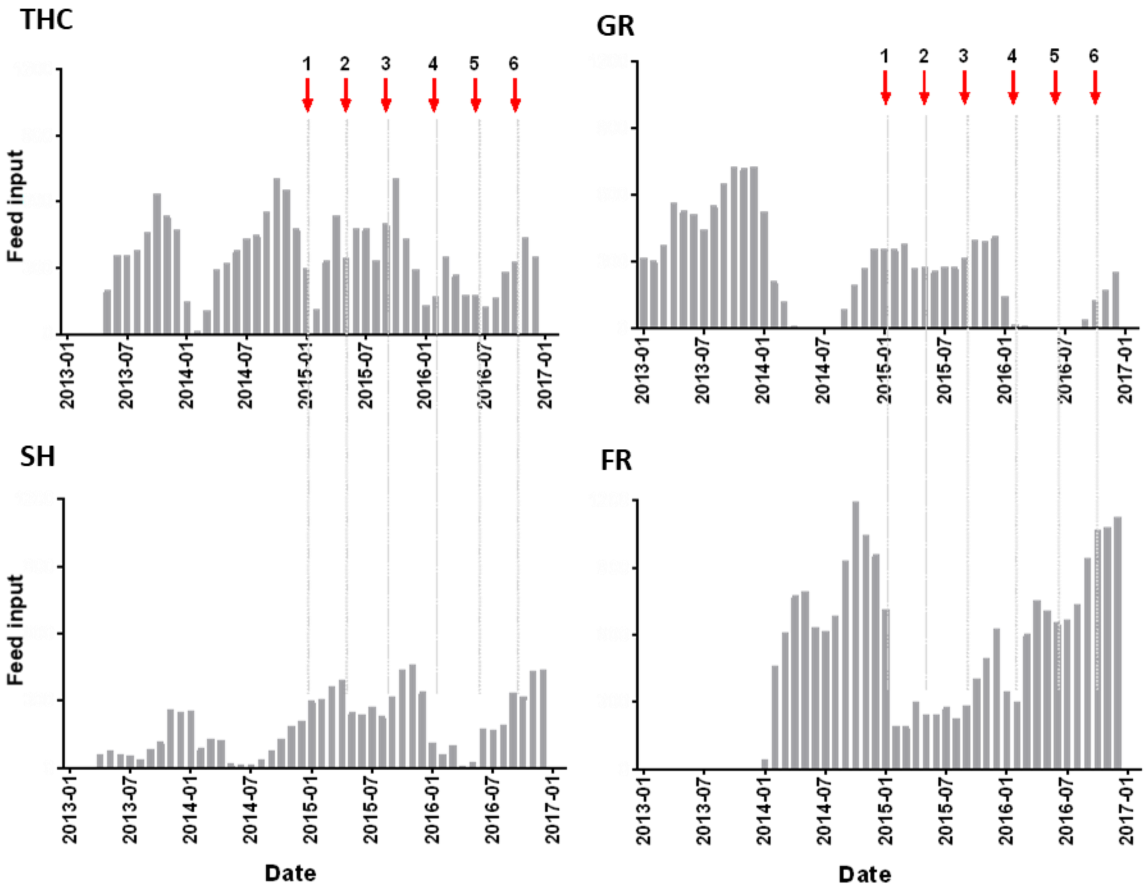

Figure 5.1-11 Monthly feed input at the Macquarie Harbour study lease from 2013-2016, showing when surveys 1 - 6 were completed in the current IMAS study. To overcome the constraints of commercial in confidence considerations, feed information is provided in relative terms.

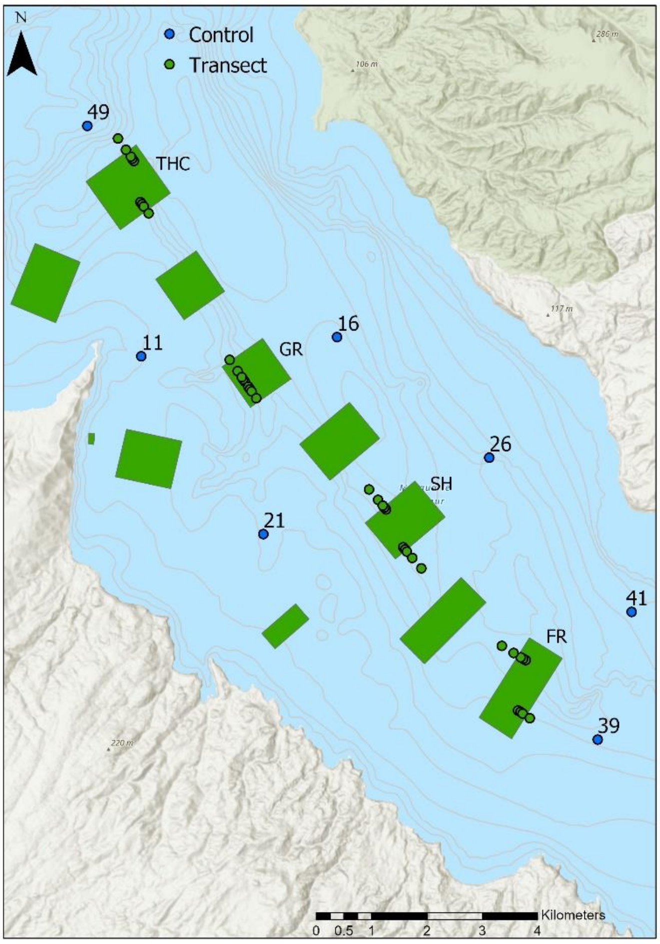

Figure 5.1-12 Map showing the Macquarie Harbour control (blue) and lease transect (blue) sites at Table Head Central (THC, MF133), Gordon (GR, MF219), Strahan (SH, MF267) and Franklin (FR, MF266). There are two transects at each of the study leases with five sites at 0, 50, 100, 250 and 500 m from the cage on each transect.

5.1.2.3 Sample Collection

Physico-chemical

At each site samples were collected using a quad-corer consisting of Perspex tubes (250 mm long, 45 mm internal diameter) to evaluate sediment sulphide, redox, particle size, organic carbon and nitrogen content and their isotopic composition ($\delta^{15}N$, $\delta^{13}C$) (see Appendix Y for a data overview). For the parameters used in the current licence conditions, the methods of collection and analysis of data were in accordance with those outlined in Schedule 3 of the Marine Farm Licence Conditions.

In the laboratory, redox was measured at 3cm depth using a Hach HQ30d oxidation-reduction potential (ORP or redox potential) probe, calibrated with ZoBell’s solution prior to analysis using the method described in Macleod et al. (2004). Redox potential was recorded once the probe had stabilised (i.e. when the meter displayed constant values for approximately 10 seconds). The probe was re-calibrated after every three measurements.

Sulphides in sediments were measured using a TPS WP-90 meter. Sub-samples of sediment (2 mL) were extracted from a port in the side of each core tube 3cm below the sediment surface using a 5 mL syringe. The samples were then placed in a glass vial containing 2 mL SAOB (refer to Macleod et al. 2004) and sulphide concentration was measured (mV) by placing the probe into the vial, and slowly stirring the sediment / buffer mix until the reading stabilised. The mV readings were converted to sulphide concentration using a calibration curve as outlined in Macleod and Forbes (2004).

Samples for carbon and nitrogen content and isotopic composition were ground and the sample for carbon analysis was acidified with a dilute HCl solution to dissolve solid carbonates. The Water Studies Centre (Monash University) analysed the samples on an ANCA GSL2 elemental analyser interfaced to a Hydra 20-22 continuous-flow isotope ratio mass-spectrometer (Sercon Ltd., UK). The precision of the elemental analysis was 0.5 μg for both C and N (n = 5). The precision of the stable isotope analysis was ±0.1‰ for ^{13}C and ±0.2‰ for ^{15}N (SD for n=5). Stable isotope data are expressed in the delta notation (δ^{13}C and δ^{15}N), relative to the stable isotopic ratio of Vienna Pee Dee Belemnite standard (RVPDB= 0.0111797) for C and atmospheric N2 (RAir = 0.0036765) for nitrogen.

Because the current licence schedule specifies that percent organic carbon is measured using the loss on ignition (LOI) technique rather than via the elemental analyser as described, samples were also analysed using the LOI technique for comparison. The top 2 cm of core was oven dried at 60 °C for 24hrs and the total organic carbon calculated based on the loss on ignition at 500 °C after 4 hours.

A profile of the physio-chemical properties of the overlying water column (dissolved oxygen, salinity, pH and temperature) was obtained at each sampling location using an YSI EXO2 Sonde, with measurements recorded every 5 m.

The top 2cm of sediment collected in cores was used to determine percentage organic carbon. In the laboratory, percent organic carbon was calculated after drying (60 °C for 24hrs) by loss of organic carbon on ignition at 500 °C.

Macrofauna

Benthic macrofauna was sampled in triplicate at each site using a Van Veen Grab (surface area 0.0675 m^2). All grab samples were wet sieved to 1mm and preserved in 10% formalin: seawater (4% formaldehyde) in the field. In the laboratory, samples were washed and stored in ethanol. After being sorted, the infauna was identified to the lowest possible taxonomic level and counted.

Visual

Underwater video surveys of the study sites were conducted in parallel[3] with the benthic sampling. Three minutes of footage were collected at each site and the footage assessed following the methods described by Macleod and Forbes (2004) and outlined in Schedule 3V. The videoing was carried out by the respective companies and subsequently analysed by IMAS staff. The original table of features used for scoring ROV in Macleod and Forbes (2004) was expanded to include additional variables that were not previously scored but were indicative of impacted or unimpacted conditions (Table 5.1-2) for all farming areas around Tasmania. The variables measured included numeric categorisation of sediment colour,

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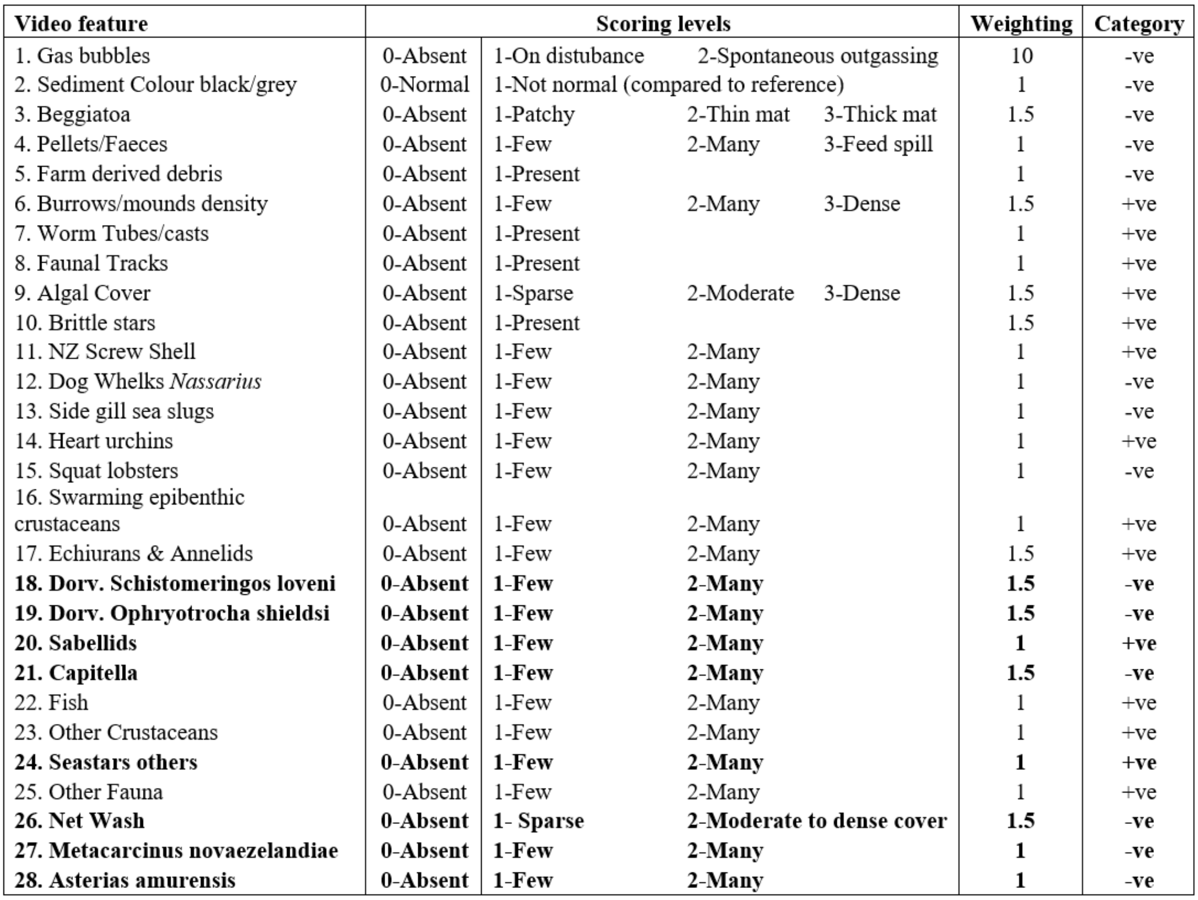
[3] The ROV surveys were not possible for all surveys and in some cases the timing was not fully aligned with the benthic sampling

Beggiatoa density, presence of gas bubbles, feed pellets and faeces, farm debris, prevalence of burrows, faunal tracks, worm tubes, and the abundance of key fauna (e.g. molluscs, ophiuroids, annelids, capitellids, dorvilleids, NZ screw shells, seastars). These measures were then used to score the footage for each site according to the key features that were determined to be indicative of impacted or unimpacted conditions (Table 5.1-1; Macleod and Forbes 2004). Features that indicated a detrimental affect (e.g. gas bubbles, pest species, opportunistic fauna, pellets and faeces) were subtracted from the score, while features that were deemed to have little or no impact (e.g. worm tubes, brittle stars) were given a positive score. Therefore, the higher the score, the better the sediment condition. This method provided a way to assess the health (or impact level) of the sediment from video footage.

Table 5.1-2 Key features and scoring levels for video assessment. Features highlighted in bold are those that have been added to the original Macleod and Forbes (2004) study.



Video feature	Scoring levels				Weighting	Category
1. Gas bubbles	0-Absent	1-On distubance	2-Spontaneous outgassing		10	-ve
2. Sediment Colour black/grey	0-Normal	1-Not normal (compared to reference)			1	-ve
3. Beggiatoa	0-Absent	1-Patchy	2-Thin mat	3-Thick mat	1.5	-ve
4. Pellets/Faeces	0-Absent	1-Few	2-Many	3-Feed spill	1	-ve
5. Farm derived debris	0-Absent	1-Present			1	-ve
6. Burrows/mounds density	0-Absent	1-Few	2-Many	3-Dense	1.5	+ve
7. Worm Tubes/casts	0-Absent	1-Present			1	+ve
8. Faunal Tracks	0-Absent	1-Present			1	+ve
9. Algal Cover	0-Absent	1-Sparse	2-Moderate	3-Dense	1.5	+ve
10. Brittle stars	0-Absent	1-Present			1.5	+ve
11. NZ Screw Shell	0-Absent	1-Few	2-Many		1	+ve
12. Dog Whelks *Nassarius*	0-Absent	1-Few	2-Many		1	-ve
13. Side gill sea slugs	0-Absent	1-Few	2-Many		1	-ve
14. Heart urchins	0-Absent	1-Few	2-Many		1	+ve
15. Squat lobsters	0-Absent	1-Few	2-Many		1	-ve
16. Swarming epibenthic crustaceans	0-Absent	1-Few	2-Many		1	+ve
17. Echiurans & Annelids	0-Absent	1-Few	2-Many		1.5	+ve
18. Dorv. Schistomeringos loveni	**0-Absent**	**1-Few**	**2-Many**		**1.5**	**-ve**
19. Dorv. Ophryotrocha shieldsi	**0-Absent**	**1-Few**	**2-Many**		**1.5**	**-ve**
20. Sabellids	**0-Absent**	**1-Few**	**2-Many**		**1**	**+ve**
21. Capitella	**0-Absent**	**1-Few**	**2-Many**		**1.5**	**-ve**
22. Fish	0-Absent	1-Few	2-Many		1	+ve
23. Other Crustaceans	0-Absent	1-Few	2-Many		1	+ve
24. Seastars others	**0-Absent**	**1-Few**	**2-Many**		**1**	**+ve**
25. Other Fauna	0-Absent	1-Few	2-Many		1	+ve
26. Net Wash	**0-Absent**	**1- Sparse**	**2-Moderate to dense cover**		**1.5**	**-ve**
27. Metacarcinus novaezelandiae	**0-Absent**	**1-Few**	**2-Many**		**1**	**-ve**
28. Asterias amurensis	**0-Absent**	**1-Few**	**2-Many**		**1**	**-ve**

Sediment processes

Samples for measuring sediment nutrient fluxes were collected using a HAPS bottom corer which takes well defined, undisturbed cores from soft sediments (Kanneworff & Nicholaisen 1973). Sample tubes (cores) were made of polycarbonate (inner diameter 127 mm; length 300 mm). All cores were carefully inspected and only those with sediment collected to a depth of 80-120 mm with an undisturbed sediment-water interface were used. All cores were capped and transferred to a bin filled with bottom water for transport back to the laboratory. To ensure that incubations conditions remained like bottom water conditions in the field, a bilge pump was used to collect 100 litres of bottom water (~ 1m above the sediments). Water column profiles of salinity, temperature and dissolved oxygen were also taken at each site using a YSI 6600 V2 Multi Parameter Water Quality Sonde with YSI 650 MDS logger.

The cores were transferred to temperature controlled baths and allowed to equilibrate overnight in site water at in situ temperature and oxygen concentrations. All cores were stirred continuously throughout the equilibration period and during the incubation via a battery-operated stirrer mounted in the core lid with the stirring rate set to ensure mixing of the water column but without agitating the sediment surface. During the equilibration period the core lids were raised approximately 10 mm above the cores such that they were not sealed and water could mix freely with tank water; this was to minimise oxygen depletion and the accumulation of analytes in the cores during the equilibration period. Note, all incubations were undertaken in the dark given that little light is likely to reach the sediment at the survey sites due to the depth (>25 m).

To start the incubation, the cores were sealed and flushed with site water (gravity fed via sample ports in the core lid) for approximately 20 minutes. Samples for measuring benthic respiration and nutrient flux (oxygen (O_2), dissolved inorganic carbon (DIC), ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-) and phosphate (PO_4^{3-}) were collected at 3-time intervals over the course of the incubation: 30 minutes after flushing was finished, mid incubation and at the end of the incubation period. Samples for analysis of NH_4^+, NO_2^-, NO_3^- and PO_4^{3-} were filtered (0.45 µm; 30 mm polypropylene housing; Bonnet) and stored in 12-mL high density polyethylene sample tubes. Nutrient samples were frozen until analysis. Samples for DIC were filtered (0.45 µm; 30 mm polypropylene housing; Bonnet) and preserved in a 12-mL Exetainer with 20 µl $HgCl_2$ and refrigerated until analysis. Simultaneously, DO concentration was measured using HACH LDO101 (HACH, Colorado USA) optical DO probe. Dissolved oxygen was measured at the same time intervals using a Hach HQ40d with DO probes. The length of incubation is determined by the rate of oxygen depletion allowing for a total drop of no more than 10-20% in oxygen saturation. This equated to incubations running for between 3 and 24 hrs. The fluxes of each analyte were calculated based on the change in concentration over time, while taking into consideration the water volume inside the cores and surface area of the sediment (Dalsgaard et al. 2000). Values were corrected for any replacement of water.

All nutrient samples (NH_4^+, NO_2^-, NO_3^- and PO_4^{3-}) were analysed at IMAS using flow injection analysis (FIA) (Lachat Quikchem 8000 Flow injection Analyser, spectrophotometric detector). All nutrient analyses followed the procedures in Standard Methods for Water and Wastewater (APHA 2005). DIC was analysed based on the Coulorometric method using a LI-7000 CO_2/H_2O gas analyzer (LI-COR Biosciences, Lincoln, NE, USA).

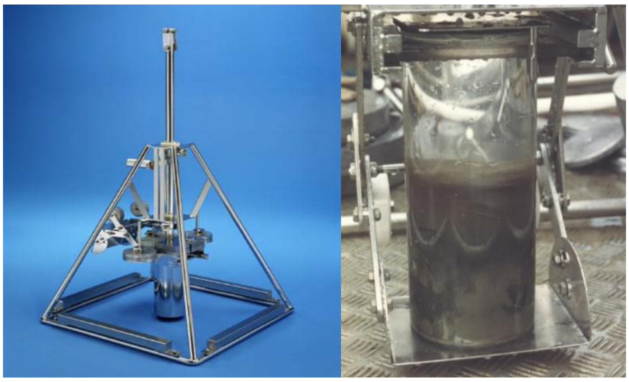


Figure 5.1-13 Haps bottom corer (left) and an undisturbed sediment core collected using a polycarbonate sample tube (right).

5.1.2.4 Data analysis

To establish the magnitude and scale of the benthic response to farming, the effect of distance on response parameters (i.e. sediment, physio-chemical and macrofauna) was investigated for each survey. To visualise the effects of distance from the source of enrichment, each parameter was graphed against distance along the transect using bar charts and heat maps produced in R (R Core Team 2014). Plots of response variables by distance from the cage with a smoother fit using loess (± SE) are also provided in Appendix A. The inner and outer control sites were included (where available) to assess changes along the transects due to farming versus natural variability observed in the area.

All response parameters were tested for the effect of distance from cages along the transect with a generalised linear model (GLM) in R (R Development Core Team 2014). Graphical analysis of residuals showed that linearity and variance homogeneity were improved by log transformation of distance from the cage. For comparison between control sites and different transect distances a factorial ANOVA was performed, and post hoc tests were conducted using the 'multcomp' package.

Lease performance

To determine whether the current set of indicators and thresholds (see BOX 1) are 'fit for purpose' in these different growing regions, the environmental performance at 35/50 m from cage and 35 m from the lease boundary were assessed via two planned comparisons conducted following the ANOVA:

1. 35 m from cage versus outer control (CO)
2. 35 m from lease versus outer control (CO).

Environmental performance at farmed leases was assessed at 35 m outside the lease boundary (see introduction). At most farm sites, cages were well within the lease boundary, and as such the distance between the source of enrichment (the cage) and the 35 m compliance point was variable. For example, at the Lippies lease, cages were approximately 200 m from the compliance point. Therefore, our planned comparisons were conducted to assess two distinct, but related questions on lease performance. Comparison 1 assessed how the impact footprint from the cage compared to the original studies, and comparison 2 assessed how the lease was performing at the actual compliance point (i.e. CI sites). The second comparison was further separated into two, with the first comparing the CO with the distance on the transect that correspond to the 35 m from lease (e.g. 200 m at Lippies) and the second comparing CO with CI. The CI sites were all 35 m from lease compliance points, but unlike the transect distance that was 35 m from the lease, they were spread around the lease boundaries and thus, represented variability in the area. Note, because the purpose of this study was to explore whether there could be other, more suitable indicators and thresholds, these tests were not limited to the indicators listed in Figure 5.1-2.

Temporal response

To determine whether the benthic response changed in time (i.e. across surveys), the response parameters were compared across surveys with a GLM at the two distances closest to the cages (0 and 35/50 m) and at the three distances furthest from the cages (500, 1000 and CO). It is hypothesized that changes observed at 0 m and 35/50 m sites are more likely attributed to farming, whereas changes observed at 500 m or greater are likely to reflect variation due to other external drivers.

Multivariate analysis

Multivariate analysis was used to investigate the changes in the macrofauna community with distance from the cage using recent developments in model-based approaches to explanatory models and ordination (see review by Warton et al. 2015). These methods specify a statistical model for abundances jointly across many taxa, accounting for correlations between species to simultaneously explore interactions across taxa and the response of abundance to environmental variables.

A multivariate GLM was used to test the relationship between distance and species composition with the ‘manyglm’ function on the ‘mvabund’ package (Wang et al. 2012) in R. Species with fewer than 10 occurrences in total were removed, as these contained little information but could strongly influence the model after rescaling. A negative binominal distribution with a log link was used, and model assumptions were evaluated by examining plots of Dunn-Smyth residuals (Wang et al. 2012). Goodness of fit was evaluated using a pseudo-R^2, calculated as the difference in log-likelihood between the full and intercept-only multivariate models. Variable significance was estimated using the Wald Statistic, accounting for correlation between variables using the identify option. Species specific univariate results were calculated using an adjusted step-down procedure. For species where a significant distance effect was found, a fourth corner model with a LASSO penalty was fitted and the resulting coefficient matrix was plotted as a heat map.

Further explanation of the multivariate data was done using joint statistical modelling, which incorporated into a single model the impact of abundance on environmental predictors and interspecific interactions (Warton et al. 2015). We used a latent variable model (LVM) with a negative binomial family and log link (Niku 2017) to create an unconstrained ordination to visualise the main trends between transect distances in terms of their species composition (Hui et al. 2015). The role of the latent variables in LVM is to account for unknown or unmeasured variability, and by inducing correlations between taxa, enable an unconstrained ordination for visualizing distance and species patterns. The shared environmental response between species to distance from the cages was evaluated using the significant correlations between columns of the response matrix in the LVM and presented as an environmental correlation plot (Warton et al. 2015). Likewise, the residual correlation and precision matrices were used to estimate correlations between species not accounted for by their shared environmental response (Pollock et al. 2014). The role of the latent variables in LVM is to account for unknown or unmeasured variables, and by inducing correlations between taxa, enables an unconstrained ordination for visualizing distance and species patterns. LVMs were estimated using Bayesian Markov chain Monte Carlo methods via the ‘boral’ package (Hui 2016) in R.

Comparison with previous conditions

To assess whether the biological community and the physical and chemical characteristics changed through time we compared our data with that recorded in previous surveys. This required a careful selection of sites to ensure data were from the same (or similar) locations. Where available we included farm, compliance, and control sites. This was not done for

Macquarie Harbour given the limited overlap in sites for comparison, but see Ross et al. (2016) and (2021) for a comparison with previous conditions

Southern Channel

Figure 5.1-14 shows the original lease and the sites surveyed in the 2004 baseline, 2016 baseline (i.e. following the change in position and size of the lease in 2004) and by IMAS from 2017-18. For the comparison of compliance sites, the following sites were used across the three data sets: 2004 (S2 & S3), 2016 (S3, S4, S5 & S6), and for 2017-18, 100 m (U100, M100, L100) and CI (CI1, CI2 CI3 CI4) sites. For the comparison at control sites, there was only the one site with overlap across all three data sets: 2004 (S6), 2016 (S9), and 2017-18 (CO4). Farm sites were only sampled and available in 2017-18 0 m (U0, M0, L0) sites.

Storm Bay

Trumpeter lease. Figure 5.1-15 shows the original lease and the sites surveyed in the 2014 baseline of the northern end and the 2016 baseline of the southern end and by IMAS from 2016-18. For the comparison of compliance sites, the following sites were used across the three data sets: 2014 (S1,S2,S3), 2016 (NA-no overlap), and for 2016-18 (CI1,CI2,CI3). For the comparison at control sites, there was one site (referred to hereafter as control site x) with overlap from the 2014 baseline (C1) and the IMAS surveys (CO2) and one site (referred to hereafter as control site x) from the 2016 baseline (S12) with the IMAS surveys (CO1). Farm sites were only sampled and available in the IMAS 2016-18 surveys 0 m (U0, M0, L0) sites.

Storm Bay 1 lease. Figure 5.1-16 shows the lease and the sites surveyed in the 2015 baseline and by IMAS from 2016-18. For the comparison of compliance sites, the following sites were used across the two data sets: 2015 (S5,S6,S7,S8,S9 & S10) and for 2016-18, 100 m (U100, M100, L100) and CI (CI4, CI5 CI6) sites. For the comparison at control sites, there were two sites with overlap from the 2015 baseline (C2 & C6) and the IMAS surveys (CO3 & CO4).

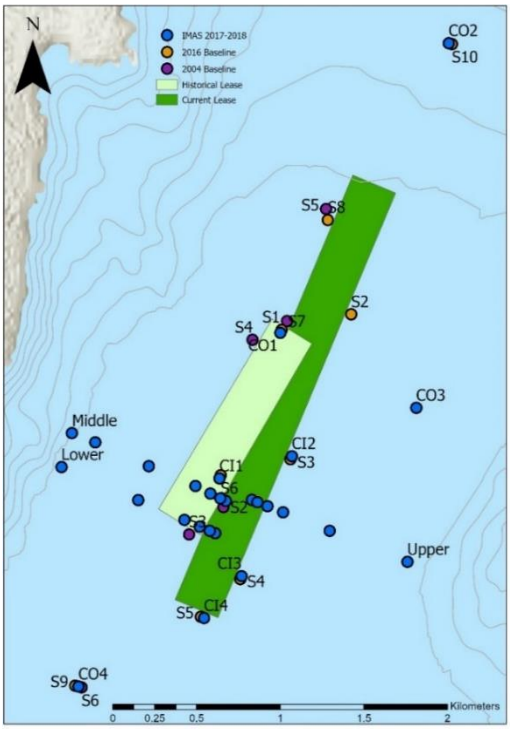


Figure 5.1-14. Map showing historical lease for Lippies and sites for baseline surveys conducted in 2004 (purple), 2016 (orange) and the current IMAS survey over 2017-2018 (blue).

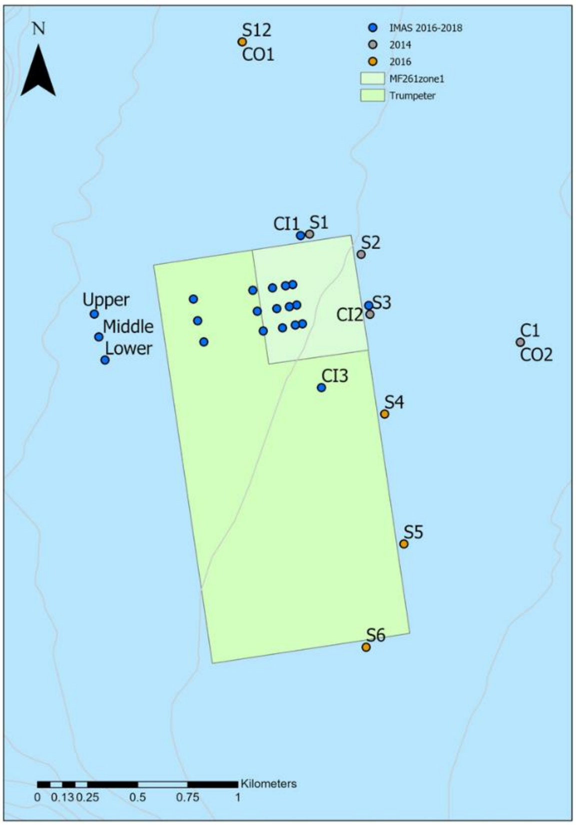


Figure 5.1-15 Map showing the Trumpeter and sites for the baseline surveys conducted in the 2014 (grey) of the northern end and the 2016 (orange) of the southern end and the current IMAS survey over 2017-2018 (blue).

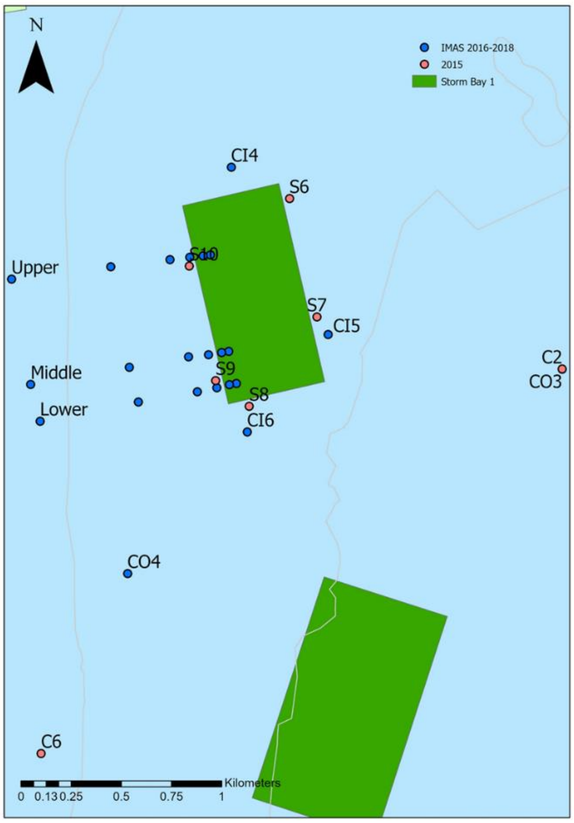


Figure 5.1-16 Map showing the Storm Bay 1 and sites for the baseline surveys conducted in the 2015 (orange) and the current IMAS survey over 2017-2018 (blue).

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5.2 Southern Channel (Lippies Lease)

5.2.1 Results

5.2.1.1 Physico-chemical

Redox

Redox potential for all three surveys was oxic and >100 mV at all sites (Figure 5.2-1). Redox potential generally increased with distance from the cages and the effect of distance was significant in all three surveys (Figure 5.2-1; Table 5.2-1). Redox potential 35 m from the cage sites was lower than at the outer control sites in all three surveys, but this difference was only significant in survey 3 (Table 5.2-1). Redox potential at the 200 m transect sites (35 m from the lease boundary) was similar to the outer control sites in surveys 1 and 2. In survey 3 redox potential was significantly higher at the outer control sites compared to the 200 m distance. Redox potential was on average higher across all sites in survey 3 compared to surveys 1 and 2, however the effect of surveys was only significant at the sites closest to the cages (0 and 35 m) and at the 500 m site, not at the two most distant sites (1000 m and outer control (CO); Table 5.2-2).

Sulphides

Sulphide concentrations (µM) decreased with distance from the cages (Figure 5.2-2) and the effect of distance was significant across all three surveys (Table 5.2-1). Although sulphide concentrations were on average higher at the 35 m sites compared to the outer control sites in each survey, the difference was not significant. Sulphide concentrations at the 200 m transect sites (35 m for lease boundary) and outer control sites were very low in all three surveys and not significantly different (Table 5.2-1). There was a significant effect of survey at the sites closest to the cages (0 and 35 m) but not at the more distant sites (500 m, 1000 m and CO; Figure 5.2-2, Table 5.2-2). This was most notable in survey 2 where sulphide concentrations were elevated at the 0 and 35m sites compared to surveys 1 and 3 (Figure 5.2-1).

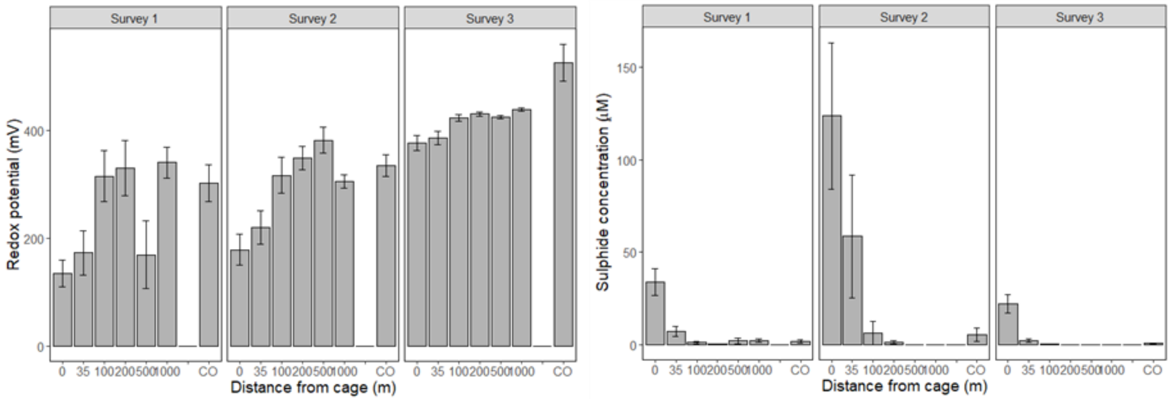

Figure 5.2-1 Average (±SE) redox potential (mV) and sulphide concentration (µM) at 3cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Lippies lease. Redox and sulphide were not measured at the inner control sites.

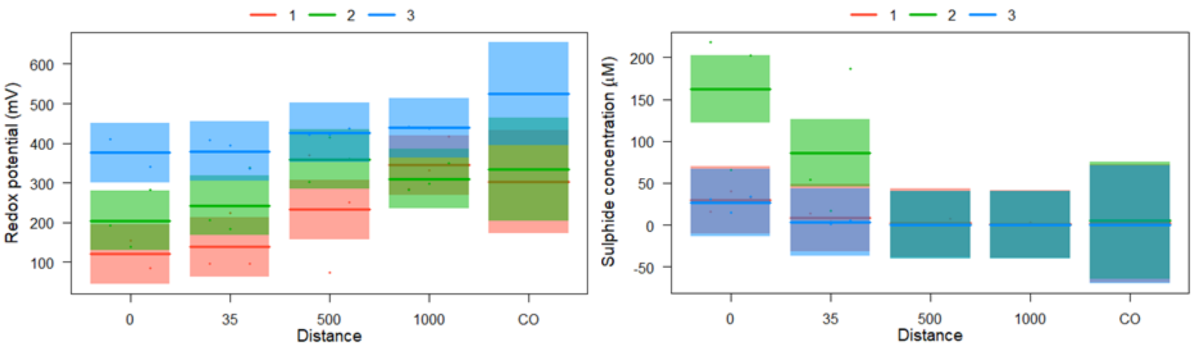

Figure 5.2-2 Redox and sulphide (predicted mean ± 95% confidence intervals) from the GLM comparing surveys 1, 2, and 3 at the two distances closest to the cages (0 and 35m) and at the distances furthest from the cages (500, 1000m and CO). It is hypothesized that changes at the closer sites are more likely attributed to farming, whereas changes observed at 500m or greater likely to reflect variation due to other, external drivers.

Sediment particle size

The sediment particle size distributions appeared broadly similar along the transect except for the 1000 m distance sites, which contained more coarse sand (i.e. greater percentages of 2 mm, 1 mm, 0.5 mm and 0.25 mm size fractions) than all other transect sites (Figure 5.2-3).

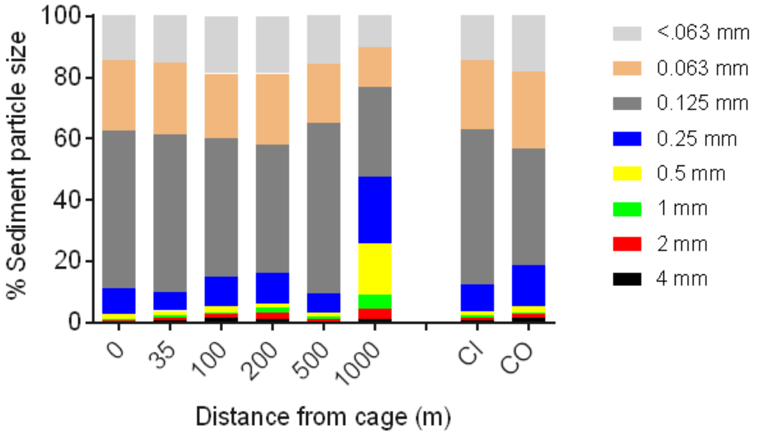

Figure 5.2-3 The cumulative percentage of standard sediment particle sizes at each distance from the cages (on transects) and at inner (CI) and outer controls (CO).

Organic carbon and nitrogen and stable isotopes

Organic carbon content (%)

The organic carbon content (%)[1] of the sediments was highly variable across the transect and control sites (Figure 5.2-5). In surveys two and three the highest values were recorded closest to the cage (0m), but the effect of distance was only significant in survey 3 (Table 5.2-1). There was no significant difference in organic carbon content between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Figure 5.2-5; Table 5.2-1). There was a significant increase in organic carbon content across surveys at the cage site (0 m), with survey 1 having a lower level than the other two surveys (Figure 5.2-6; Table 5.2-2). There also appeared to be a general increase in

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[1] Organic carbon in survey 1 was also measured using the loss on ignition technique (LOI) with values ranging for 2-6% across sites compared to 0.15-1.1% as measured with the elemental analyser on the same survey,

organic carbon content across other sites between survey 1 and 2, but the effect of survey was not significant at those distances (i.e. 35 m, 500, 100 m and CO).

Organic nitrogen content (%)

In all three surveys the organic nitrogen content (%) of the sediment was higher at cage (0 m) sites compared to the transect and control sites (Figure 5.2-4); the effect of distance was significant in all three surveys (Table 5.2-1). There was no significant difference in organic nitrogen content between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in any of the surveys (Figure 5.2-4; Table 5.2-1). Although the organic nitrogen content at 0m increased between survey 1 and 2, the effect of survey was not significant at 0 m or the other distances tested (Figure 5.2-5; Table 5.2-2).

C:N molar ratio

The carbon to nitrogen molar ratio (C:N) of the sediment was more depleted (typical of feed inputs) closest to the cages at 0 m sites, particularly in surveys 1 and 2 (Figure 5.2-4); however, the effect of distance was only significant in survey 2. There was no significant difference in C:N ratio between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Figure 5.2-4; Table 5.2-1). Although the C:N ratio at 0 m increased between survey 1 and 2, the effect of survey was not significant at 0 m or the other distances tested (Figure 5.2-5; Table 5.2-2).

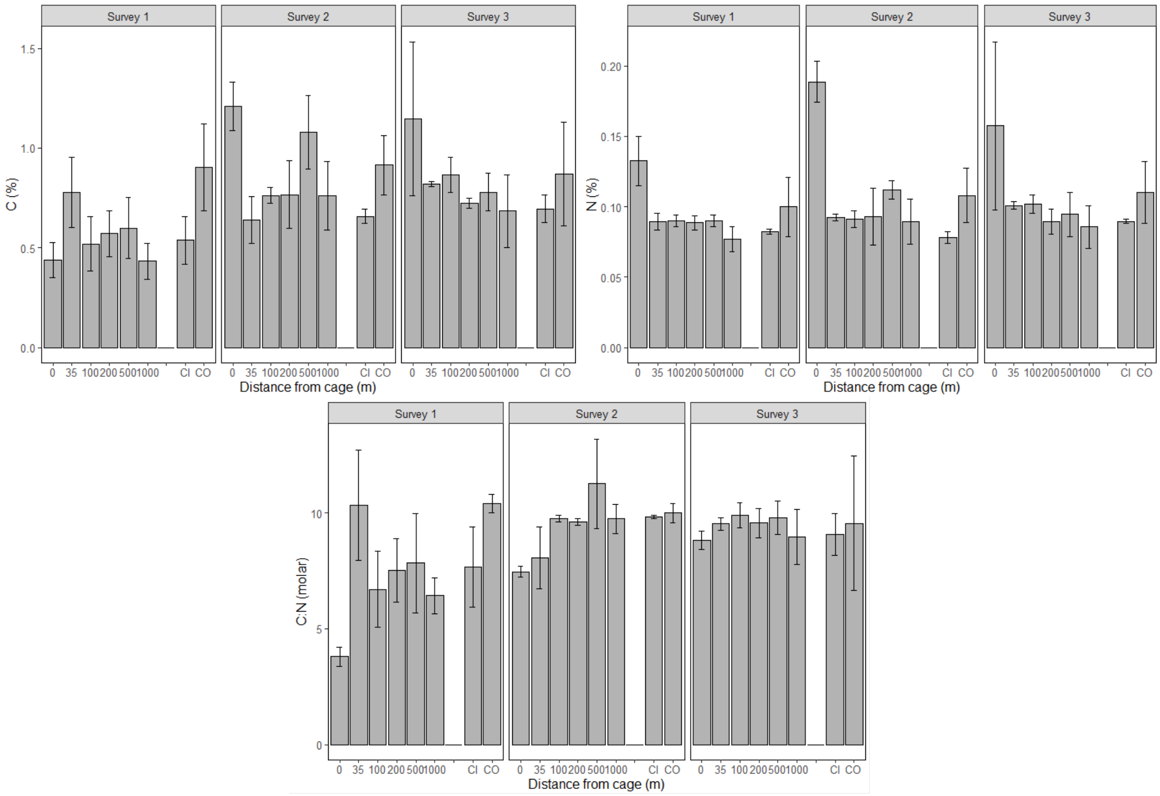

Figure 5.2-4 Average (±SE) of percent carbon (%C) and nitrogen (%N) and C:N molar ratio with distances from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Lippies lease.

Carbon (δ^{13}C) and Nitrogen (δ^{15}N)

The carbon and nitrogen isotopes of the sediments at the Lippies lease showed values typical of the Channel environment, averaging -22.50 (±1.20 ‰) for δ^{13}C and 8.28 (±0.66 ‰) for δ^{15}N across all three surveys (Figure 5.2-6; see BEMP Annual Report 2017/18), but with the 0 m sites clearly distinguished from the remaining transect sites and controls on the dual isotope plot (Figure 5.2-7). This pattern reflected the slightly depleted carbon isotope signatures at the 0 m sites in survey 2 and 3 where the effect of distance was significant, and the depleted nitrogen isotope signature at the 0 m sites in all three surveys (Table 5.2-1). Whilst there was no significant difference in the carbon or nitrogen isotope signature between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Figure 5.2-4; Table 5.2-1), the influence of farm derived organic matter is evident at the 35 m from cage site in dual isotope plot for surveys 2 and 3 (Figure 5.2-7). The effect of survey was significant at the 0 m site for both carbon and nitrogen isotopes but not at the 35 m and more distant sites (500 m, 1000 m and CO; Figure 5.2-8, Table 5.2-2).

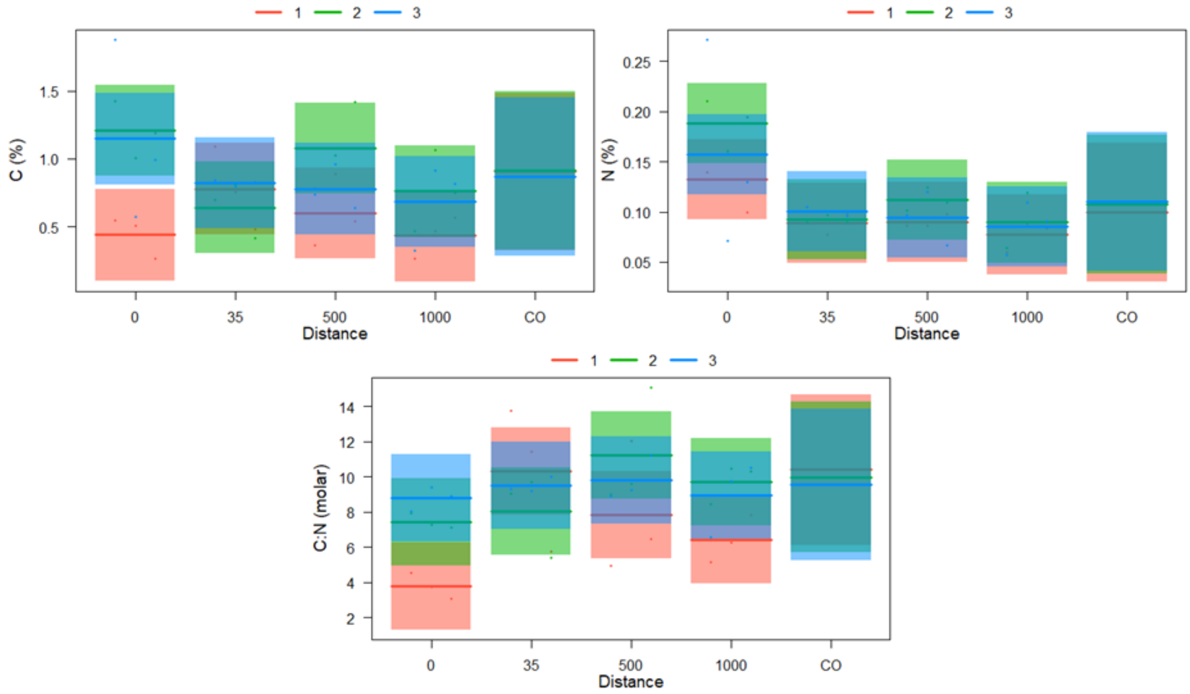

Figure 5.2-5 Percent carbon (%C) and nitrogen (%N) and C:N molar ratio (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO).

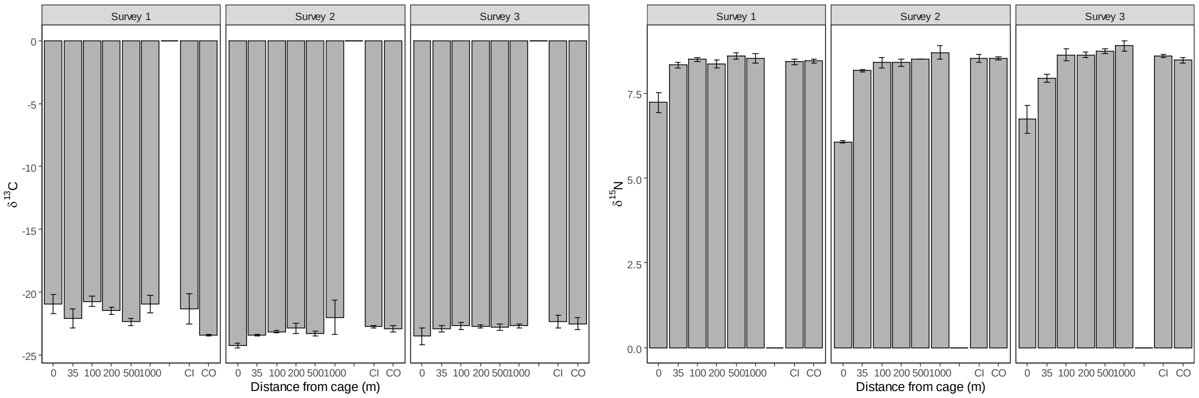

Figure 5.2-6 Carbon (δ^{13}C) and nitrogen (δ^{15}N) stable isotopes values (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Lippies lease.

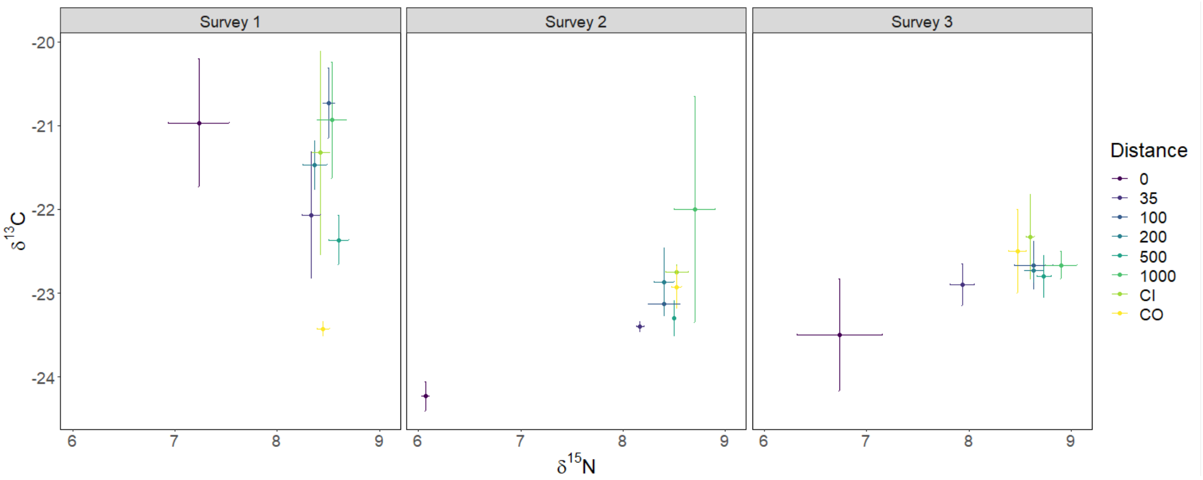

Figure 5.2-7 Distribution of sites/distances based on carbon and nitrogen stable isotopes with error bars for each survey at the Lippies lease.

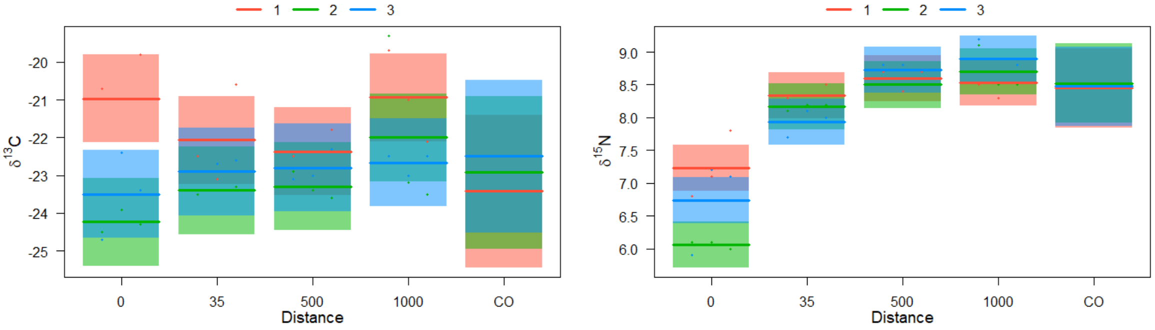

Figure 5.2-8 Carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) stable isotopes values (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO).

Dissolved oxygen

There was some evidence of slightly depleted bottom (measurement taken at 1 m above the substrate) water oxygen levels at the sites closest to the cages, but the effect of distance was only significant in survey 2 (Figure 5.2-9, Table 5.2-1). There was no significant difference in bottom water DO between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in any of the surveys (Table 5.2-1). There was a significant change in bottom water DO between surveys as shown by the comparison of sites both close and more distant from the cages (Table 5.2-2). Bottom water DO ranged from ~6.5 mg/L (80% saturation) in survey 1 (Autumn 2017) to ~ 8.6 mg/L (99% saturation) in survey 2 (Spring 2018), and ~ 7.4 mg/L (93% saturation) in survey 3 (Autumn 2018) (Figure 5.2-10).

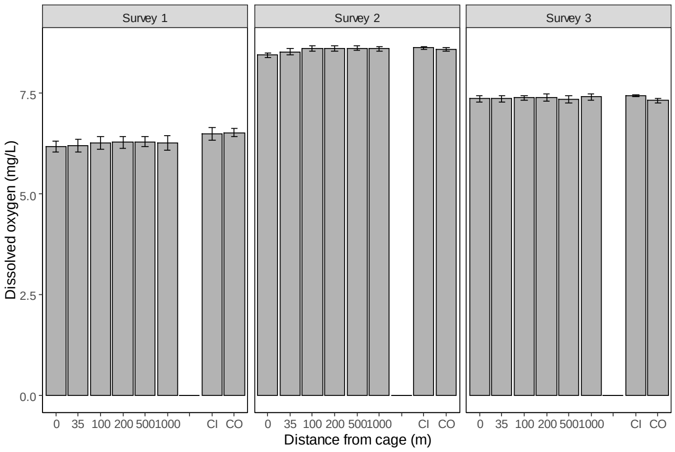

Figure 5.2-9 Average dissolved oxygen (DO) concentration (mg/L) in bottom water with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Lippies lease.

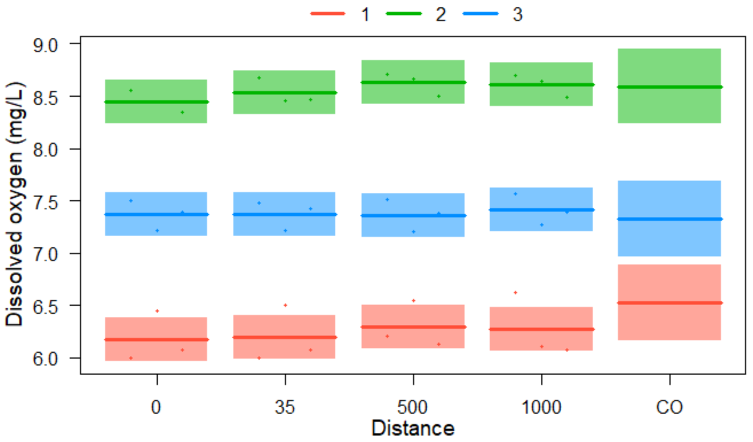

Figure 5.2-10 Average dissolved oxygen (DO) concentration (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO).

Table 5.2-1 Results of the generalised linear model testing for the effect of distance for all physico-chemical parameters for each survey and the planned contrasts from the factorial ANOVA comparing the control sites and transect distances. Significant responses ($p < 0.05$) are shown in bold.

			Planned contrasts		
	F ratio	**P**	**CO v 35 m**	**CO v 200 m**	**CO v CI**
Redox					
Survey 1	9.734	**0.007**	0.19455	0.78460	
Survey 2	9.114	**0.008**	0.485	0.999	
Survey 3	16.778	**0.001**	**<0.001**	**<0.001**	
Sulphide					
Survey 1	42.224	**<0.001**	0.533	0.973	
Survey 2	26.22	**<0.001**	0.339	0.998	
Survey 3	47.191	**<0.001**	0.875	0.992	
Organic Carbon (%)					
Survey 1	0.021	0.887	0.942	0.479	0.567
Survey 2	2.093	0.167	0.695	0.930	0.830
Survey 3	4.799	**0.044**	0.999	0.967	0.969
Organic Nitrogen (%)					
Survey 1	21.458	**<0.001**	0.896	0.882	0.790
Survey 2	19.654	**<0.001**	0.893	0.905	0.696
Survey 3	5.798	**0.028**	0.997	0.962	0.982
δ^{13}C					
Survey 1	0.299	0.592	0.546	0.247	0.354
Survey 2	6.727	**0.020**	0.965	1.000	0.999
Survey 3	4.632	**0.047**	0.904	0.978	0.995
δ^{15}N					
Survey 1	39.675	**<0.001**	0.971	0.989	1.000
Survey 2	106.696	**<0.001**	0.299	0.916	1.000
Survey 3	75.955	**<0.001**	0.467	0.970	0.991
C:N					
Survey 1	1.479	0.242	1.000	0.737	0.853
Survey 2	7.115	**0.017**	0.694	0.996	1.000
Survey 3	0.454	0.510	1.000	1.000	0.987
Dissolved Oxygen (mg/L)					
Survey 1	0.543	0.472	0.607	0.803	1.000
Survey 2	6.673	**0.020**	0.956	0.996	0.992
Survey 3	0.069	0.796	0.986	0.968	0.922
Total Abundance					
Survey 1	19.948	**<0.001**	1.000	1.000	1.000
Survey 2	57.557	**<0.001**	0.885	1.000	0.999
Survey 3	6.623	**0.020**	0.997	0.970	0.992
Species Richness					
Survey 1	1.66	0.216	0.840	0.262	0.696
Survey 2	14.666	**0.001**	0.932	0.966	0.777
Survey 3	6.366	**0.023**	0.926	0.739	0.906
Species Diversity					
Survey 1	42.532	**<0.001**	0.926	0.507	0.766
Survey 2	47.925	**<0.001**	0.750	0.843	0.944
Survey 3	9.177	**0.008**	0.973	1.000	0.998
ROV Score					
Survey 1	na	na	na	na	na
Survey 2	76.442	**<0.001**	**<0.001**	0.427	0.298
Survey 3	47.871	**<0.001**	**0.002**	0.113	0.155

Table 5.2-2 Results of the generalised linear model testing for the effect of survey at the two distances closest to the cages (0 and 35/50m) and at the three distances furthest from the cages (500, 1000 and CO). Significant responses ($p < 0.05$) are shown in bold.

	0 m	**35 m**	**500 m**	**1000 m**	**CO**
Redox	**<0.001**	**<0.001**	**0.004**	0.074	0.074
Sulphide	**<0.001**	**0.026**	1.000	1.000	1.000
Organic Carbon (%)	**0.010**	1.000	0.547	1.000	1.000
Organic Nitrogen (%)	0.7443	1.000	1.000	1.000	1.000
$\delta^{13}C$	**0.001**	0.827	1.000	0.456	1.000
$\delta^{15}N$	**<0.001**	1.000	1.000	1.000	1.000
C:N	0.077	0.879	0.615	0.615	0.963
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Total Abundance	**<0.001**	1.000	1.000	1.000	1.000
Species Richness	0.060	**0.004**	0.131	0.131	0.131
Species Diversity	**<0.001**	1.000	0.849	1.000	1.000
ROV Score	**0.032**	1.000	0.784	1.000	1.000

5.2.1.2 Macrofauna

Across all three surveys, 60,747 invertebrates were collected across 234 grab samples, comprising 295 different taxa. Overall, polychaetes were the most abundant taxon at the Lippies lease, making up 68% of the total taxa abundance, followed by crustaceans (22%), molluscs (7%) and nemerteans (1%), echinoderms (1%) and others (< 1%). Polychaetes and crustaceans were present in 100% of the samples, molluscs in 98%, nemerteans in 90% and echinoderms in 80% of the samples.

Forty four percent of the taxa were crustaceans, 31% polychaetes, 17% molluscs, 3% echinoderms, 1% nemerteans and 4% other. The most abundant taxon at the Lippies lease over the three surveys were *Capitella* sp., making up 43% of the total abundance, followed by the amphipods *Byblis mildura* (8%) and *Ampelisca cf. australis* (6%), capitellid polychaete *Mediomastus* sp. (4%), spionid *Paraprionospio coora* (3%), scaligregmatid polychaete *Scalibregma cf. inflatum* (3%), spionid *Prionospio kulin* (2.7%), terebellid *Ampharetid* sp. (2.5%) and other taxa (1.5% or less). All these taxa remained reasonably abundant across all three surveys.

Total abundance

The total abundance was clearly elevated at the cage site compared to the other transect distances in all three surveys (Figure 5.2-11), with the effect of distance significant in all three surveys (Table 5.2-1). There was no significant difference in total abundance between the 35 m and outer controls, 200m and outer controls or between the inner and outer control sites in any of the surveys (Table 5.2-1). There was a significant effect of survey at the cages (0 m) but not at 35 m or more distant sites (500 m, 100 m and CO; Figure 5.2-12, Table 5.2-2). At the cage sites, total abundance increased from an average of 14,369 ind. m^{-2} in survey 1 to 26,193 ind. m^{-2} in survey 2, before decreasing to 6,503 ind. m^{-2} in survey 3.

Species richness

In survey 1, there was no clear pattern of species richness with distance from the cage, but in surveys 2 and 3 species richness was reduced at the cage sites compared to the other distances (Figure 5.2-11, Table 5.2-1). There was no significant difference in species richness between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Table 5.2-1). Apart from the cage sites, species richness increased across all distance and control sites between surveys 1 and 2; however, the effect of survey was only significant at the 35 m sites (Figure 5.2-12, Table 5.2-2).

Species diversity

The species diversity index (H’) was clearly lower at the cage site compared to the other transect distances in all three surveys (Figure 5.2-11), with the effect of distance significant in all three surveys (Table 5.2-1). There was no significant difference in species diversity between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Table 5.2-1). There was a significant effect of survey at the cages (0m) but not at 35m or more distant sites (500 m, 100 m and CO; Figure 5.2-12, Table 5.2-2). At the cage sites, the diversity index value decreased from an average of 1.0 in survey 1 to 0.3 in survey 2, before increasing to 1.9 in survey 3 (Figure 5.2-11).

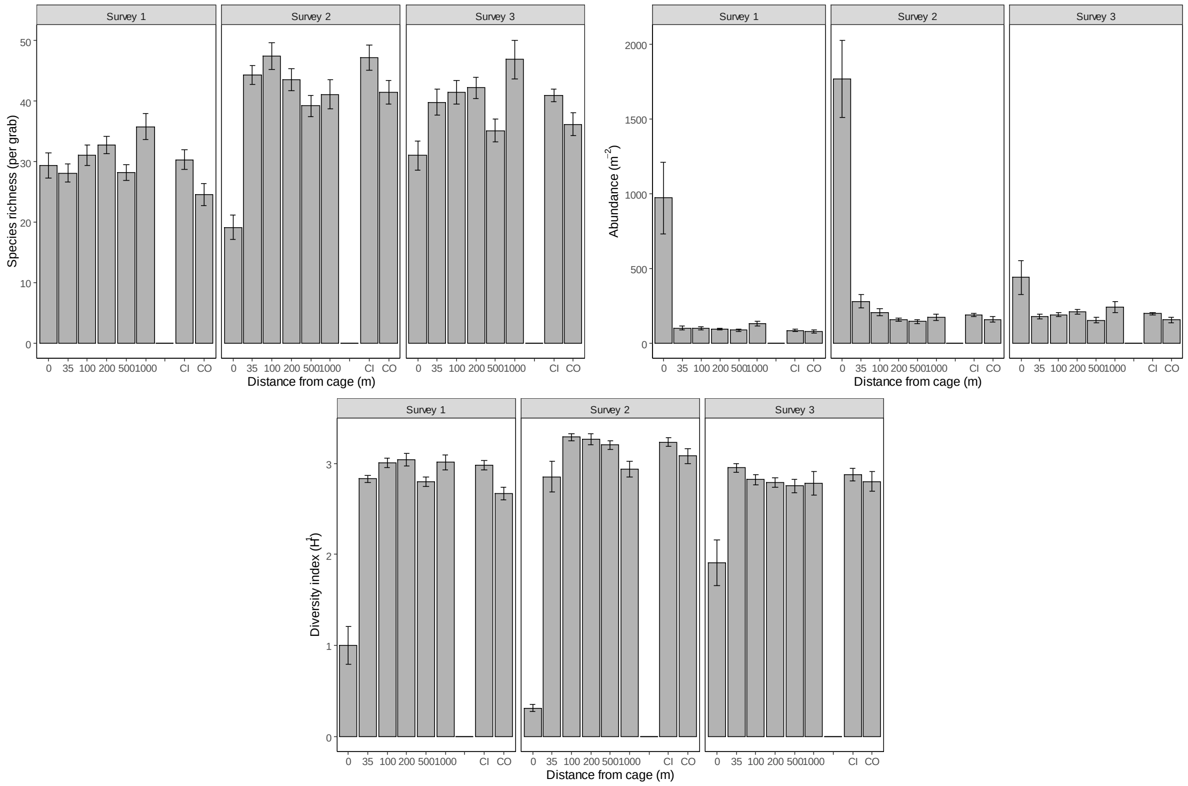

Figure 5.2-11 Average abundance (ind. m-2), species richness (number of species per grab) and species diversity (Shannon-Wiener diversity index, H’) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Lippies lease.

Multivariate analysis

The multivariate GLM showed distance significantly influenced the composition of the faunal assemblages in all three surveys. For the species that showed a significant effect of distance the fourth corner coefficients are plotted using a LASSO penalty for each survey in Figure 5.2-13, 5.2-14 & 5.2-15 (see Appendix A for plots of individual species abundance by distance.)

Of the more common species in survey 1 the polychaete *Capitella* sp., showed the strongest positive relationship with proximity to the cage, and the amphipods *Bybilus mildura* and *Ampelisca cf. australis*, a strong negative relationship with proximity to the cage (Figure 5.2-13). Other species to show strong positive relationships with proximity to the cages included the polychaetes *Euchone limnicola*, *Glycerid* sp., *Leitoscoloplos bifurcates* and opisthobranch gastropod *Philine* sp1. Other species that showed strong negative relationships with proximity to the cages included the amphipod *Phoxocephalid* sp3, the brittle star

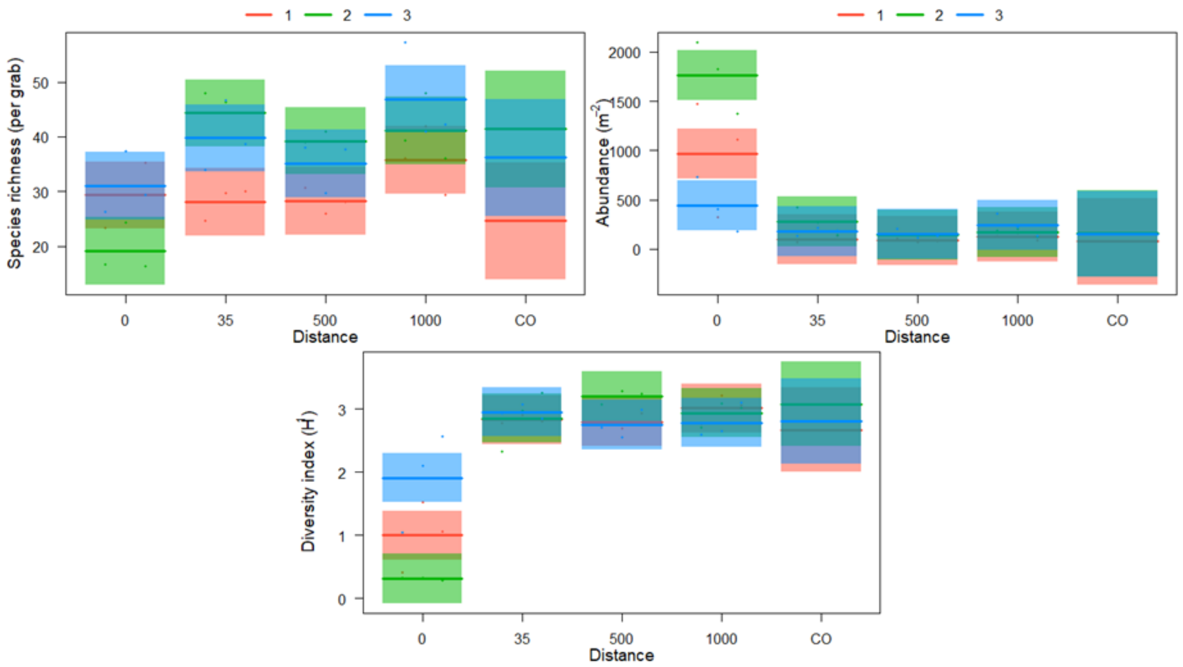

Figure 5.2-12 Average abundance, species richness and species diversity (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO).

Amphuria elandiformis, the tanaid *Apseudid* sp2 and the polychaete, *Armandia* sp. In survey 2 double the number of species showed a significant relationship with distance (Figure 5.2-14), with the polychaete *Capitella* sp. again the most common species showing a positive relationship with proximity to the cages. Other species to show strong positive relationships with proximity to the cages included the other capitellid polychaete *Mediomasutus* sp., the nereid polychaete *Perinereis* sp., bivalves *Theora lubrica* and *Corbula gibba*, gastropods *Syrnola bifasciata* and *Turbonilla fusca,* and the anemone *Edwardsia* sp. Species indicating negative relationships with proximity to the cages included the amphipods *Bybilus mildura* and *Ampelisca cf. australis*, brittle star *Amphuria elandiformis*, bivalve *Pratulum thetidis* and polychaete *Armandia* sp. In survey 3 (Figure 5.2-15) the number of species showing strong positive relationships with proximity to the cages increased relative to surveys 1 and 2. These species included the polychaetes *Pectinaria cf. dodeka*, *Mediomasutus* sp., *Perinereis* sp., *Phyllodocid* sp2, *Leitoscoloplos bifurcates*. *Glycerid* sp. and *Cirratulid* sp2., amphipods *Phoxocephalid* sp2 and *Oedicerotid* sp2., bivalves *Theora lubrica and Corbula gibba*, gastropods *Syrnola bifasciata* and *Turbonilla fusca*, the heart urchin *Echinocardium cordatum* and the cumacean *Dimorphostylis cottoni*. Similarly, there was an increase in the number of species identified to have a strong negative relationship with proximity to the cages; these species included amphipods *Bybilus mildura*, brittle star *Amphuria elandiformis*, tanaid *Apseudid* sp2, bivalves *Pratulum thetidis*, *Ennucula obliqua*, *Channelaxinus adelaideanus* and *Ungulinid* sp., and spionid polychaete *Prionospio kulin*, terebellid polychaete *Terebellides kowinka* and scalibregmatid polychaete *Scalibregma inflatum.*

The unconstrained ordination from the LVM showed a separation of samples based on their species composition and relationship with distance from the cage (i.e. along latent variable 1; Figure 5.2-16). In survey 1, species composition at the 0m and to a lesser extent species at 35 m were distinct from the assemblage found at 100m and further from the cages. Of the ten most important species identified in the LVM, the polychaetes *Capitella* sp., *Euchone limnicola*, and *Glycerid* sp., were characteristic of the assemblages found near the cages and the source of enrichment, whilst the polychaete *Armandia* sp., the brittle star *Amphuria elandiformis* and amphipods *Phoxocephalid* sp3, *Bybilus mildura*, and *Ampelisca cf. australis* were characteristic of the assemblages more distant from the cages and source of enrichment. Variability in species composition at the 1000 m most likely reflected the coarser sediments found at this distance (Figure 5.2-3). Both the bivalve *Corbula gibba* and the amphipod *Aorid* sp1. were particularly common at the 1000 m sites.

In survey 2, a clearer separation in species composition according to distance was apparent, with the assemblages at 0 m and 35 m becoming more distinct and separation emerging between 100 m and 200 m (Figure 5.2-16). The polychaete *Capitella* sp. was again a characteristic species of the community at 0 m but was found in much higher abundances at both 0 m and 35 m compared to survey 1. In survey 2, the bivalve *Theora lubrica* and gastropods *Syrnola bifasciata* and *Turbonilla fusca* were also characteristic of the assemblages of the sites nearest the cages, and more so at 35 m; in survey 1 these three mollusc species were rare or absent across all sites. The species identified in the LVM as most characteristic of sites more distant from the cages in survey 2 were the brittle star *Amphuria elandiformis*, bivalve *Pratulum thetidis*, tanaid *Apseudid* sp1, and the polychaetes *Armandia* sp. *Epidiopatra hupferiana*; all these species were more abundant at 100 m and more distant sites.

Species composition at 0 m and 35 m from the cage was again distinct from that observed at the more distant sites in survey 3, but the assemblages at the other distances appeared more similar unlike the situation in survey 2. *Capitella* sp. were clearly the most abundant species at 0 m but their abundance was markedly lower (~ 4000 ind. m^{-2}) than that observed at 0 m in surveys 1 (~12,600 ind. m^{-2}) and 2 (~25,000 ind. m^{-2}). They were also far less abundant at 35 m in survey 3 (~ 5 ind. m^{-2}) compared to survey 2 (~ 1100 ind. m^{-2}). In survey 3, the bivalve *Theora lubrica* was again identified in the LVM as characteristic of sites in proximity to the cages, but this time it was far more abundant at 0 m than 35 m. Cirratulid and Glycerid polychaetes were also identified as characteristic of the 0m sites; in surveys 1 and 2 they were also predominately found at the sites closest to the cages (0 and 35 m). The mussel *Mytilus galloprovincialis* was characteristic of the sites closest to the cage in all surveys, although not as abundant in surveys 1 and 2 and was only recorded at the sites closest to the cages. The presence of *Mytilus galloprovincialis* is most likely a function of biofouling on farm infrastructure being dislodged. Another mytilid *Amygdalum striatum* was characteristic of the 35 m sites in survey 3 but was very rare or absent at other distances and was not found in the previous 2 surveys. Another species that was far more abundant in survey 3 compared to the previous two surveys, and characteristic of the 0 and 35 m sites, was the heart urchin *Echinocardium cordatum*. The species identified in the LVM as most characteristic of sites distant from the cages in survey 3 were the brittle star *Amphuria elandiformis* and terebellid *Amaena trilobata*, with *Amphuria* more common from 100 m and *Amaena* from 35 m and further from the cages. In survey 3, the tanaid *Kalliapseudes struthi* was identified in the LVM characteristic of the variability in species composition observed at the 1000 m sites.

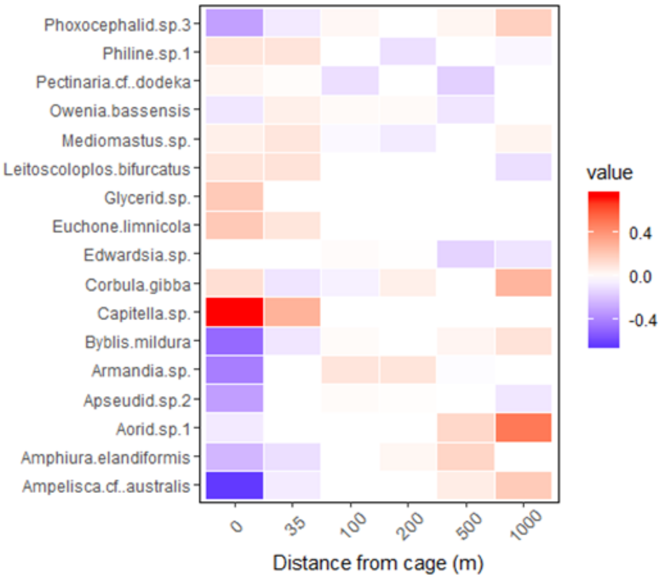

Figure 5.2-13 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the farm in survey 1.

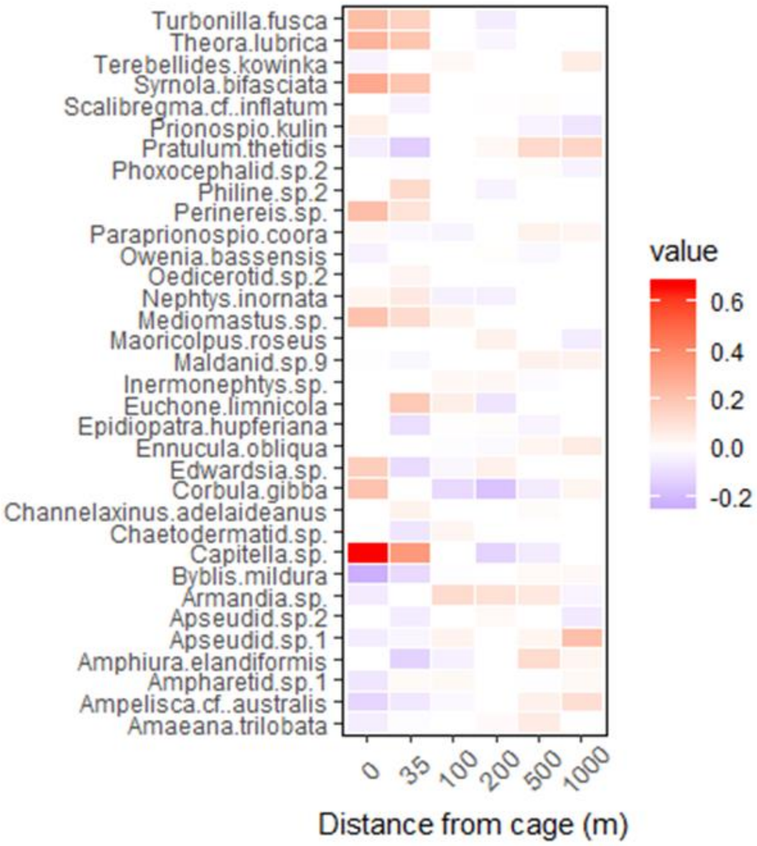

Figure 5.2-14 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the farm in survey 2.

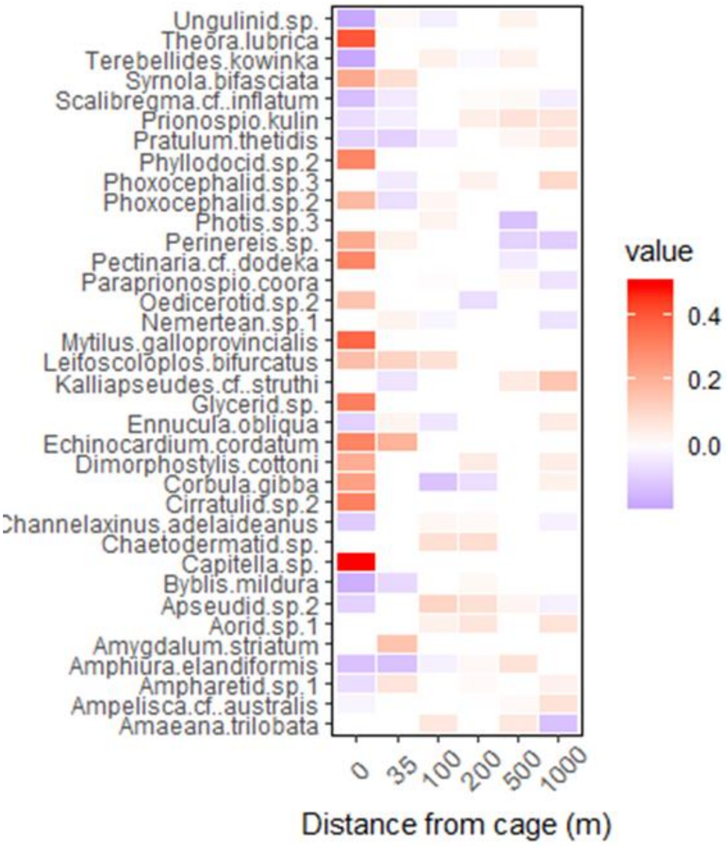

Figure 5.2-15 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the farm in survey 3.

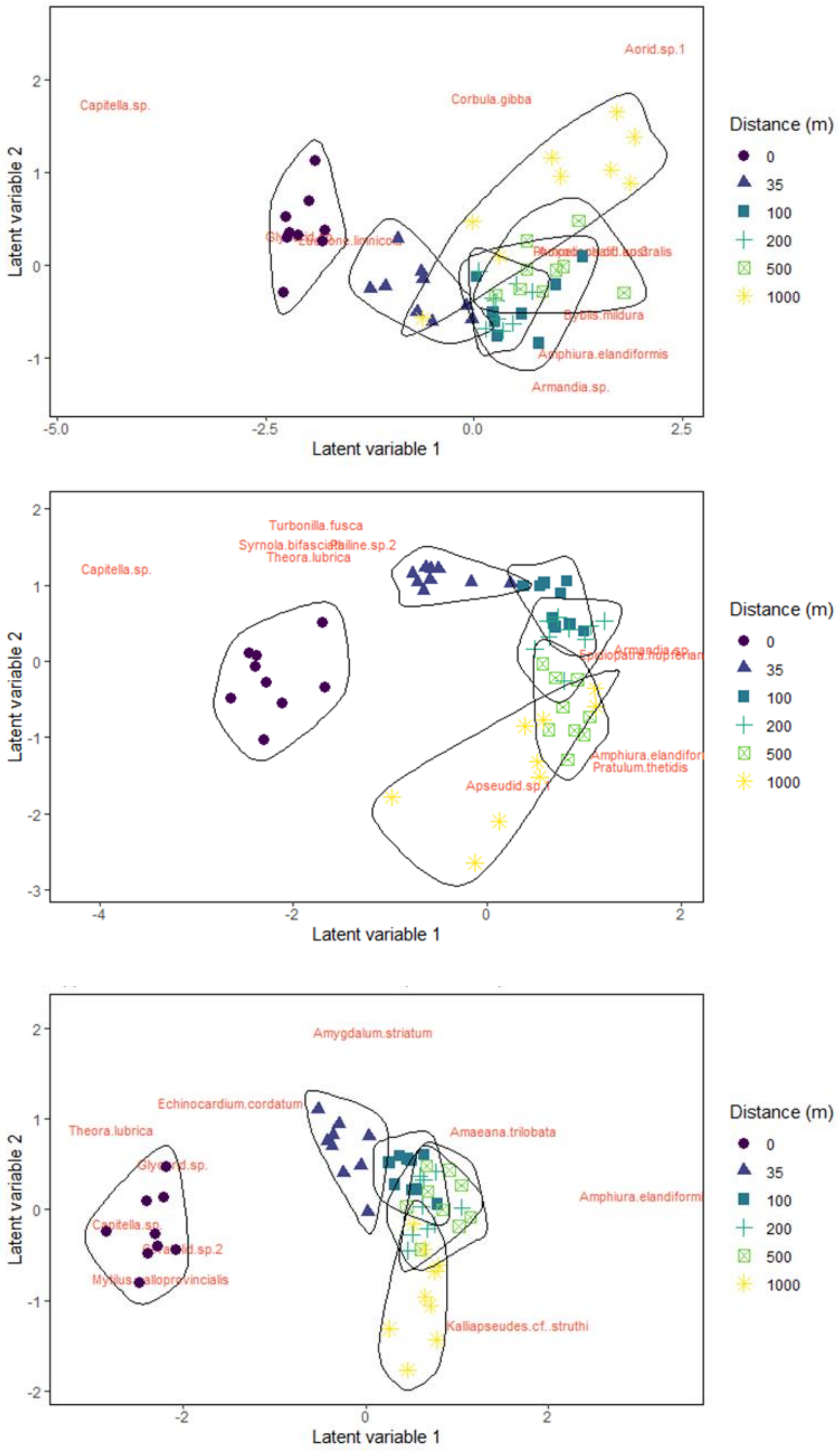

Figure 5.2-16 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1 (top), 2 (middle) and 3 (bottom). Samples are collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

5.2.1.3 Visual

There was a significant effect of distance on ROV health scores, with 0 m sites having the lowest scores of -2 (± 0.3) and 1.2 (± 1.4) for surveys 2 and 3 respectively (Figure 5.2-17, Table 5.2-1). ROV scores improved at 35 m to 3 (± 0.8) in survey 2 and 3.5 (± 0.8) in survey 3, and scores were > 5 from 100 m to 1000 m on the transects and at the inner and outer control sites. There was a significant difference in ROV scores between the 35 m and outer controls in both surveys, with ROV scores lower at the 35 m sites, but there was no significant difference in scores between the 200 m transect sites (35 m for lease boundary) and outer controls or between the inner and outer control sites in each of the surveys (Table 5.2-1). There was a significant effect of survey at the cages (0 m) but not at 35 m or more distant sites (500 m, 100 m and CO; Figure 5.2-18, Table 5.2-2).

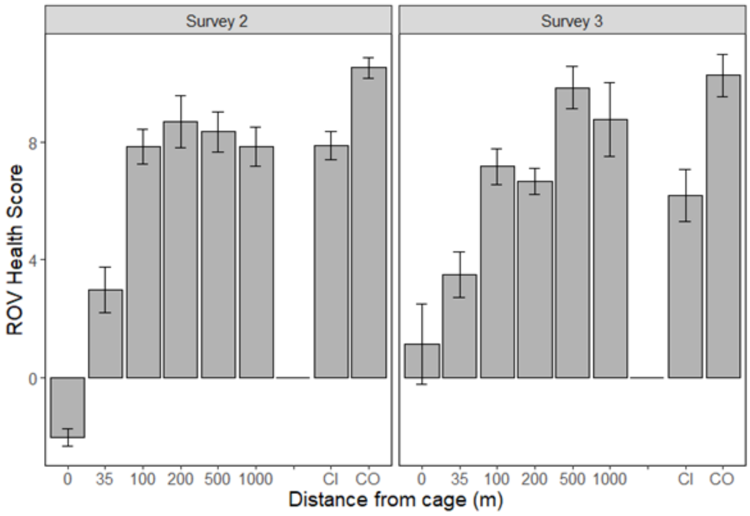

Figure 5.2-17 ROV scores (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for survey 2 and 3 at the Lippies lease. ROV assessments were not conducted in survey 1.

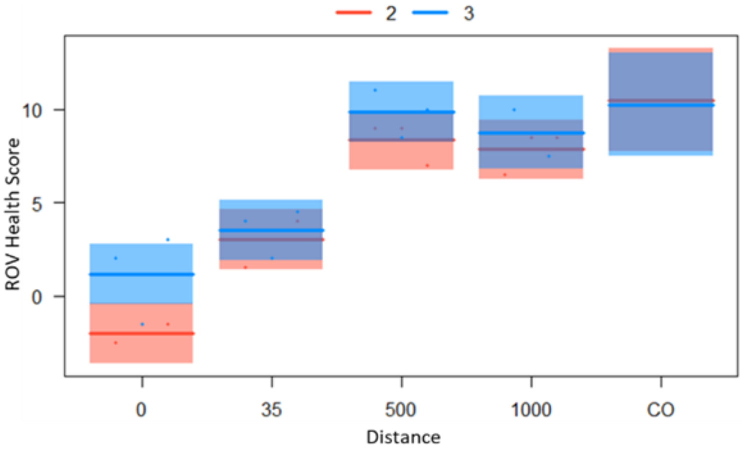

Figure 5.2-18 ROV scores (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500m, 1000m and CO). It is hypothesized that changes at the closer sites were more likely attributed to farming, whereas changes observed at 500m or greater were likely to reflect variation due to other external drivers.

5.2.1.4 Sediment Processes

Sediment respiration measured via dissolved oxygen consumption decreased as a function of distance from cage in both surveys 2 and 3 (Figure 5.2-19, Table 5.2-3). Similarly, respiration measured as dissolved inorganic carbon production (i.e. principally carbon dioxide produced during respiration) decreased as a function of distance from cage in both surveys (Figure

5.2-19, Table 5.2-3). Rates of respiration measured by both oxygen consumption and dissolved inorganic carbon production were significantly higher in survey 2 at the 0m cage sites (~4800 O_2 µmol m^{-2} $h^{-1}$and ~ 5600 DIC µmol m^{-2} h^{-1}) compared to survey 3 at the 0 m cage sites (~2100 O_2 µmol m^{-2} $h^{-1}$and ~ 1800 DIC µmol m^{-2} h^{-1}). At the other distances respiration rates were generally higher in survey 3 compared to survey 2, but the difference was not significant at the two distances tested (i.e. 35 m and 500 m; Figure 5.2-20,Table 5.2-3).

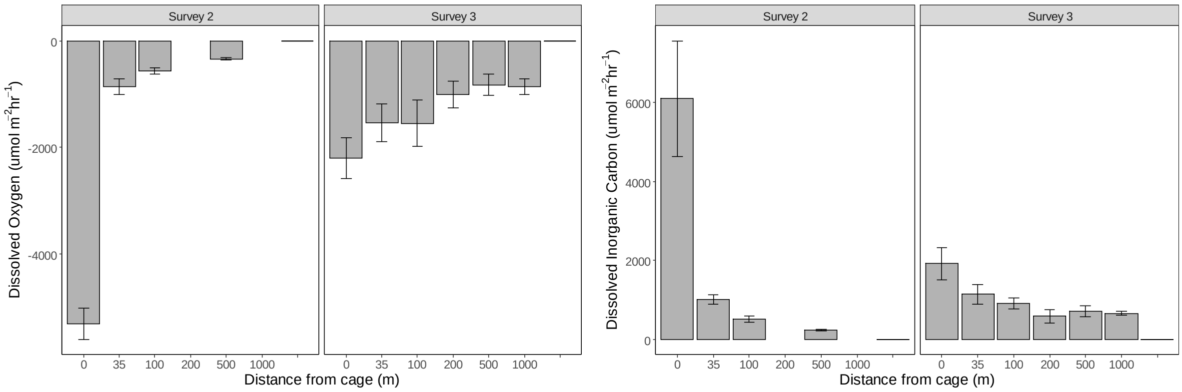

Figure 5.2-19 Sediment respiration: dissolved oxygen and dissolved inorganic carbon fluxes (umol m-2 h-1) (± SE) with distance from the cages on transects in survey 2 and 3 at the Lippies lease. Sediment flux assessments were not conducted in survey 1 or at inner and outer controls.

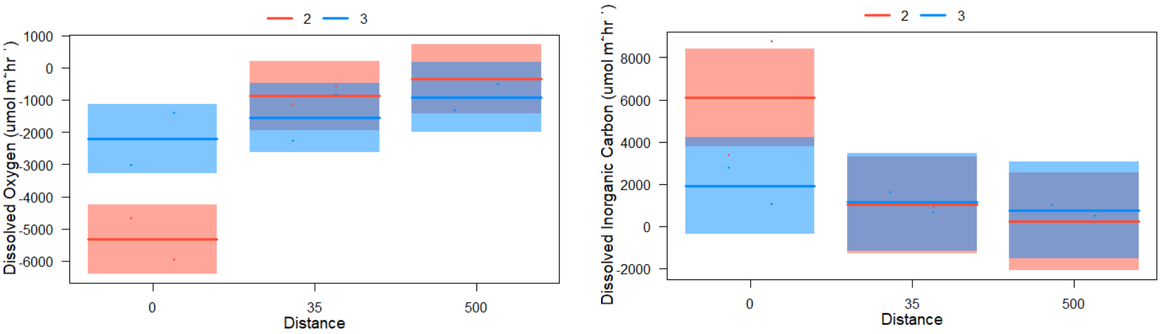

Figure 5.2-20 Sediment respiration rates (umol m^{-2} h^{-1}) for dissolved oxygen and dissolved inorganic carbon (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distance furthest from the cage (500 m). It is hypothesized that changes at the closer sites were more likely attributed to farming, whereas changes observed at 500m or greater were likely reflecting variation due to other, external drivers.

Fluxes of ammonium also decreased as a function of distance from cage in both surveys (Figure 5.2-21, Table 5.2-3). Although average ammonium fluxes were higher in survey 3 at 0, 35 and 500 m, the difference between surveys was not significant (Figure 5.2-22, Table 5.2-3). For nitrate + nitrite, fluxes were directed into the sediment (i.e. uptake) at several sites in both surveys (Figure 5.2-21). In survey 2, there was a significant effect of distance, with the flux into the sediment decreasing with distance from the cage and at 500m, there was a release of nitrate + nitrite from the sediment (Table 5.2-3). In survey 3, there was also a general trend of nitrate + nitrate shifting from net uptake to net release as a function of distance from the cage; however, the fluxes were quite variable, and the effect of distance was not significant (Table 5.2-3). The rate of nitrate + nitrate uptake was significantly higher in survey 2 compared to survey 3, but there was no significant difference in rates at the two other distances tested (35 and 500 m; Figure 5.2-22,Table 5.2-3).

Fluxes of phosphate were low (< 4 µmol m^{-2} h^{-1}) across both surveys and all distances except for the cage site in survey 2, where the phosphate flux was 139 µmol m^{-2} h^{-1} (Figure 5.2-23). Despite the elevated phosphate flux at 0 m in survey 2, the effect of distance and survey were not significant (Figure 5.2-24, Table 5.2-4).

Table 5.2-3 Results of the generalised linear model testing for the effect of distance for sediment nutrient fluxes for surveys 2 and 3 and results of the generalised linear model testing for the effect of survey at the two distances closest to the cages (0 and 35) and at the distance furthest from the cages that was measured in both surveys (i.e. 500 m). Significant responses ($p < 0.05$) are shown in bold.

			Effect of survey		
	Effect of Distance		**0m**	**35 m**	**500 m**
	F ratio	**p**	**p**	**p**	**p**
Dissolved Oxygen					
Survey 2	38.648	**<0.001**	**<0.001**	0.759	0759
Survey 3	5.052	**0.048**			
Dissolved Inorganic					
Survey 2	13.143	**0.011**	**0.038**	1.000	1.000
Survey 3	8.453	**0.016**			
Ammonium					
Survey 2	11.98	**0.013**	1.000	1.000	1.000
Survey 3	15.676	**0.003**			
Nitrate + Nitrite					
Survey 2	40.476	**<0.001**	**<0.001**	0.879	0.903
Survey 3	1.647	0.228			
Phosphate					
Survey 2	4.787	0.071	0.108	1.000	1.000
Survey 3	2.344	0.157			

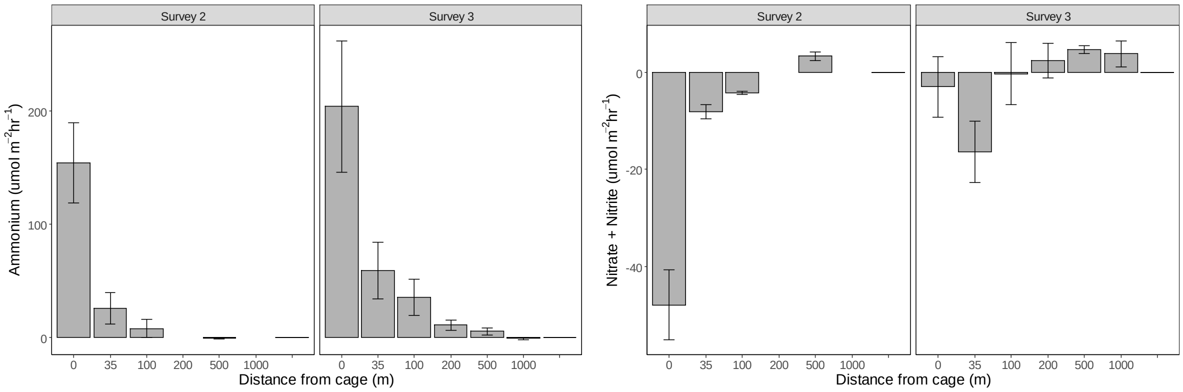

Figure 5.2-21 Sediment ammonium and nitrate + nitrite fluxes (umol m^{-2} h^{-1}) (± SE) with distance from the cages on transects in surveys 2 and 3 at the Lippies lease. Sediment flux assessments were not conducted in survey 1 or at inner and outer controls.

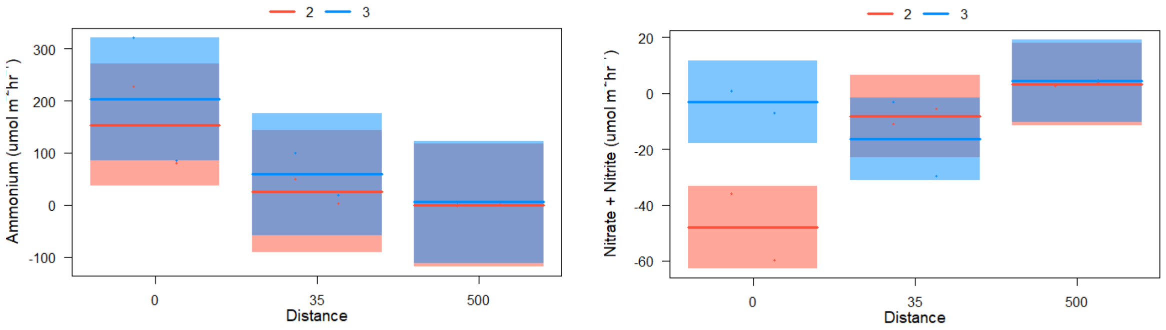

Figure 5.2-22 Sediment flux rates (umol m^{-2} h^{-1}) for ammonium and nitrate + nitrite (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distance furthest from the cage (500 m).

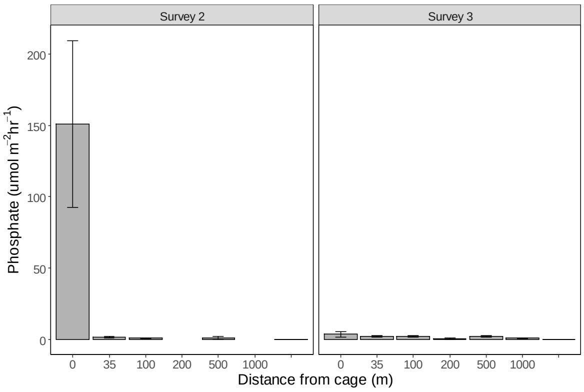

Figure 5.2-23 Sediment phosphate flux (umol m^{-2} h^{-1}) (± SE) with distance from the cages on transects in surveys 2 and 3 at the Lippies lease. Sediment flux assessments were not conducted in survey 1 or at inner and outer controls.

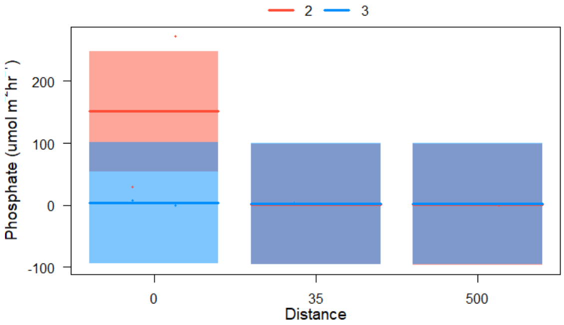

Figure 5.2-24 Sediment flux rates (umol m^{-2} h^{-1}) for phosphate (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distance furthest from the cage (500 m).

5.2.1.5 Historical comparison

Redox potential values (mV) were variable across surveys pre farming, ranging from an average of 111 mV in 2004 to 278 mV in 2016 at control sites. Average values at the control sites were higher in the more recent surveys, ranging from 342 – 674 mV (Table 5.2-4). Redox values at the compliance sites were similar to the control sites in the baseline surveys, while in two of the three surveys since farming commenced average redox values were lower (albeit highly variable in 2017 survey 1) at the compliance sites compared to the control site. Sulphide concentrations at the control site were negligible (<2 µM) in both 2004 and 2016 years. Across the three recent surveys, average sulphide concentrations ranged from 0 – 4 µM at the control site location. Except for one reading at a compliance site in 2017 survey 2 (31 µM, site average 10.4 µM), sulphide concentrations were also negligible (<2 µM) at compliance sites pre and post farming. Farm sites showed the highest sulphide concentrations (ranging from 22 - 124 µM) and the lowest redox potential (ranging from 103-376 mV) relative to the compliance and control sites across the three recent surveys.

Macrofaunal abundance (ind. m^{-2}) was variable across the two baseline surveys prior to farming, ranging from an average of 4311 ind. m^{-2} in 2004 to 2474 ind. m^{-2} in 2016 at the control site. A similar range in abundances was observed across the three surveys conducted since farming commenced, ranging from an average of 1753 ind. m^{-2} to 3477 ind. m^{-2} at the control site. Average abundance at the compliance sites showed a similar pattern and whilst it was lower relative to the control sites, this pattern was consistent across all surveys, both pre and post commencement of farming. As such the difference likely reflects natural spatial variation between the compliance and control sites independent of farming.

The average number of macrofaunal families at the control site ranged from 17 to 27 in 2004 and 2016 respectively, and from 24 to 39 across the more recent surveys. A similar pattern was observed at the compliance sites. Macrofauna diversity ranged from 0.7 to 2.6 in 2004 and 2016 respectively, and from 1.9 to 2.9 across the more recent surveys. Diversity was typically higher at the compliance compared to the control sites across all surveys, both pre and post the commencement of farming. Apart from the number of families in survey 1, farm sites generally showed the highest abundance, lowest diversity and lowest number of families compared to the compliance and control sites.

The unconstrained ordination from the LVM showed a separation of samples based on their family composition and relationship with sample location (i.e. CI, CO and 0 m) and survey (Figure 5.2-25). Not surprisingly, there was a clear separation of farm (0 m) sites from the compliance and control sites with Capitellidae (mostly *Capitella* sp.), the key indicator family of farm sites. The families Semelidae (the introduced bivalve *Theora lubrica*) and Glyceridae (polychaete *Glycera tridactyla*) were far less common but characteristic of the farm sites and more notable in the 2018 survey. There was no clear or consistent separation in family composition between compliance and control sites across the surveys pre and post the commencement of farming. However, the 2004 community was closely grouped and distinct from the 2016 baseline and 2017-18 surveys. The families driving this separation were amphipods from the family Maeridae and Isaeidae, Sphaeromatidae isopods and Nuculanidae bivalves, largely only recorded in the 2004 survey. In contrast, squat lobsters from the family Galatheidae and callinassid shrimp were largely only found in the other surveys. It is important to note it was not changes in the most common family Ampeliscidae (amphipods) at the compliance and control sites across all surveys that drove the separation but changes in the less common families.

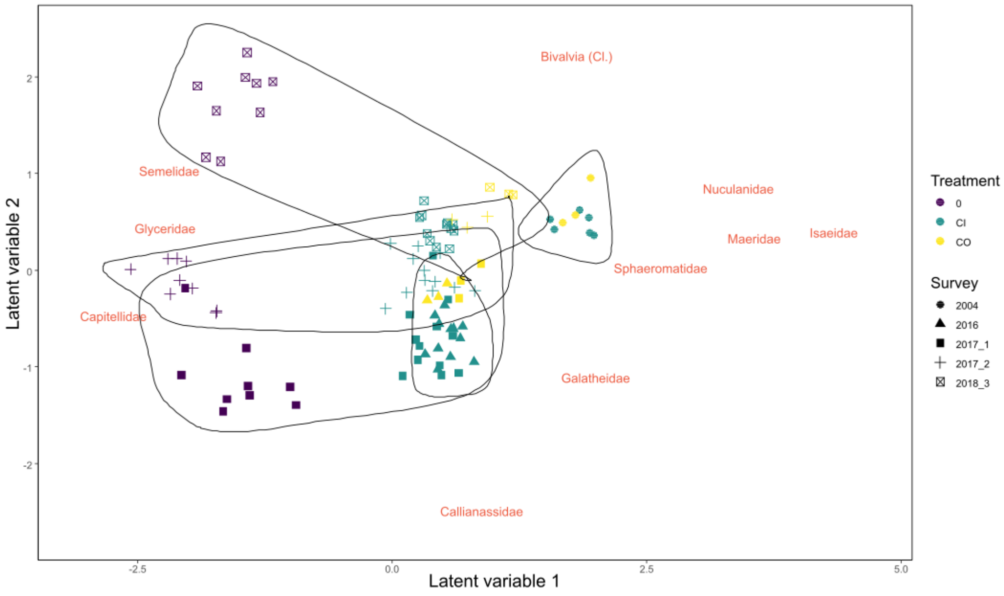

Figure 5.2-25 Biplot showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for inner and outer control sites from baseline surveys in 2004, 2016 and IMAS surveys 1 -3 conducted over 2017-18. For comparison with the effects of farming, sites sampled directly adjacent to the cages (0m) in the surveys conducted since the commencement of farming are also included. Survey and sample location are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

Table 5.2-4 Historical comparison of average physiochemical and macrofaunal parameters between pre- 2004 and 2016) and post-farming (three recent IMAS surveys) baselines surveys at the Lippies lease. Note, for sulphide and redox, the 2017-2018 data only include one site (ELM3).

	Sulphide (µM)			**Redox (mV)**			**Number of Families**			**Abundance (indiv. m^{-2})**			**Diversity (H')**		
Surveys	**AVG**	**SD**	**COUNT**	**AVG**	**SD**	**COUNT**	**AVG**	**SD**	**COUNT**	**AVG**	**SD**	**COUNT**	**AVG**	**SD**	**COUNT**
2004															
compliance	0.0	(0.1)	6	113	(18)	6	19	(5)	6	3943	(1504)	6	1.0	(0.3)	6
control	0.0	(0)	3	111	(52)	3	17	(3)	3	4311	(1625)	3	0.7	(0.1)	3
farm															
2016															
compliance	1.7	(2.6)	12	282	(75)	12	27	(3)	12	1975	(372)	12	2.6	(0.2)	12
control	0.2	(0.2)	3	278	(83)	3	27	(5)	3	2474	(640)	3	2.3	(0.3)	3
farm															
2017 survey 1															
compliance	1.1	(1.9)	3	302	(144)	3	25	(4)	12	1295	(377)	12	2.7	(0.2)	12
control	4.0	(4.6)	3	421	(15)	3	24	(5)	3	1753	(109)	3	2.3	(0.2)	3
farm	33.8	(16.4)	5	134	(56)	5	25	(5)	9	14369	(10659)	9	0.8	(0.5)	9
2017 survey 2															
compliance	10.4	(17.9)	3	346	(32)	3	37	(6)	12	2705	(547)	12	2.9	(0.2)	12
control	4.1	(7)	3	342	(20)	3	39	(5)	3	3477	(586)	3	2.5	(0.5)	3
farm	123.6	(88.8)	5	178	(65)	5	16	(5)	9	26193	(11508)	9	0.2	(0.1)	9
2018 survey 3															
compliance	0.3	(0.1)	3	420	(19)	12	35	(3)	12	2958	(518)	12	2.5	(0.2)	12
control	0.0	(0)	3	673	(13)	3	30	(1)	3	3022	(374)	3	1.9	(0.5)	3
farm	22.0	(11.2)	5	376	(31)	5	27	(7)	9	6504	(5040)	9	1.8	(0.8)	9

5.2.2 Discussion

The benthic community at Lippies was highly diverse with 165 different taxa recorded at the control sites. Crustaceans were the most abundant (47% of total abundance), followed by polychaetes (34%), molluscs (13%) and echinoderms (3%). The most abundant taxa were the amphipods *Byblis mildura* and *Ampelisca cf. australis,* polychaetes *Paraprionospio coora, Prionospio kulin, Scalibregma cf. inflatum, Mediomastus sp., Ampharetid sp.1,* bivalve *Pratulum thetidis,* brittle star *Amphiura elandiformis* and *Nemertean sp.1.*

Response to enrichment

The overall benthic community changed discernibly with proximity to the cages, and that response pattern varied between surveys. The variation appeared to be influenced by the stocking regime (intensity of farming); the grid had been stocked for seven months at survey 1, thirteen months at survey 2 (with feed inputs increasing over this period), whilst in survey 3 the grid had been largely fallow for the preceding three months.

The biodiversity metrics, species richness (S), total abundance (N), and Shannon-Weaver Diversity Index (H’) showed response patterns consistent with prior research and the guidelines previously established for the south (Macleod and Forbes 2004). Total abundance peaked at the cage sites, whilst species diversity and richness were typically reduced at the cage sites before peaking at the intermediate distances from the cage. This is in line with previous research and the expectation that highly impacted sites can be less diverse (Keeley et al. 2015; Macleod and Forbes 2004; Pearson and Rosenberg 1978). In survey 1 species diversity, but not richness was reduced at the cage sites compared to the controls but in the next survey both diversity and richness were significantly lower at the cage sites compared to the control sites. This is consistent with the longer stocking period and increase in feed inputs at survey 2 compared to survey 1. According to the guidelines produced for sandy and more exposed sites by Macleod and Forbes (2004), these metrics are consistent with a moderate (diversity index H’ >1<2) and major impact (diversity index H’ <1, No spp. <50% ref) in each survey, respectively. The increase in total abundance at the cage sites relative to the control sites in surveys one and two (~x10-12 ref) is considered indicative of major effects for a sandy site. In survey 3 after the grid had been largely fallow for three months all three metrics were indicative of improved conditions at the cage sites, falling into the ‘minor to moderate effects’ impact categories for an exposed sandy site; total abundance was <x3, the diversity index H’ ~2 and the number of species was >80% relative to reference conditions at the control sites.

Community structure at the cages was also characteristic of sites impacted by organic enrichment (Figure 5.2-26); *Capitella* sp., was the dominant taxon making up 88, 99 and 62% of the total abundance in surveys one, two and three, respectively. Several other taxa were characteristic of the cage sites, but aside from another capitellid polychaete, *Mediomastus* sp., they were not common. However, in survey 3 there was a notable increase in the abundance of several taxa at the cage sites, including the polychaetes *Pectinaria cf. dodeka* and *Phyllodocid* sp.2, heart urchin *Echinocardium cordatum,* dogwhelk *Nassarius nigellus* and two introduced bivalves *Corbula gibba* and *Theora lubrica.* This result likely reflects improved sediment conditions. Macleod et al. (2008) found *Nassarius nigellus*, *Theora lubrica* and *Corbula gibba* indicative of recovering sediments and minor/moderate impacts and that *Echinocardium cordatum* appeared to thrive at low levels of enrichment. Edgar et al. (2005) in a metanalysis of macrobenthic and sediment data across 20 leases in south-eastern Tasmania highlighted the dominance of the dogwhelk *Nassarius nigellus,* the bivalve *Mysella donaciformis,* the polychaete *Terebellides* sp., the heart urchin *Echinocardium* *cordatum* and the introduced bivalve *Corbula gibba* in communities within the farm lease area but distant from the cages. Further from the cage, the change in the community was also consistent with improving sediment conditions, with several other taxa increasing in abundance at the intermediate distances (35-200 m from the cage). This included the polychaetes *Prionospio kulin*, *Scalibregma cf. inflatum*, *Terebellides kowinka*, *Ampharetid* sp.1, *Amaeana trilobata*, tanaid crustacean *Apseudid* sp.2, bivalve *Channelaxinus adelaideanus.* These taxa were likely benefiting from low levels of enrichment and indicate minor effects.

In this study, at the more distant sites which were 500 m and greater from the cages, many of these species remained common, but the presence of species such as the bivalve *Pratulum thetidis*, amphipod *Bybilus mildura* and brittle star *Amphiura elandiformis* characterised a different type of community where there appeared to be no evidence of the impacts of organic enrichment. These results are consistent with previous findings of a state-wide assessment of benthic monitoring at salmon farms in Tasmania (Edgar et al. 2010). Subtle changes to macrofaunal communities were evident out to at least 35 m from the lease boundary (i.e., compliance sites) or 50-150 m from cages (Edgar et al. 2010). In this study the subtle changes in macrofaunal communities were evident out to 200 m from the cages, noting the inner controls (CI) or compliance sites were on average 100 m from cages at Lippies.

It is also important to acknowledge the potential challenge of spatial confounding when interpreting the patterns in the context of enrichment. The 1000 m sites had a different macrofaunal community but also a different sediment type. The coarser sediments at 1000 m compared to the other transect distances and control sites was the likely driver of the different macrofaunal community in this instance, with the four crustacean taxa notably more common, namely the tanaids *Kalliapseudes struthi* and *Apseudid* sp.1 and amphipods *Aorid* sp.1. and *Ampelisca cf. Australis* being those often found in sandier sediments. This highlights the importance of establishing baseline conditions prior to farming and monitoring through time to distinguish farm induced change from natural variability. Nonetheless, the inclusion of control sites that represent a broader range of environmental conditions and ecology and not only the conditions at the lease is imperative when assessing impacts.

Changes in sediment chemistry were also consistent with the response to organic enrichment, but most of the chemical parameters appeared less sensitive than the biological measures. Redox potential was depleted, and sulphide concentrations elevated at the sites in closer proximity to the cages (i.e. 0 & 35 m), and in survey 3 both measures improved, consistent with the period of fallowing. In contrast, C and N signatures (content, ratio and isotopes) were indicative of enrichment effects, but typically only at the 0 m cage site with $\delta^{15}N$ and the nitrogen (%) content the most sensitive measures. Similarly, previous research in Tasmania reported most abiotic metrics were not sensitive enough to detect the more subtle effects of farming (Edgar et al. 2005; Macleod et al. 2004).

The sediment process measurements provided greater insight into the broader footprint of organic loading (Figure 5.2-27). Rates of organic matter mineralisation were clearly elevated

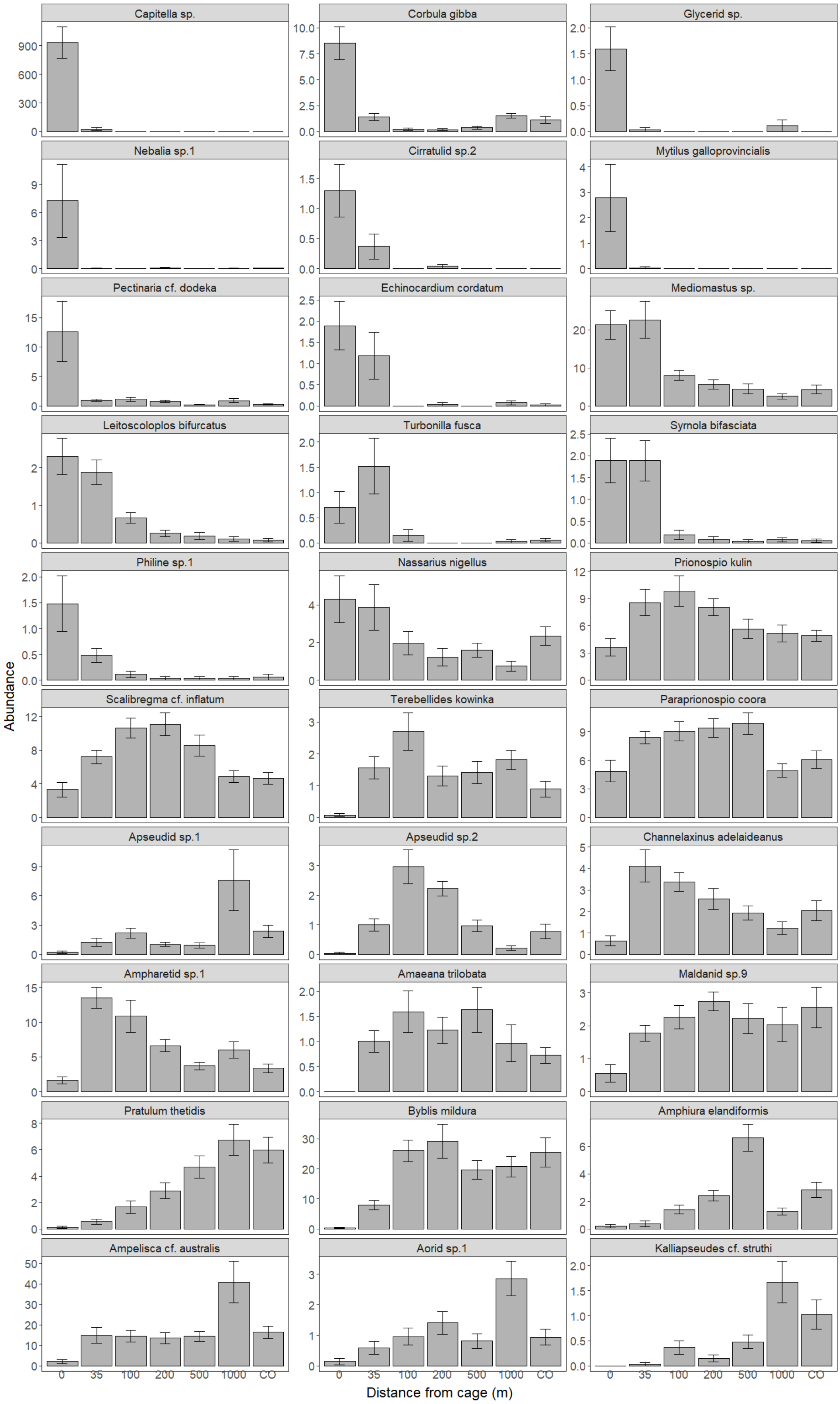

Figure 5.2-26 Average abundance (ind. per grab) of key taxa with distance from the cages (on transects) and at inner (CI) and outer controls (CO) across surveys at the Lippies lease.

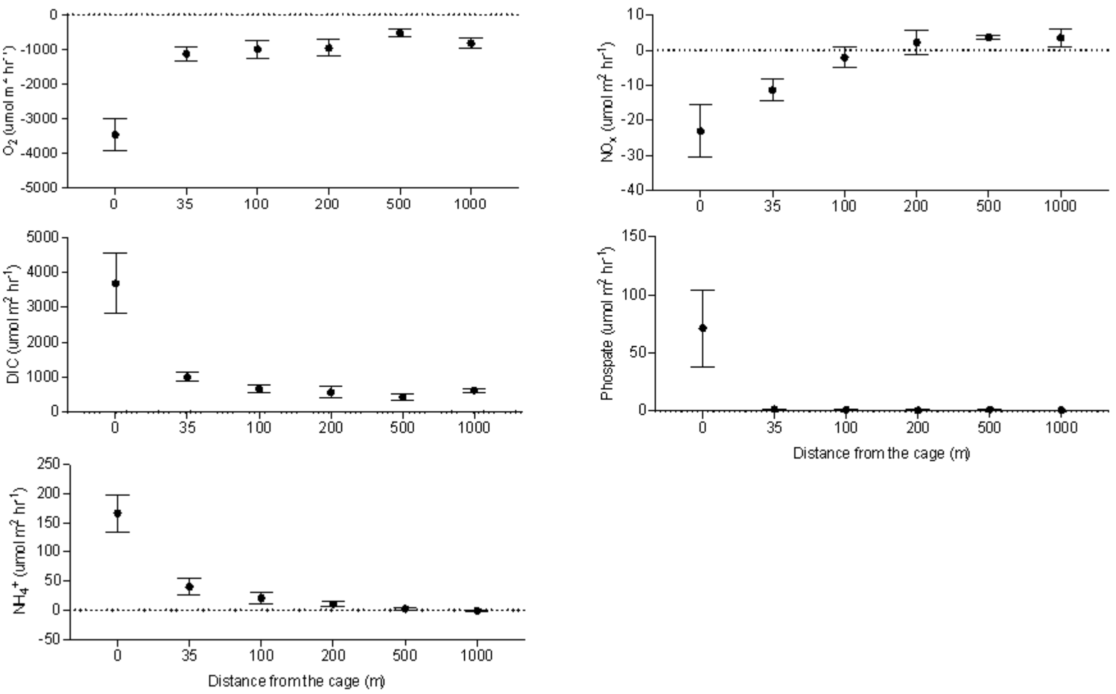

Figure 5.2-27 Summary of sediment fluxes at the Lippies lease. Fluxes of oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages averaged (± SE) across surveys.

at the 0 m cage sites relative to the other distances, a finding consistent with the physico-chemical and macrofaunal response parameters. The bulk of organic matter deposition and processing that causes more significant benthic changes occurred under or very close to (< 35 m) the cages. Importantly though, the process measurements documented the more extensive, albeit subtler, spatial input and influence of organic matter deposition. Elevated respiration rates and ammonia production were evident out to 100 -200 m from the cages, even though at much lower levels than at the cage. This was consistent with the more subtle changes in benthic communities evident at the same spatial scales. Another important finding was the level of responsiveness observed in the process rates of the near cage sediments. In the second round of process measurements undertaken following three months of fallowing, oxygen consumption and dissolved inorganic carbon production (principally carbon dioxide) were twofold lower compared to the previous assessment when fish production biomass was close to its peak within the study grid. Although rates of ammonia production were similar across surveys, the uptake of nitrate + nitrite was much lower following fallowing. Interestingly, process rates at the other distances were more elevated in the second survey compared to the first. This may reflect the influence of higher temperatures in March compared with the first survey undertaken in September and the resultant influence on metabolic rates. It may also reflect a reduced ecophysiological responsiveness at the more distant sites given the inherently lower faunal numbers and stability at these sites and between surveys relative to the cage sites. At the cages, faunal numbers significantly reduced between surveys. For example, there was a fourfold decline in faunal abundance from total 26,193 ind. m^{-2} in the first survey to 6503 ind. m^{-2} in the second survey.

The strong benthic effect gradient near the cages was reflected in the visual health score. The cage sites clearly had the lowest health scores and the sites 35 m from cages were also clearly more affected than the more distant sites. Not surprisingly though, the more subtle effects revealed by the detailed assessment of the faunal communities and process rates were not distinguishable from the visual health score. The visual assessment method could discriminate broad categories of impact, i.e. what would be considered minor to major impacts, but was less able to distinguish between no evidence of impacts to minor impacts.

Implications for monitoring and management

The benthic response at the Lippies lease was largely consistent with the findings of previous research in southeast Tasmania. Highly enriched sediment conditions and major impacts were restricted to the immediate proximity of the cage at the 0 m sites where deposition rates were known to be significantly elevated relative to other distances (see chapter 7.3). This was further illustrated by the markedly higher processing rates measured at the 0 m sites compared to all other distances. However, at 35 m from the cage there were still notable changes in the faunal community. More subtle effects were also evident out to 100-200 m from the cages based on changes in faunal community composition. This gradient was reflected in the change in process rates with distance from the cage and much lower rates of organic matter deposition relative to that seen immediately adjacent to the cages. The key physico-chemical measures were consistent with moderate to minor benthic effects as described by Macleod and Forbes (2004) for sandy/exposed sites. Changes in the biotic and abiotic response parameters between surveys also demonstrated the role of, and response to, farm management with improved sediment conditions from most parameters at the cage site in the third survey after three months of fallowing.

From a monitoring perspective, the results also documented the value of the visual health assessments that are a critical element of ongoing monitoring in Tasmania. Although the method may not have the sensitivity to reliably discriminate the more subtle benthic effects, the visual scores were in close agreement with the other parameters in detecting the major to minor effects observed at cage and 35 m sites. Consistent with the assessments at the other locations in this study, the findings at Lippies highlight the importance of establishing baseline conditions prior to farming and ongoing monitoring when distinguishing farm induced changed from natural variability.

5.2.3 References

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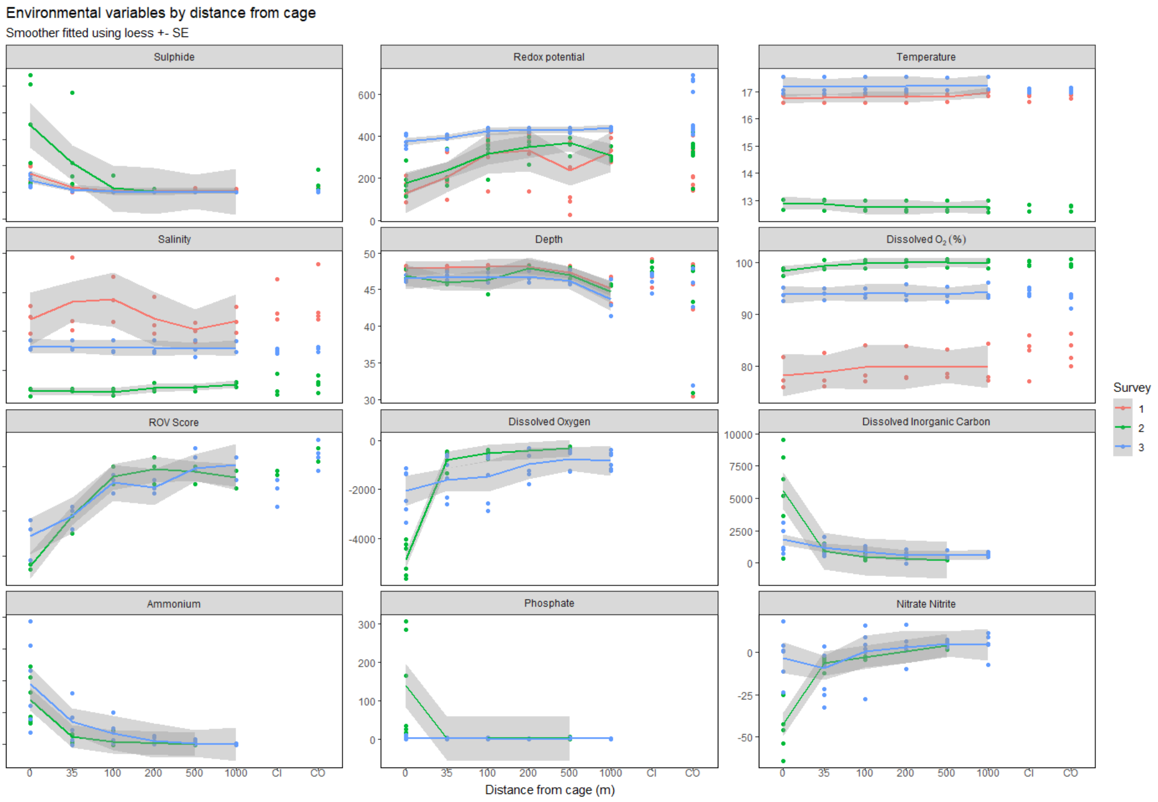
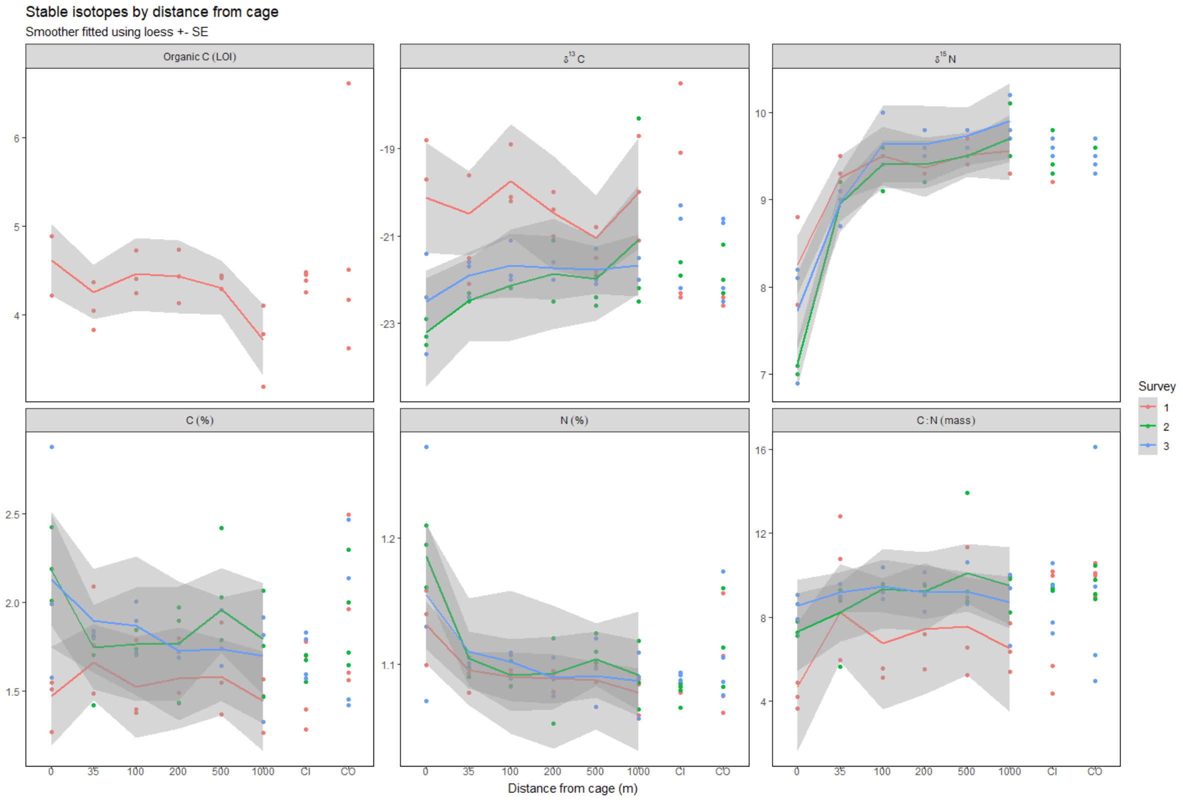
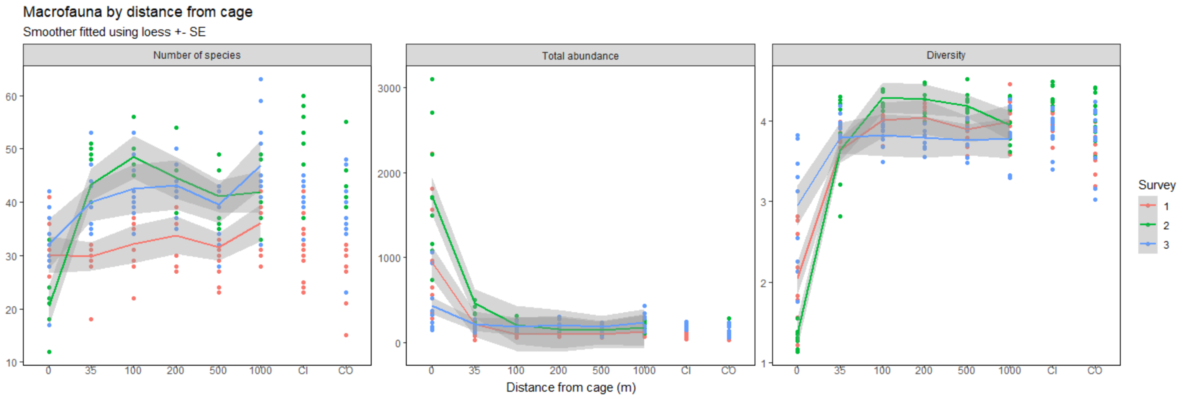
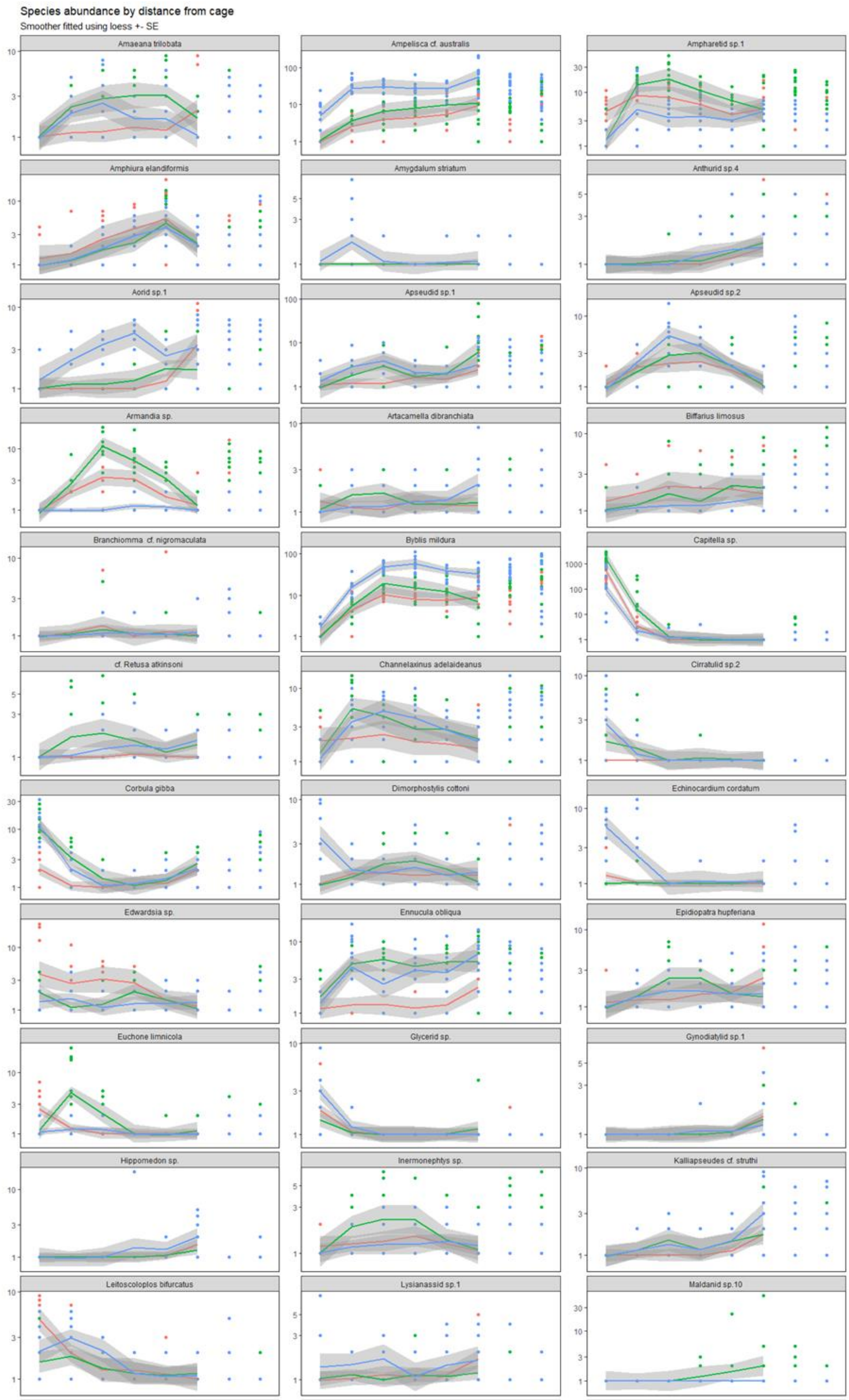
5.2.4 Appendix




Stable isotopes by distance from cage
Smoother fitted using loess +- SE
Organic C (LOI)
δ13C
δ15N
C (%)
N (%)
C : N (mass)
6
5
4
-19
-21
-23
10
9
8
7
2.5
2.0
1.5
1.2
1.1
16
12
8
4
0
35
100
200
500
1000
CI
CO
Distance from cage (m)
Survey
1
2
3


Macrofauna by distance from cage
Smoother fitted using loess +- SE
Number of species
Total abundance
Diversity
60
50
40
30
20
10
3000
2000
1000
0
4
3
2
1
0
35
100
200
500
1000
CI
CO
Distance from cage (m)
Survey
1
2
3


Species abundance by distance from cage
Smoother fitted using loess +- SE
Amaeana trilobata
Ampelisca cf. australis
Ampharetid sp.1
Amphiura elandiformis
Amygdalum striatum
Anthurid sp.4
Aorid sp.1
Apseudid sp.1
Apseudid sp.2
Armandia sp.
Artacamella dibranchiata
Biffarius limosus
Branchiomma cf. nigromaculata
Byblis mildura
Capitella sp.
cf. Retusa atkinsoni
Channelaxinus adelaideanus
Cirratulid sp.2
Corbula gibba
Dimorphostylis cottoni
Echinocardium cordatum
Edwardsia sp.
Ennucula obliqua
Epidiopatra hupferiana
Euchone limnicola
Glycerid sp.
Gynodiatylid sp.1
Hippomedon sp.
Inermonephtys sp.
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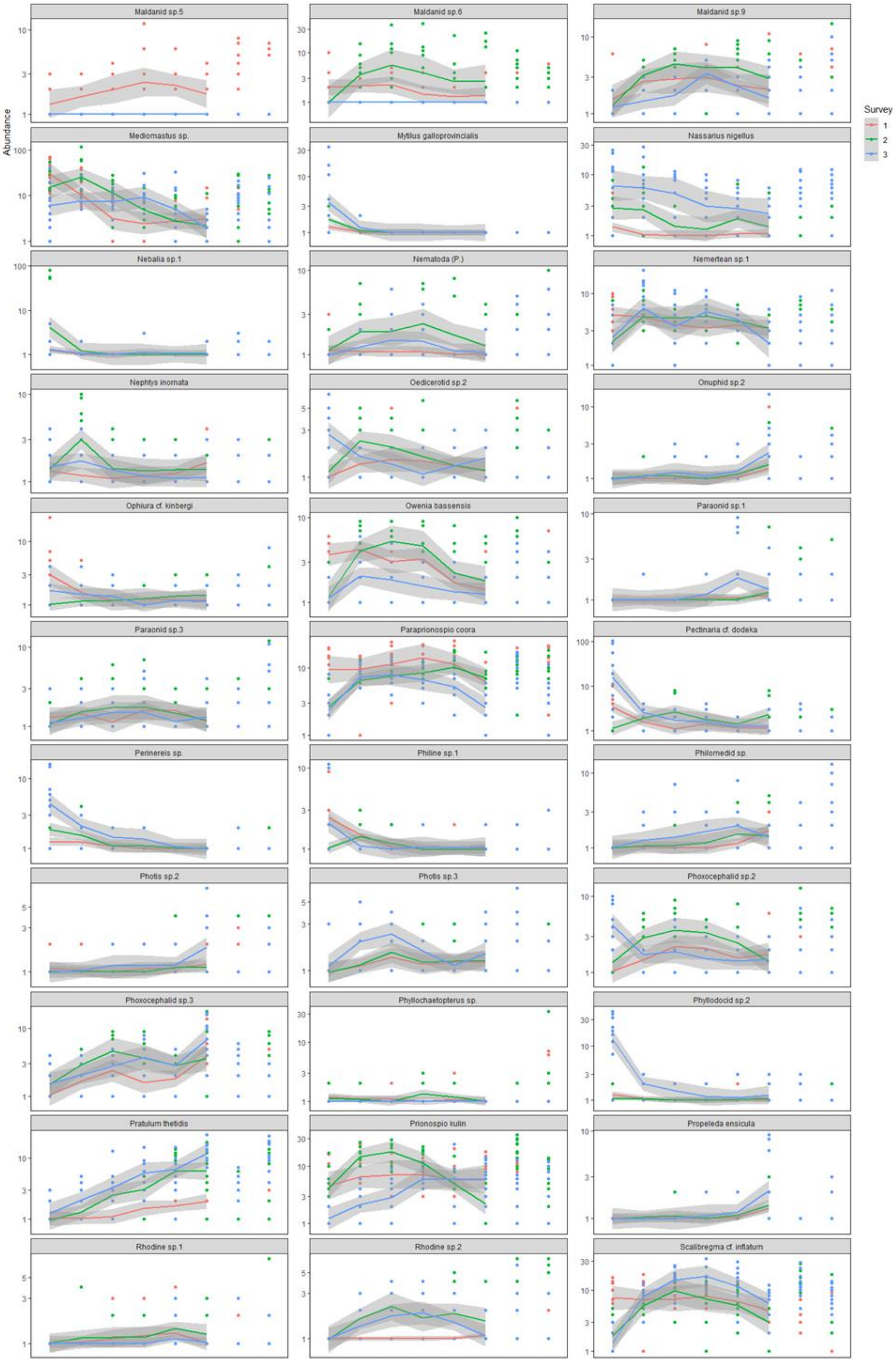
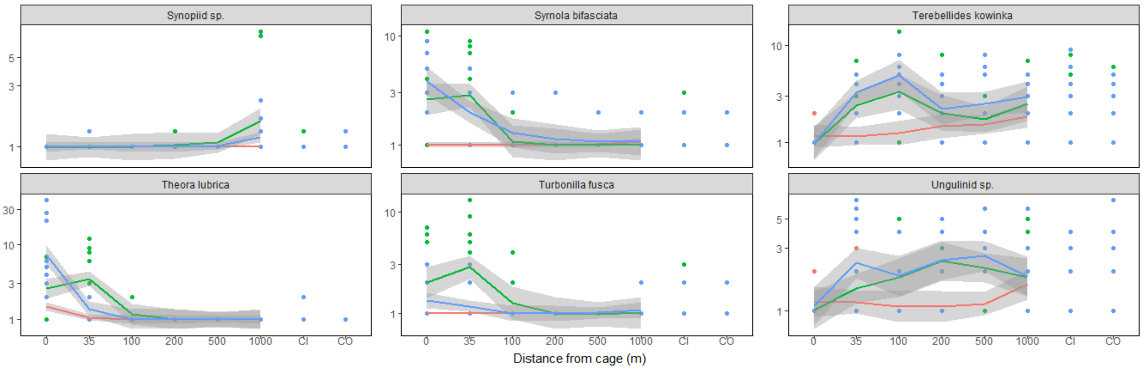
3
5
10
30
100
1000


Maldanid sp.5
Maldanid sp.6
Maldanid sp.9
Mediomastus sp.
Mytilus galloprovincialis
Nassarius nigellus
Nebalia sp.1
Nematoda (P.)
Nemertean sp.1
Nephtys inornata
Oedicerotid sp.2
Onuphid sp.2
Ophiura cf. kinbergi
Owenia bassensis
Paraonid sp.1
Paraonid sp.3
Paraprionospio coora
Pectinaria cf. dodeka
Perinereis sp.
Philine sp.1
Philomedid sp.
Photis sp.2
Photis sp.3
Phoxocephalid sp.2
Phoxocephalid sp.3
Phyllochaetopterus sp.
Phyllodocid sp.2
Pratulum thetidis
Prionospio kulin
Propeleda ensicula
Rhodine sp.1
Rhodine sp.2
Scalibregma cf. inflatum
Abundance
Survey
1
2
3


Synopiid sp.
Syrnola bifasciata
Terebellides kowinka
Theora lubrica
Turbonilla fusca
Ungulinid sp.
1
3
5
10
30
0
35
100
200
500
1000
CI
CO
Distance from cage (m)

5.3 Storm Bay

5.3.1 Results

5.3.1.1 Physico-chemical

Redox

Trumpeter. Redox potential generally increased with distance from the cages. Apart from survey 4, the effect of distance was significant in all surveys (Figure 5.3-1, Redox potential at the 35 m sites was on average lower than at the outer control sites in all three surveys, but this difference was only significant in survey 3 (Table 5.3-1). Redox potential at the 200 m sites (35 m from lease boundary) was similar to the outer control sites in surveys 1 and 4; in surveys 2 and 3 redox potential was lower at 200 m compared to the outer control sites, but this difference was only significant in survey 3. The effect of survey was significant at the 0 m cage site and at 1000 m but not at the other distances (35 m, 500 m and CO; Figure 5.3-2, Table 5.3-2); redox potential was lower at the 0 m cage sites in surveys 1 and 2 compared to surveys 3 and 4.

Storm Bay 1. Redox potential increased with distance from the cages in all 4 surveys. The effect of distance was significant in surveys 2 and 3but not 1 and 4 (Figure 5.3-1,Table 5.3-1). Redox potential at the 35 m sites was significantly lower than at the outer control sites in surveys 1 and 2 but not 3 and 4 (Table 5.3-1). Redox potential at the 200 m transect sites (35 m for lease boundary) was significantly lower than that at the outer control sites in survey 1 but not in the other three surveys. There was a significant effect of survey at the sites closest to the cages (0 and 35 m) but not at the more distant sites (500 m, 1000 m and CO; Figure 5.3-2, Table 5.3-2); redox potential was higher at the 0 and 35 m cage sites in survey 4 compared to the earlier surveys.

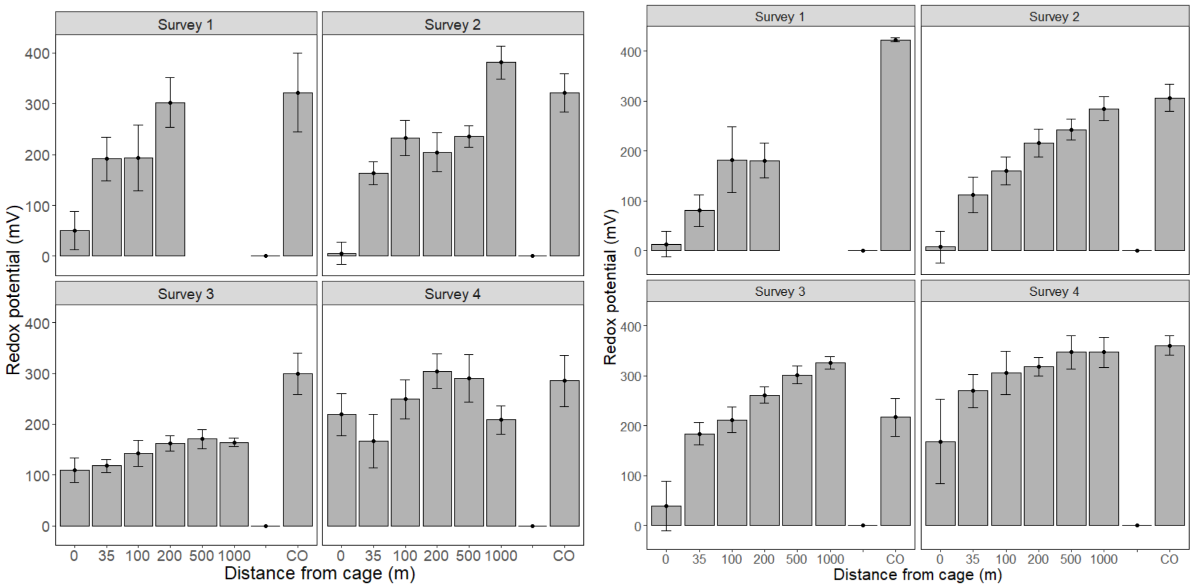


Figure 5.3-1 Average (±SE) redox potential (mV) at 3 cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay (right) leases. Redox was not measured at the inner control sites and 500 and 1000 m were not sampled in survey 1.

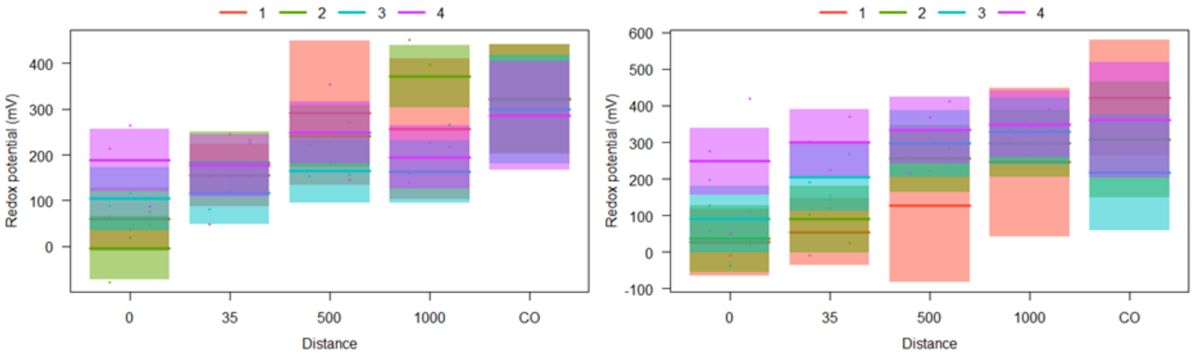


Figure 5.3-2 Redox (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay (right) leases. It is hypothesized that changes at the closer sites were more likely attributed to farming, whereas changes observed at 500 m or greater were likely to reflect variation due to other external drivers.

Sulphides

Trumpeter. Sulphide concentrations (µM) decreased with distance from the cages in all three surveys where it was measured (surveys 2, 3 & 4). The effect of distance was significant in all three of these surveys (Figure 5.3-1, Table 5.3-1). Sulphide at the 35 m sites was higher than at the outer control sites in all three surveys, but this difference was only significant in survey 3 (Table 5.3-1). Sulphide concentrations at the 200 m transect sites (35 m for lease boundary) were also higher than the outer control sites in all three surveys, but the differences were not significant. Sulphide concentrations varied more between surveys at the sites closest to the cages (0 and 35 m) compared to the more distant sites (500 m, 1000 m and CO; Figure 5.3-2, Table 5.3-2) but the effect of survey was only significant at 35 m.

Storm Bay 1. Sulphide concentration (µM) generally decreased with distance from the cages in all three surveys where it was measured (surveys 2, 3 & 4); the effect of distance was significant in all three of these surveys (Figure 5.3-3, Table 5.3-1), but it was clearly more variable in survey 3. Sulphide at the 35 m sites was higher than at the outer control sites in all three surveys, but this difference was only significant in survey 3 (Table 5.3-1). Sulphide concentration at the 200 m transect sites (35 m for lease boundary) was also higher than that at the outer control sites in all three surveys, but the differences were not significant. There was a significant effect of survey at the sites closest to the cages (0 and 35 m) but not at the more distant sites (500 m, 1000 m and CO; Figure 5.3-4, Table 5.3-2); sulphide concentrations were higher at the 0 and 35 m cage sites in surveys 3 and 4 compared with 2.

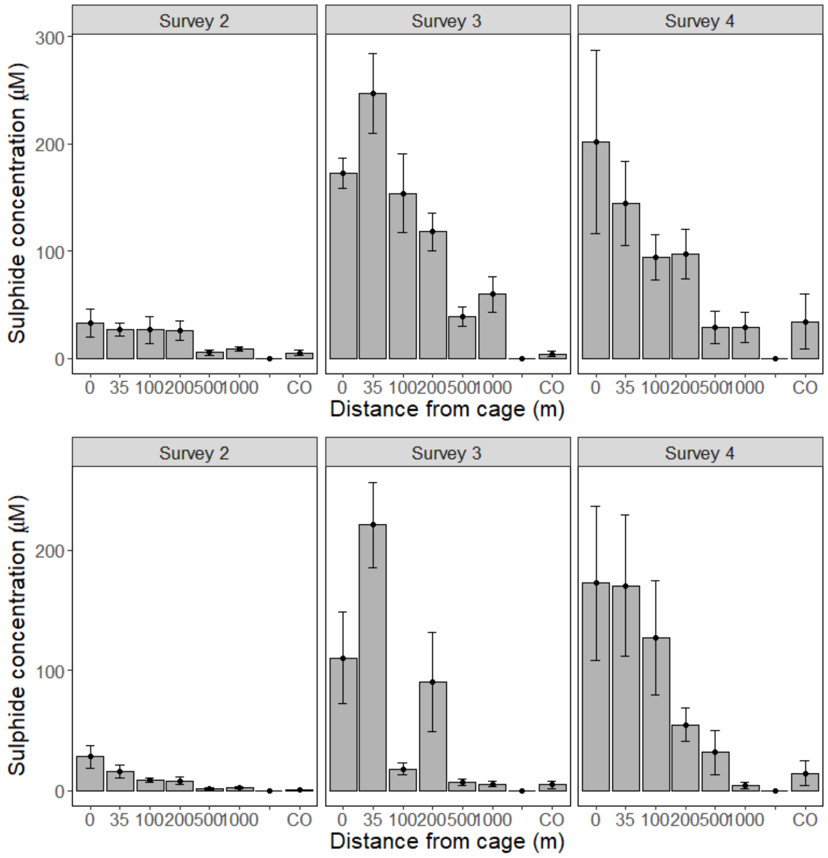


Figure 5.3-3 Average (±SE) sulphide concentration (µM) at 3 cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Trumpeter (top) and Storm Bay (bottom) leases. Sulphide was not measured in survey 1 or at the inner control sites.

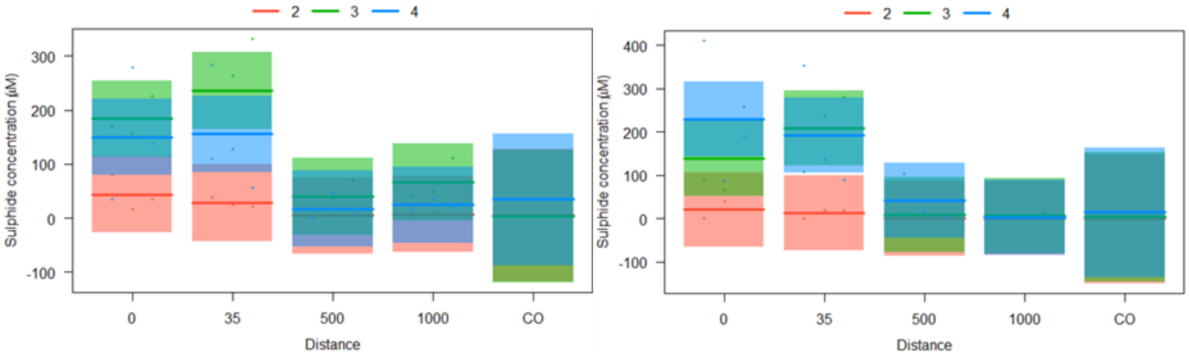


Figure 5.3-4 Sulphide (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay (right) leases. It is hypothesized that changes at the closer sites were more likely attributed to farming, whereas changes observed at 500 m or greater were likely to reflect variation due to other external drivers.

Sediment Particle Size

The cumulative sediment particle sizes appeared broadly similar with distance from the cage for both Trumpeter and Storm Bay leases (Figure 5.3-5). At both leases, sites closer to the cages showed a lower percentage of very fine sands (0.063 mm fraction) but higher percentage of fine sands (0.125 mm fraction) compared to sites more distant from the cages. The other notable difference across sites is the greater percentages of medium sands (0.25 mm size fraction) at the outer controls for both leases and the inner controls for Storm Bay.

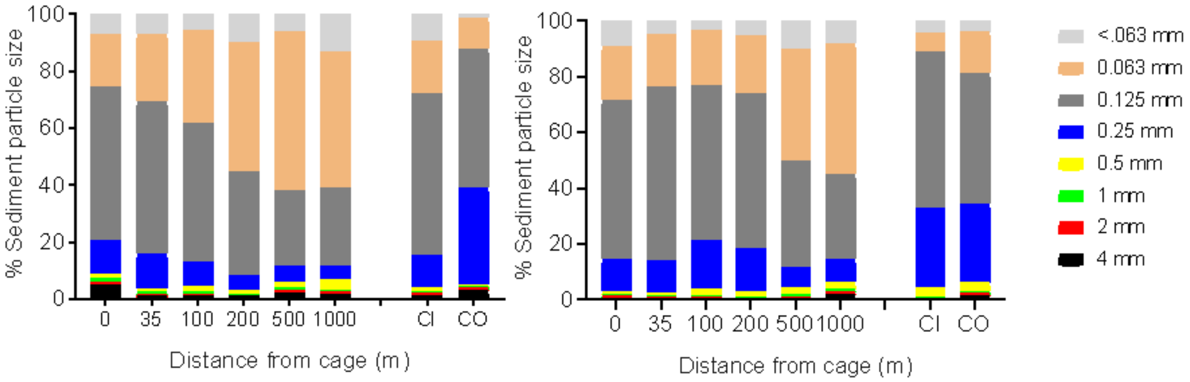


Figure 5.3-5 The cumulative percentage of standard sediment particle sizes at distance from the cages (m) and inner and outer control sites at the Trumpeter (left) and Storm Bay 1 (right) leases.

Organic carbon and nitrogen and stable isotopes

Organic carbon content (%)

Trumpeter. The effect of distance on organic carbon content (%) of the sediments was significant in surveys 1 and 2 but not in surveys 3 and 4 (Figure 5.3-6, Table 5.3-1); in surveys 1 and 2 the organic carbon was notably elevated at the 0 m sites compared to the other distances. Organic carbon at the 35 m sites was generally higher than at the outer control sites in all surveys, but the differences were not significant (Table 5.3-1). Organic carbon content at the 200 m transect sites (35 m for lease boundary) was higher than the outer control sites in all four surveys, but the differences were not significant. There was a significant effect of survey at the 0 m cage site but not at the 35 m or more distant sites (500 m, 1000 m and CO; Figure 5.3-7, Table 5.3-2).

Storm Bay 1. The effect of distance on organic carbon content (%) of the sediments was significant in surveys 1 - 3 but not in survey 4 (Figure 5.3-6, Table 5.3-1). In surveys 1 and 3 the organic carbon was notably elevated at the 0 m sites compared to the other distances. The organic carbon content at the 35 m sites was not significantly different to the outer control sites in each of the surveys (Table 5.3-1). Except for survey 4, organic carbon content at the 200 m transect sites (35 m for lease boundary) was also not significantly different to the outer control sites. In survey 4, the significant effect reflects a very low organic carbon content measured at the outer control sites There was a significant effect of survey at the 0 m cage site but not at the 35 m or more distant sites (500 m, 1000 m and CO; Figure 5.3-7, Table 5.3-2).

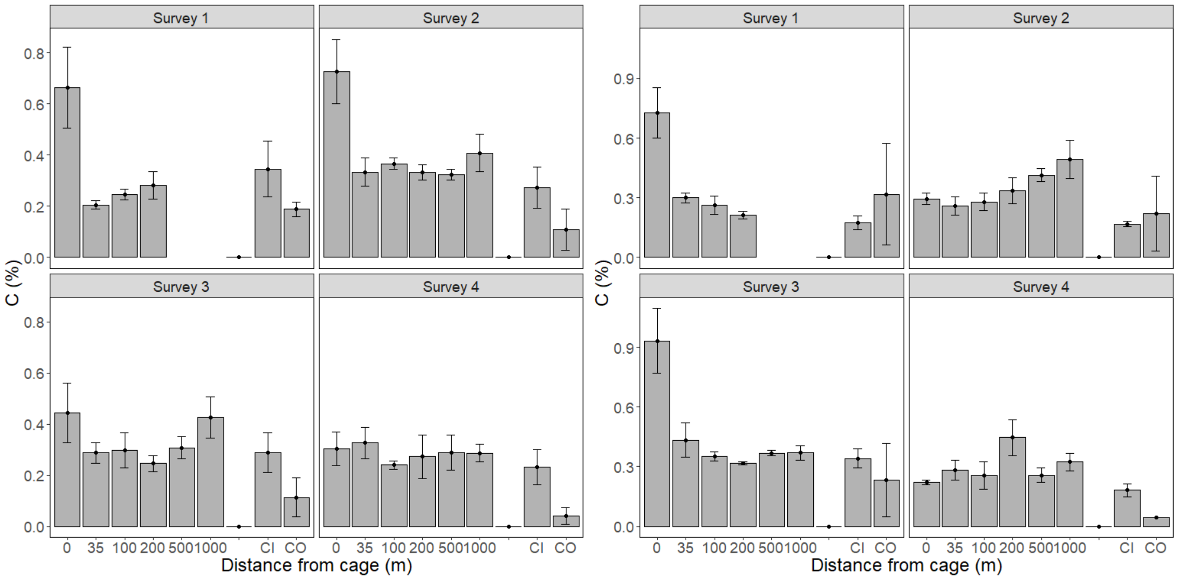


Figure 5.3-6 Average (±SE) of percent carbon (%C) with distances from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

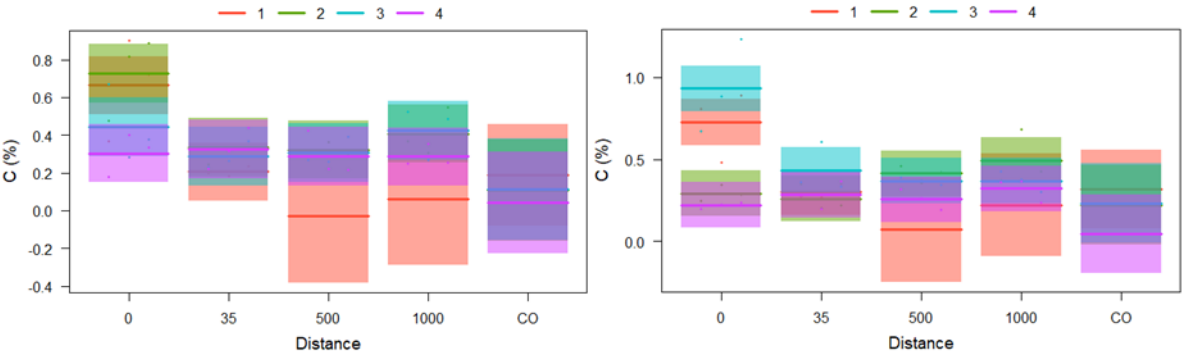


Figure 5.3-7 Percent carbon (%C) (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Organic nitrogen content (%)

Trumpeter. The effect of distance on organic nitrogen content (%) of the sediments was significant in surveys 1, 2 and 4 but not in survey 3 (Figure 5.3-8, Table 5.3-1); in surveys 1 and 2 the organic nitrogen was notably elevated at the 0 m sites compared to the other distances. Organic nitrogen at the 35 m sites was generally higher than at the outer control sites in all surveys, but the differences were not significant (Table 5.3-1). Similarly, organic nitrogen content at the 200 m transect sites (35 m for lease boundary) was higher than the outer control sites in all four surveys, but again the differences were not significant. There was no survey effect on organic nitrogen content at the sites closest to the cages (0 and 35 m) or the more distant sites (500 m, 1000 m and CO; Figure 5.3-9, Table 5.3-2).

Storm Bay 1. The effect of distance on organic nitrogen content (%) of the sediments was significant in surveys 1 and 3 but not in surveys 2 and 4 (Figure 5.3-8, Table 5.3-1). In surveys 1 and 3 the organic nitrogen content was notably elevated at the 0 m sites compared to the other distances, whereas in surveys 2 and 4 organic nitrogen content increased with distance from the cage. Organic nitrogen content at the 35 m and 200 m sites was not significantly different to the outer control sites in any of the four surveys (Table 5.3-1). There was a significant effect of survey at the 0 m cage site but not at the 35 m or more distant sites (500 m, 1000 m and CO; Figure 5.3-9, Table 5.3-2).

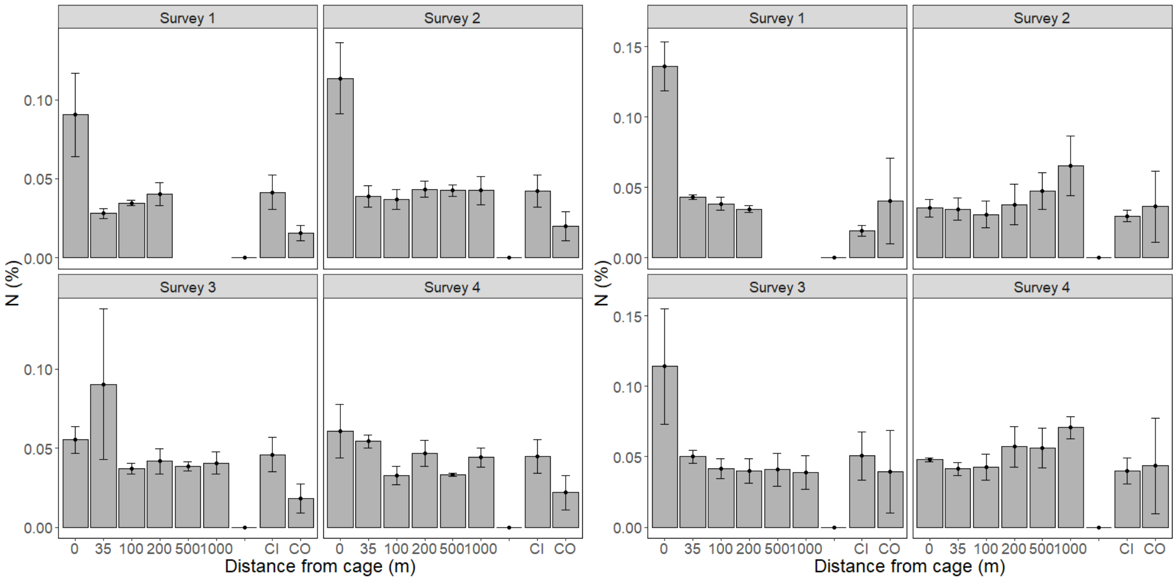


Figure 5.3-8 Average (±SE) of percent nitrogen (%N) with distances from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

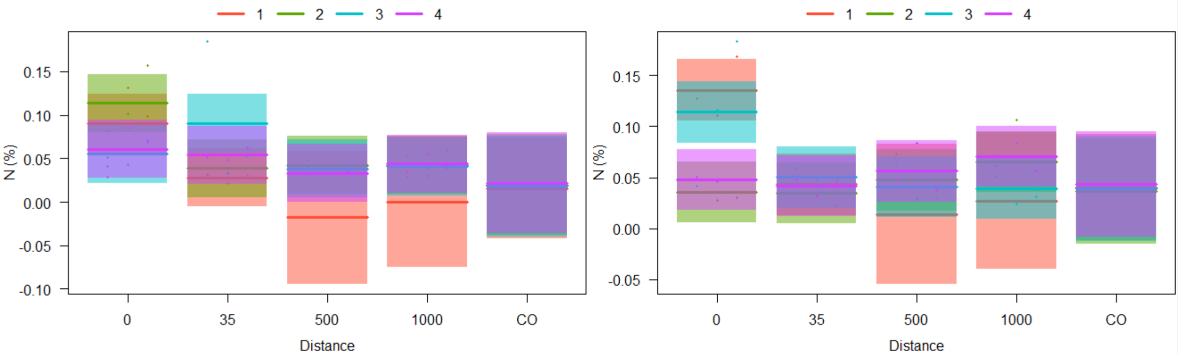


Figure 5.3-9 Percent nitrogen (%C) (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

C:N molar ratio

Trumpeter. There was no clear change with distance for the C:N ratio of the sediments in any of the surveys (Figure 5.3-10, Table 5.3-1). There was also no clear or consistent difference in C:N ratio at the 35 m and 200 m sites compared to the outer control sites across the four surveys; in survey 1 the comparisons with the outer control were significant owing to the high C:N ratios sampled at the outer control (Table 5.3-1). There was no survey effect on the C:N ratio at the sites closest to the cages (0 and 35 m) or the more distant sites (500 m, 1000 m and CO), but again the more distant sites appeared more variable (Figure 5.3-11, Table 5.3-2).

*Storm Bay 1***.** There was no clear change with distance for the C:N ratio of the sediments in any of the surveys (Figure 5.3-10, Table 5.3-1). The C:N ratio was higher at the 35 m and 200 m sites compared to the outer control sites in three of the four surveys, but the differences were only significant in survey 4 (Table 5.3-1). There was no survey effect on the C:N ratio at the sites closest to the cages (0 and 35 m) or the more distant sites (500 m, 1000 m and

CO); however, there was clearly more variation at the more distant sites (Figure 5.3-11, Table 5.3-2).

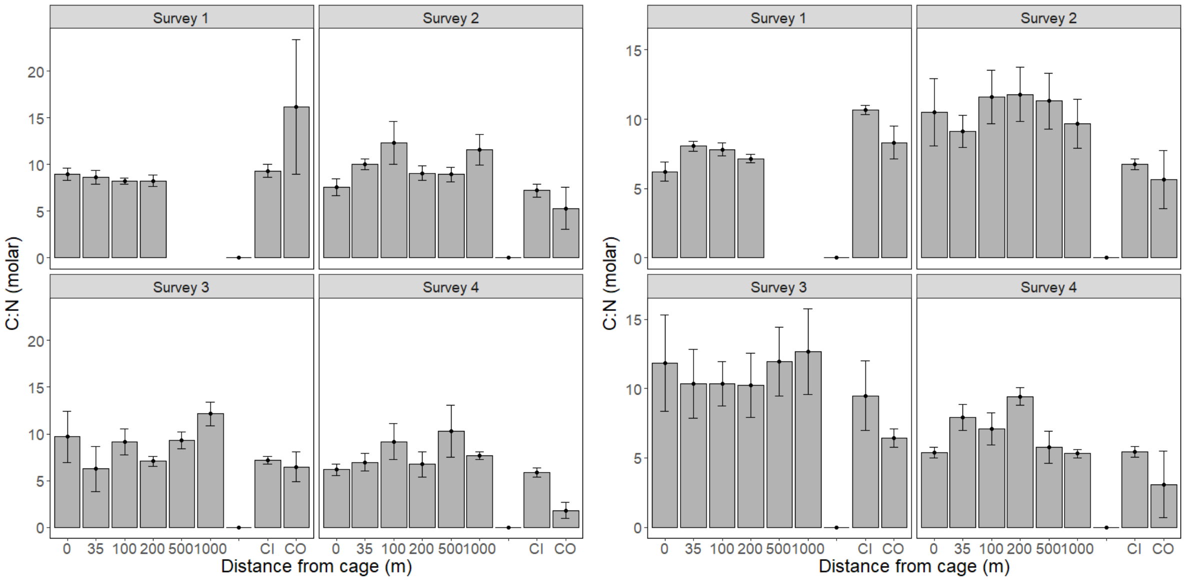


Figure 5.3-10 Average (±SE) C:N molar ratio with distances from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

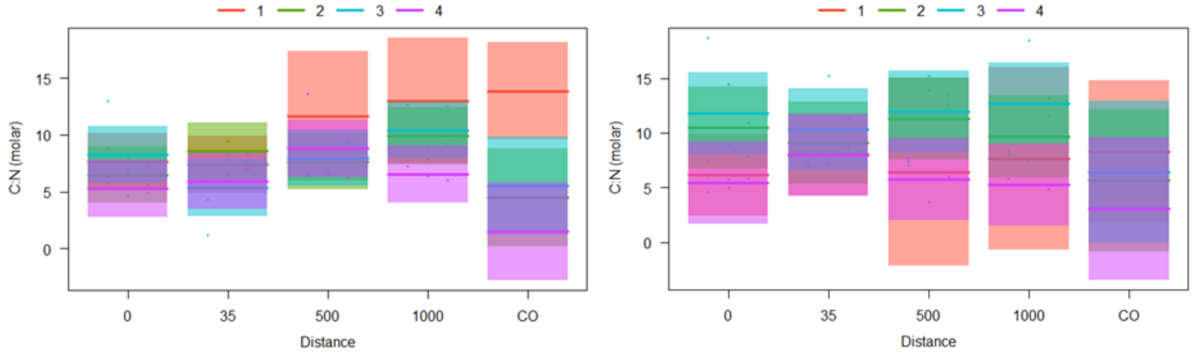


Figure 5.3-11 C:N molar ratio (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Carbon ($\delta^{13}C$)

Trumpeter. The effect of distance on $\delta^{13}C$ of the sediments was significant in surveys 1 and 2 but not in surveys 3 and 4 (Figure 5.3-12, Table 5.3-1); in survey 1, $\delta^{13}C$ was more enriched closer to the cage, while in survey 2 the opposite pattern was observed. Except for survey 4, there was no significant difference in $\delta^{13}C$ between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites. Survey 4 reflected the enriched $\delta^{13}C$ at the outer control sites and interestingly more enriched $\delta^{13}C$ levels were also recorded in survey 4 at the Storm Bay 1 lease at several the sites. There was a significant effect of survey at the 0 m and outer control sites but not at the 35 m, 500 m, and 1000 m sites (Figure 5.3-13, Table 5.3-2).

Storm Bay 1. The effect of distance on $\delta^{13}C$ of the sediments was significant in surveys 2 and 3 but not in surveys 1 and 4 (Figure 5.3-12, Table 5.3-1); in both surveys 2 and 3, $\delta^{13}C$ was more enriched closer to the cage. In survey 1 there was a significant difference in $\delta^{13}C$ between the 200 m and outer control sites. Otherwise, there were no significant differences in $\delta^{13}C$ between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites across all four surveys. There was no significant effect of survey at the 0 and 35 m sites or more distant sites (500 m, 1000 m and CO; Figure 5.3-13, Table 5.3-2).

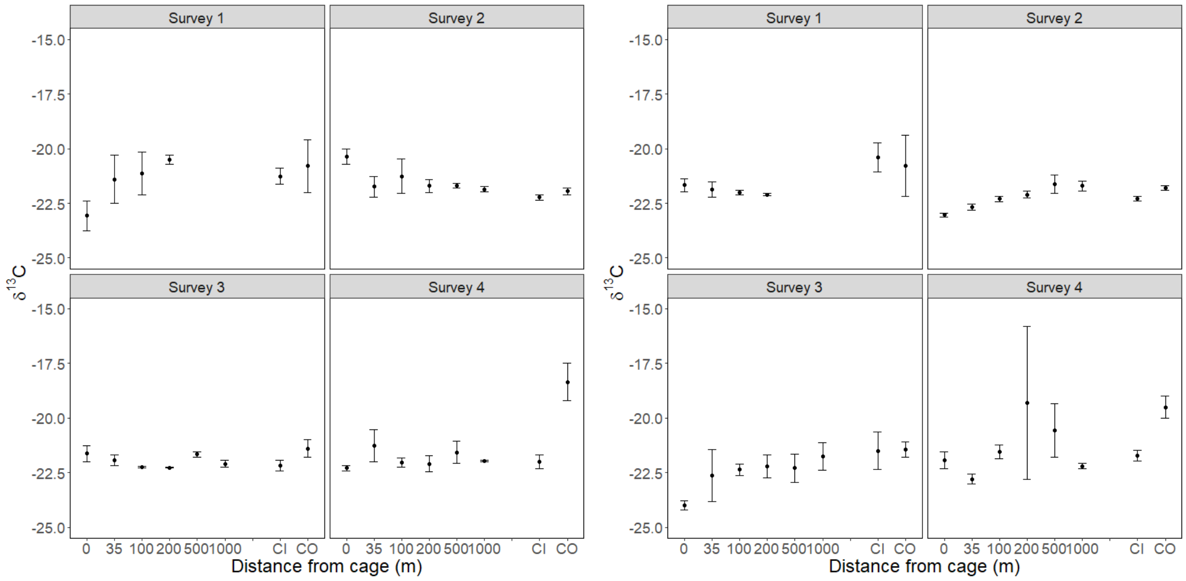


Figure 5.3-12 Carbon ($\delta^{13}C$) stable isotopes values (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

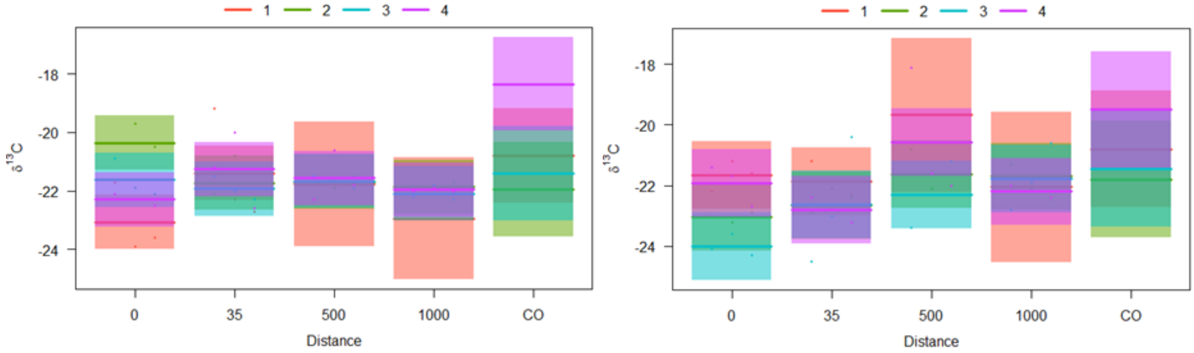


Figure 5.3-13 Carbon ($\delta^{13}C$) stable isotopes value (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Nitrogen ($\delta^{15}N$)

*Trumpeter***.** The effect of distance on $\delta^{15}N$ of the sediments was significant in all surveys (Figure 5.3-14, Table 5.3-1). $\delta^{15}N$ was more depleted closer to the cage in all surveys. $\delta^{15}N$ of sediments at the 35 m sites was more depleted than at the 200 m, inner and outer control sites in all surveys, but the differences were not significant (Table 5.3-1). There was no significant effect of survey at the 0 m, 35 m or more distant sites (500 m, 1000 m and CO; Figure 5.3-15, Table 5.3-2).

Storm Bay 1. There was a gradient of more depleted $\delta^{15}N$ sediment closer to the cage in all of the surveys, but the effect of distance was only significant in surveys 1 and 4 (Figure 5.3-14, Table 5.3-1). $\delta^{15}N$ of sediments at the 35 m sites was typically more depleted than at the 200 m and outer control sites, but only in survey 1 was there a significant difference between the 35 m outer control sites and between the inner and outer control sites (Table 5.3-1). There was no significant effect of survey at the 0 m, 35 m or more distant sites (500 m, 1000 m and CO; Figure 5.3-15, Table 5.3-2).

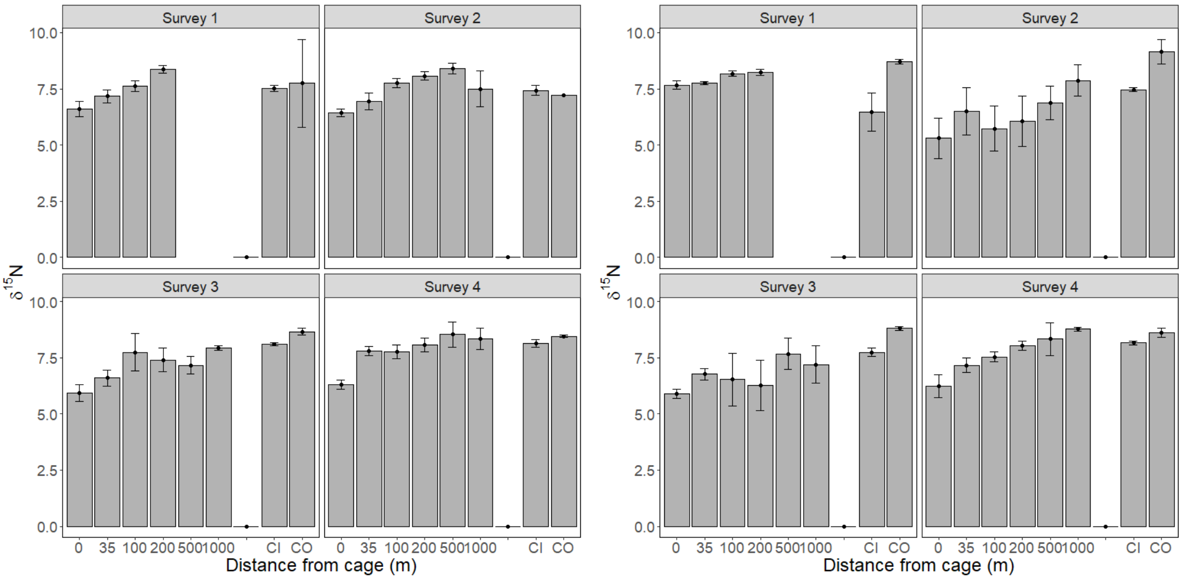


Figure 5.3-14 Nitrogen ($\delta^{15}N$) stable isotopes values (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

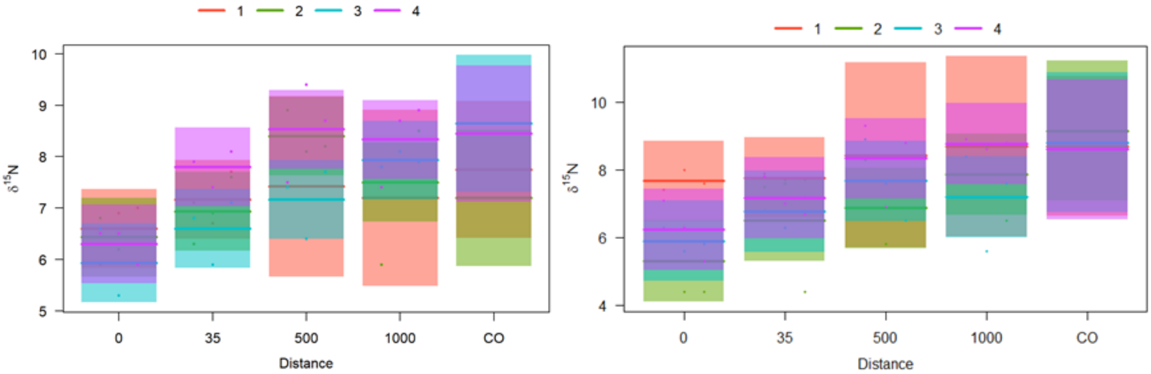


Figure 5.3-15 Nitrogen ($\delta^{15}N$) stable isotope value (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

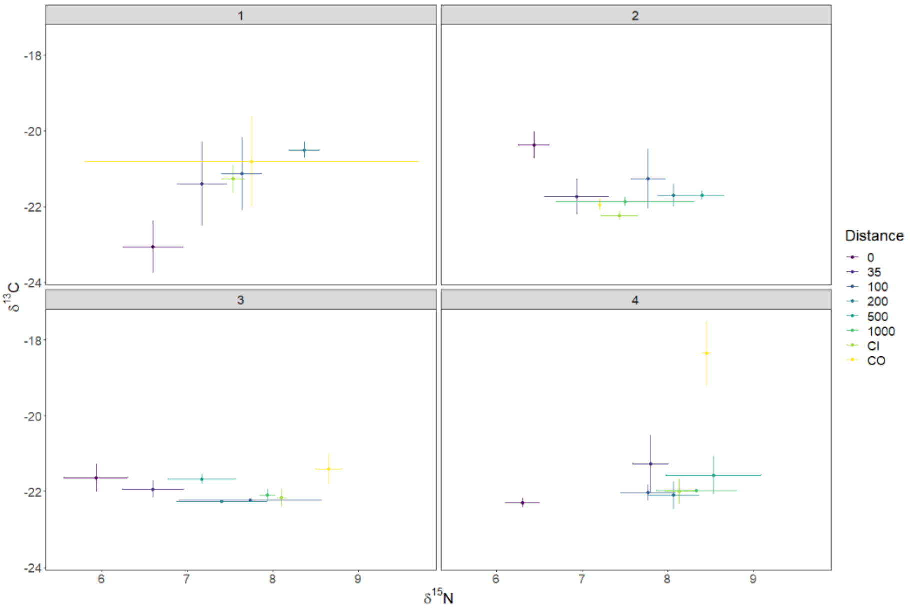


Figure 5.3-16 Distribution of sites/distances based on carbon and nitrogen stable isotopes for each survey at the Trumpeter lease.

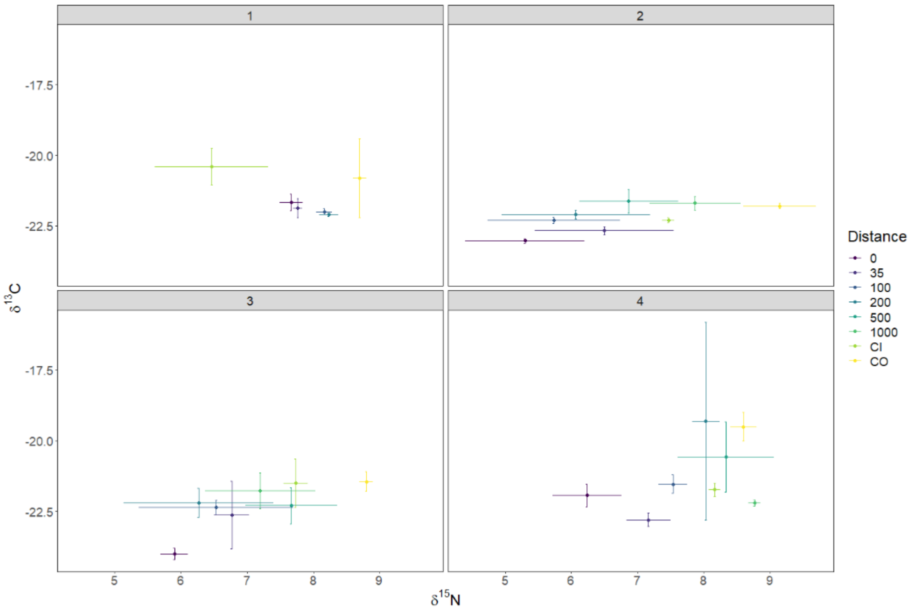


Figure 5.3-17 Distribution of sites/distances based on carbon and nitrogen stable isotopes for each survey at the Storm Bay 1 lease.

Dissolved oxygen (DO)

*Trumpeter***.** There was some evidence of slightly depleted bottom water oxygen levels at the sites closest to the cages, but the effect of distance was only significant in survey 2 (Figure 5.3-18, Table 5.3-1). There was no significant difference in bottom water DO between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in surveys 2 and 4 (Table 5.3-1); in survey 3 limited replication precluded the planned comparisons. There was a significant effect of survey across all sites (0 m, 35 m, 500 m, 1000 m and CO; Figure 5.3-19, Table 5.3-2), with lower bottom water DO in survey 2 compared with surveys 3 and 4.

Storm Bay 1. There was some evidence of slightly depleted bottom water oxygen levels at the sites closest to the cages, but the effect of distance was only significant in survey 2 (Figure 5.3-18, Table 5.3-1). Replication only allowed the planned comparisons to be run in survey 2 and showed significant differences in bottom water DO between the 35 m and outer controls, but not between the 200 m and outer controls or between the inner and outer control sites (Table 5.3-1). There was a significant effect of survey across all sites compared (0 m, 35 m, 500 m, 1000 m; Figure 5.3-19, Table 5.3-2), with bottom water DO being lower in survey 2 than surveys 3 and 4, particularly at the sites closest to the cage (0 and 35 m).

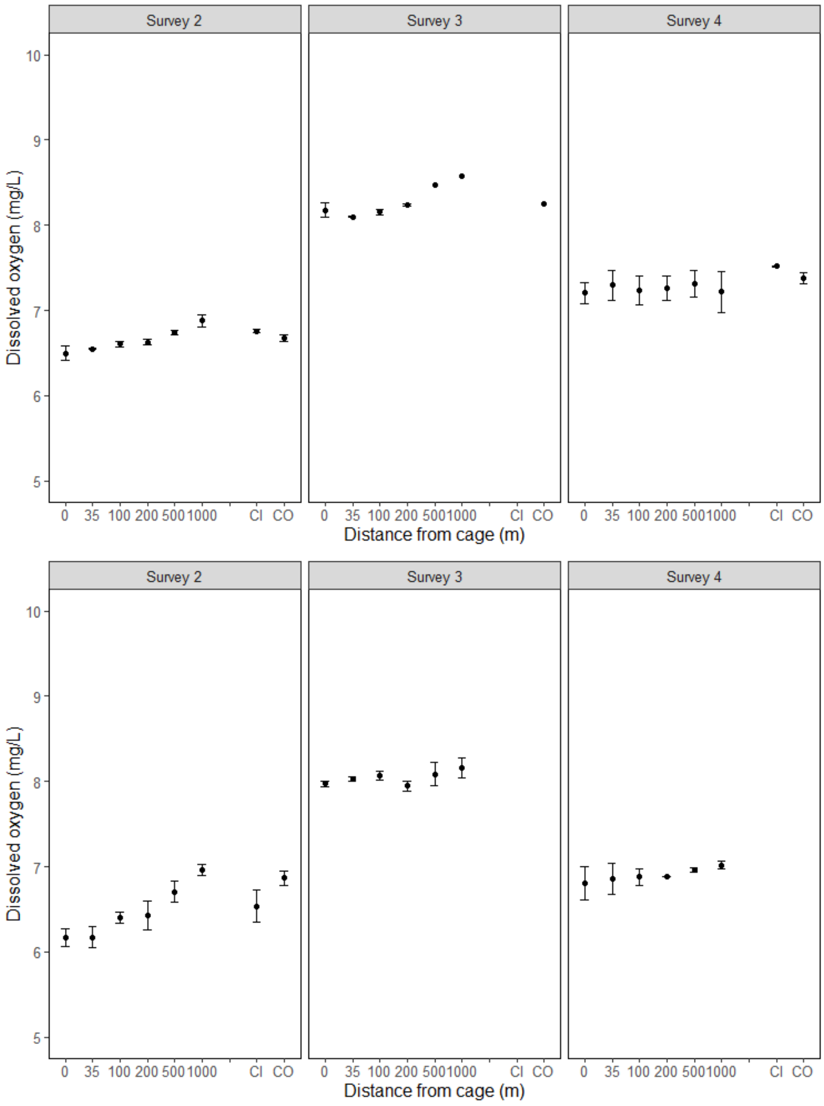


Figure 5.3-18 Bottom water dissolved oxygen concentration (mg/L; average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (top) and Storm Bay 1 (bottom) leases. No data collected for Survey 1 in each case.

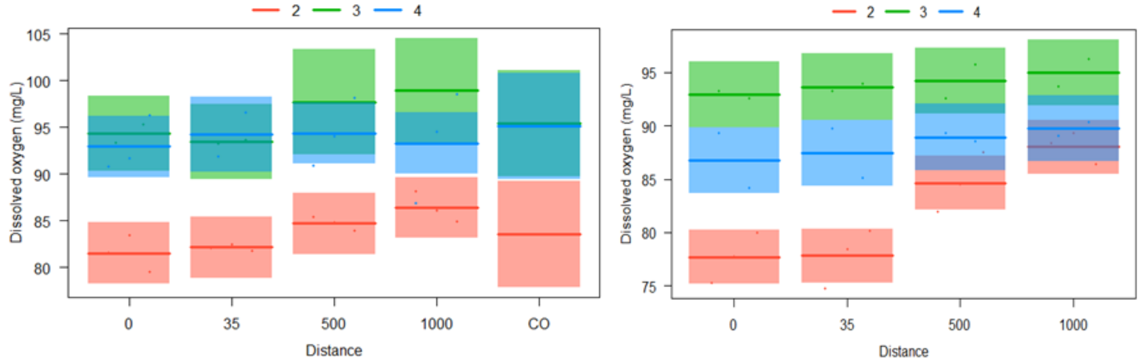


Figure 5.3-19 Bottom water dissolved oxygen concentration (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Table 5.3-1 Results of the generalised linear model testing for the effect of distance for all physico-chemical parameters for each survey and the planned contrasts from the factorial ANOVA comparing the control sites and transect distances. Significant responses ($p < 0.05$) are shown in bold.

			Planned contrasts		
	F ratio	**P**	**CO v 35 m**	**CO v 200 m**	**CO v CI**

Redox					
Trumpeter					
Survey 1	7.035	**0.026**	0.425	0.987	
Survey 2	33.015	**<0.001**	0.227	0.227	
Survey 3	16.838	**0.001**	**<0.001**	**<0.001**	
Survey 4	0.79	0.387	0.452	0.986	
Storm Bay 1					
Survey 1	3.307	0.099	**<0.001**	**0.016**	
Survey 2	34.139	**<0.001**	**0.008**	0.325	
Survey 3	33.438	**<0.001**	0.983	0.783	
Survey 4	2.419	0.139	0.859	0.961	
Sulphide					
Trumpeter					
Survey 2	14.936	**0.001**	0.386	0.669	
Survey 3	9.202	**0.008**	**<0.001**	0.278	
Survey 4	7.226	**0.016**	0.336	0.818	
Storm Bay 1					
Survey 2	11.498	**0.004**	0.477	0.875	
Survey 3	5.142	**0.038**	**0.032**	0.273	
Survey 4	10.253	**0.006**	0.301	0.948	
Organic Carbon (%)					
Trumpeter					
Survey 1	13.277	**0.005**	0.999	0.917	0.818
Survey 2	14.05	**0.002**	0.247	0.249	0.660
Survey 3	0.746	0.401	0.505	0.701	0.653
Survey 4	0.183	0.674	0.058	0.161	0.886
Storm Bay 1					
Survey 1	38.976	**<0.001**	0.999	0.818	0.763
Survey 2	5.167	**0.037**	0.999	0.665	0.970
Survey 3	28.988	**<0.001**	0.475	0.923	0.913
Survey 4	1.796	0.199	0.104	**0.001**	0.674
Organic Nitrogen (%)					
Trumpeter					
Survey 1	9.429	**0.012**	0.953	0.739	0.814
Survey 2	19.434	**<0.001**	0.749	0.614	0.774
Survey 3	0.969	0.340	0.215	0.906	0.915
Survey 4	4.663	**0.046**	0.173	0.373	0.606
Storm Bay 1					
Survey 1	65.305	**<0.001**	0.998	0.983	0.694
Survey 2	1.962	0.180	1.000	1.000	0.994
Survey 3	12.507	**0.003**	0.986	1.000	0.991
Survey 4	2.388	0.142	0.999	0.855	0.998
δ^{13}C					
Trumpeter					
Survey 1	6.552	**0.028**	0.973	0.996	0.993
Survey 2	8.19	**0.011**	0.991	0.986	0.989
Survey 3	1.791	0.199	0.435	0.087	0.302
Survey 4	0.423	0.525	**0.001**	**<0.001**	**0.001**
Storm Bay 1					
Survey 1	2.232	0.159	0.066	**0.017**	0.846
Survey 2	29.423	**<0.001**	0.150	0.862	0.719
Survey 3	8.221	**0.011**	0.739	0.913	1.000
Survey 4	0.35	0.562	0.618	1.000	0.901
δ^{15}N					
Trumpeter					
Survey 1	15.718	**0.003**	0.617	0.576	0.981
Survey 2	10.079	**0.006**	0.980	0.616	0.992
Survey 3	10.158	**0.006**	0.107	0.477	0.949
Survey 4	26.602	**<0.001**	0.740	0.929	0.976
Storm Bay 1					
Survey 1	8.86	**0.014**	**0.002**	0.227	**<0.001**

Survey 2	3.426	0.083	0.394	0.270	0.830
Survey 3	2.205	0.157	0.498	0.315	0.923
Survey 4	29.167	**<0.001**	0.229	0.855	0.958
C:N					
Trumpeter					
Survey 1	1.14	0.311	**<0.001**	**<0.001**	**<0.001**
Survey 2	2.572	0.128	0.209	0.384	0.901
Survey 3	0.333	0.572	1.000	0.997	0.997
Survey 4	1.806	0.198	0.276	0.305	0.627
Storm Bay 1					
Survey 1	3.754	0.081	0.989	0.525	0.133
Survey 2	0.004	0.845	0.727	0.292	0.993
Survey 3	0.022	0.845	0.827	0.838	0.947
Survey 4	0.107	0.883	**0.012**	**<0.001**	0.562
Dissolved Oxygen (mg/L)					
Trumpeter					
Survey 2	18.034	**0.001**	0.665	0.994	0.903
Survey 3	3.976	0.081			
Survey 4	0.092	0.766	0.997	0.987	0.973
Storm Bay 1					
Survey 2	17.392	**0.001**	**0.010**	0.185	0.574
Survey 3	1.932	0.195			
Survey 4	2.997	0.114			
Total Abundance					
Trumpeter					
Survey 1	11.572	**0.007**	1.000	0.993	0.997
Survey 2	0.483	0.497	0.828	0.981	0.994
Survey 3	3.122	0.096	0.989	0.986	0.996
Survey 4	0.074	0.790	0.997	0.731	0.956
Storm Bay 1					
Survey 1	11.964	**0.006**	0.985	1.000	0.997
Survey 2	0.698	0.416	0.996	0.999	0.999
Survey 3	16.563	**0.001**	0.993	0.999	1.000
Survey 4	0.187	0.671	0.981	0.991	1.000
Species Richness					
Trumpeter					
Survey 1	18.046	**0.002**	0.507	0.299	0.390
Survey 2	34.843	**<0.001**	**0.021**	0.864	0.930
Survey 3	1.546	0.232	0.992	0.820	0.533
Survey 4	25.628	**<0.001**	0.883	0.914	0.928
Storm Bay 1					
Survey 1	10.907	**0.008**	0.642	0.663	0.145
Survey 2	16.537	**0.001**	0.999	0.999	0.910
Survey 3	7.991	**0.012**	1.000	0.762	0.628
Survey 4	16.287	**<0.001**	0.591	0.758	0.914
Species Diversity					
Trumpeter					
Survey 1	516.965	**<0.001**	**0.014**	0.477	0.762
Survey 2	52.032	**<0.001**	**0.013**	0.060	0.677
Survey 3	15.968	**0.001**	0.320	0.994	0.958
Survey 4	43.839	**<0.001**	0.813	0.998	0.981
Storm Bay 1					
Survey 1	127.219	**<0.001**	0.758	0.983	0.228
Survey 2	32.65	**<0.001**	0.842	0.936	0.939
Survey 3	44.428	**<0.001**	0.698	0.329	0.345
Survey 4	25.291	**<0.001**	0.359	0.824	0.984
ROV Score					
Trumpeter					
Survey 2	28.275	**<0.001**	**0.024**	1.000	0.611
Survey 3	82.378	**<0.001**	0.430	0.685	0.592
Survey 4	19.826	**<0.001**	0.477	0.842	0.279

Storm Bay 1					
Survey 2	108.05	**<0.001**	0.916	0.988	0.654
Survey 3	109.92	**<0.001**	0.070	0.950	0.769
Survey 4	1.260	0.235			

Table 5.3-2 Results of the generalised linear model testing for the effect of survey at the two distances closest to the cages (0 and 35/50 m) and at the three distances furthest from the cages (500, 1000 and CO). Significant responses ($p < 0.05$) are shown in bold.

	0 m	35 m	500 m	1000 m	CO
Trumpeter					
Redox	**0.001**	0.644	0.448	**<0.001**	0.907
Sulphide	0.067	**0.001**	1.000	1.000	1.000
Organic Carbon (%)	**<0.001**	1.000	1.000	1.000	1.000
Organic Nitrogen (%)	0.126	0.336	1.000	1.000	1.000
$\delta^{13}C$	**0.001**	1.000	1.000	1.000	**<0.001**
$\delta^{15}N$	0.838	0.372	0.149	0.839	0.839
C:N	0.848	0.793	0.908	0.440	0.908
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.004**
Total Abundance	1.000	0.815	0.681	1.000	1.000
Species Richness	0.184	**0.014**	**<0.001**	**<0.001**	0.184
Species Diversity	**0.002**	0.165	0.171	**0.020**	0.616
ROV Score	**<0.001**	0.554	1.000	1.000	1.000
Storm Bay 1					
Redox	**0.031**	**0.041**	1.000	1.000	1.000
Sulphide	**0.014**	**0.011**	1.000	1.000	1.000
Organic Carbon (%)	**<0.001**	0.551	0.551	0.551	0.551
Organic Nitrogen (%)	**<0.001**	1.000	1.000	1.000	1.000
$\delta^{13}C$	0.253	1.000	0.474	1.000	0.719
$\delta^{15}N$	1.000	1.000	1.000	1.000	1.000
C:N	0.261	1.000	0.261	0.192	1.000
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	**0.003**	
Total Abundance	**0.013**	1.000	1.000	1.000	1.000
Species Richness	0.232	0.754	**0.032**	0.083	0.754
Species Diversity	**<0.001**	**0.022**	0.488	1.000	1.000
ROV Score	**<0.001**	0.116	0.726	**<0.001**	0.726

5.3.1.2 Macrofauna

Across the four surveys, 33,997 and 42,156 invertebrates were collected from a total of 258 grab samples at Trumpeter and Storm Bay 1 leases respectively, comprising 339 different taxa at Trumpeter and 335 different taxa at Storm Bay 1. Overall, polychaetes were the most abundant taxon at the Trumpeter lease, making up 40% of the total taxa abundance, followed by crustaceans (34%), molluscs (13%), cnidarians (10%), echinoderms (2%), nemerteans (1%) and others (< 1%). At Storm Bay 1 polychaetes made up a bigger proportion of the community (52%) followed by crustaceans (25%), molluscs (10%), cnidarians (9%), echinoderms (2%), nemerteans (1%) and others (< 1%).

At Trumpeter, polychaetes and crustaceans were present in 100% of the samples, molluscs in 98%, cnidarians in 74%, echinoderms in 70% and nemerteans in 63% of the samples. At Storm Bay 1, polychaetes and crustaceans were present in 100% of the samples, molluscs in 99%, cnidarians in 73%, echinoderms in 64% and nemerteans in 72% of the samples.

At Trumpeter, 38% of the taxa were crustaceans, 28% polychaetes, 26% molluscs, 3% echinoderms and 5% other. Storm Bay 1 was similar, with 38% of the taxa being crustaceans, 29% polychaetes, 25% molluscs, 4% echinoderms and 4% other. The most abundant taxon at Trumpeter over the four surveys was *Capitella* sp., making up 18% of the total abundance, followed by the anemone *Edwardsia* sp. (9.7%), amphipod *Ampelisca cf. australis* (5.5%), pectinarid polychaete *Pectinaria antipoda* (4.5%), gastropod *Nassarius nigellus* (3.9%), amphipods *Tethygeneia* sp. (3.2%), *Isaeid* sp.1 (2.3%) and *Tipimegus cf. thalerus* (1.8%), spionid polychaetes *Spionid* sp.4 (3%) and *Paraprionospio coora* (1.8%), bivalve *Placamen* *placidum* (1.6%) and other taxa (1.5% or less). All these taxa remained reasonably abundant across all four Trumpeter surveys.

At Storm Bay 1 the most abundant taxa over the four surveys were *Capitella* sp., making up 32% of the total abundance, followed by the anemone *Edwardsia* sp. (9.2%), pectinarid polychaete *Pectinaria antipoda* (3.2%), spionid polychaete *Prionospio kulin* (3.2%), tanaid *Apseudid* sp.1 (2.8%), gastropod *Nassarius nigellus* (2.7%), amphipods *Aorid* sp.1 (2.6%) and *Ampelisca cf. australis* (2.4%), polychaete *Paraonis* sp.1 (2.4%) and other taxa (1.4% or less). Again, all these taxa remained reasonably abundant across all four Storm Bay 1 surveys.

Total abundance

Trumpeter. Total abundance was often elevated at the cage site compared to the other transect sites, but the effect of distance was only significant in survey 1 (Figure 5.3-20, Table 5.3-1). There was no significant difference in total abundance between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites for any of the surveys (Table 5.3-1). There was no significant effect of survey at the 0 and 35 m sites or more distant sites (500 m, 100 m and CO; Figure 5.3-21, Table 5.3-2).

*Storm Bay 1***.** Total abundance was often elevated at the cage site compared to the other transect distances, but the effect of distance was only significant in surveys 1 and 3 (Figure 5.3-20, Table 5.3-1); in the other surveys abundance was more variable across sites and the pattern with distance was not as evident. There was no significant difference in total abundance between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in any of the surveys (Table 5.3-1). There was a significant effect of survey at the 0 m cage site but not at the other distances (35 m, 500 m, 100 m and CO; Figure 5.3-21, Table 5.3-2); in surveys 2 and 4 the average abundance at the 0 m sites was <200 ind. m^{-2} compared with >400 ind. m^{-2} in surveys 1 and 3.

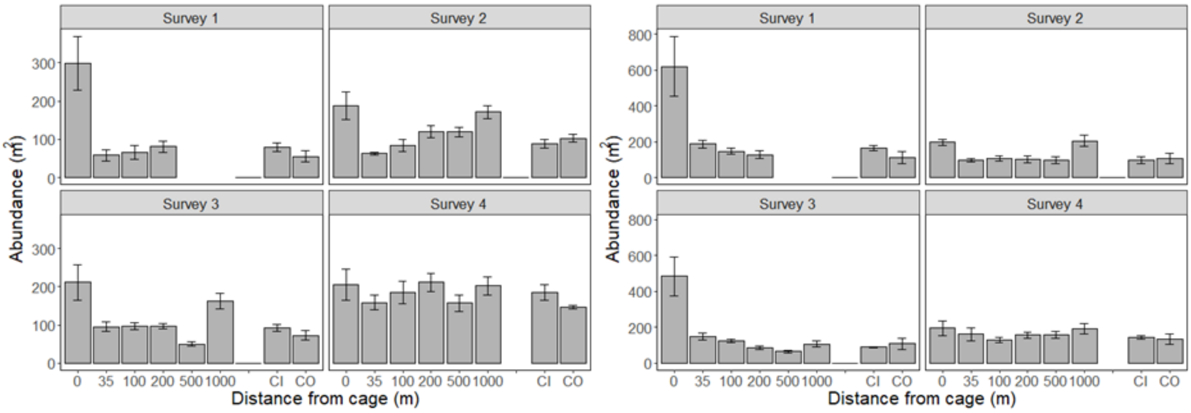


Figure 5.3-20 Total abundance (m^{-2}; average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases. Data not available for 500 m and 1000 m locations in Survey 1.

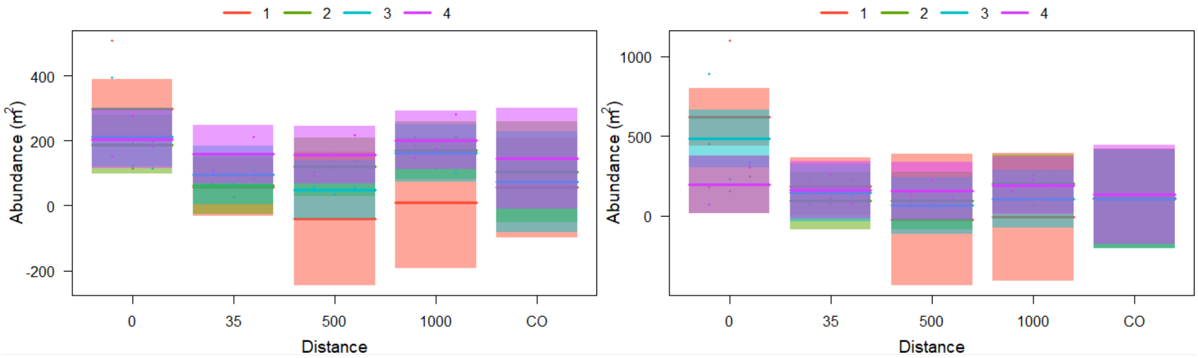


Figure 5.3-21 Total abundance (m^{-2}; predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Species richness

Trumpeter. Species richness decreased closer to the cage on the transects in all four surveys, and except for survey 3, the effect of distance was significant with levels being generally lower nearest to the cages than elsewhere on the transects or than at control locations (Figure 5.3-22, Table 5.3-1). There was no significant difference in species richness between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys, except between 35 m and the outers control sites in survey 2 (Table 5.3-2). There was an increase in species richness across surveys at all distances. The effect of survey was significant at the 35 m, 500 m and 1000 m sites but not at the sites closest (0 m) and most distant from the cage (CO) (Figure 5.3-23, Table 5.3-2).

Storm Bay 1. Species richness decreased closer to the cage on the transects in all four surveys and the effect of distance was significant in all surveys (Figure 5.3-22, Table 5.3-1). There was no significant difference in species richness between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in each of the surveys (Table 5.3-2). The effect of survey was significant at the 500 m site but not at the other distances (0 m, 35 m, 1000 m & CO) (Figure 5.3-23, Table 5.3-2).

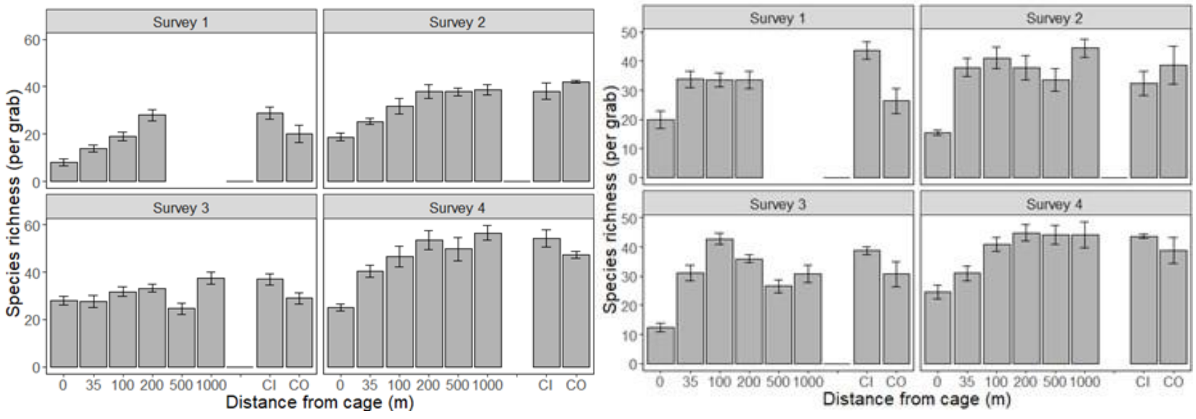


Figure 5.3-22 Species richness (per grab; average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases.

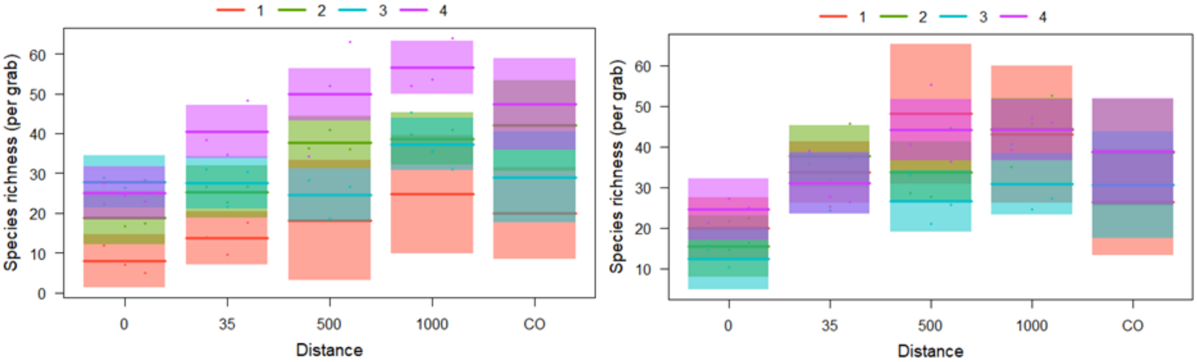


Figure 5.3-23 Species richness (per grab; predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Species diversity

Trumpeter. Species diversity index (H’) was lower closer to the cage (particularly at the 0 m cage site) compared to the other transect distances in all four surveys (Figure 5.3-24), with the effect of distance significant in all four surveys (Table 5.3-1). In surveys 1 and 2, species diversity was significantly lower at the 35 m compared to the outer control sites. All other planned comparisons across the 4 surveys were not significantly different (Table 5.3-1). Species diversity was typically lower across all sites in survey 1 compared to the other surveys; however, the effect of survey was only significant at 0 m and 1000 m (Figure 5.3-25, Table 5.3-2).

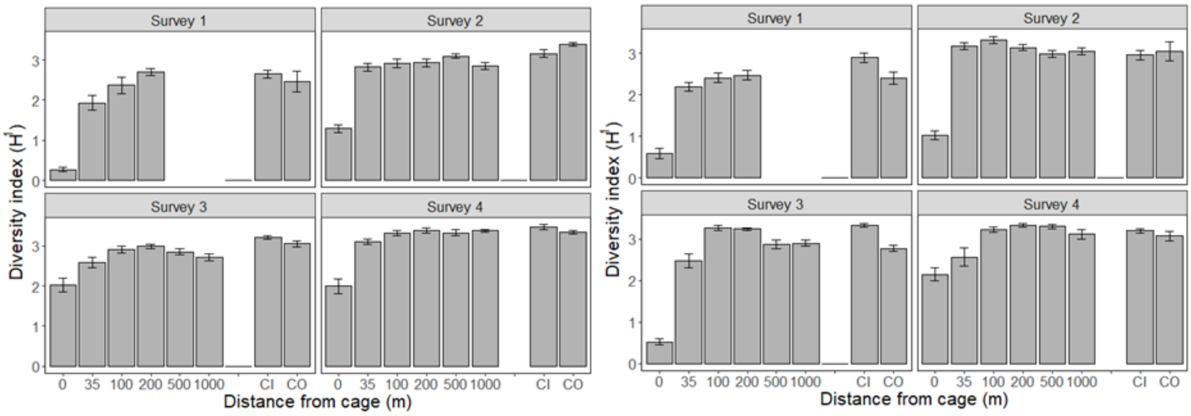


Figure 5.3-24 Species diversity index (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases. Data not available for 500 m and 1000 m locations in Survey 1.

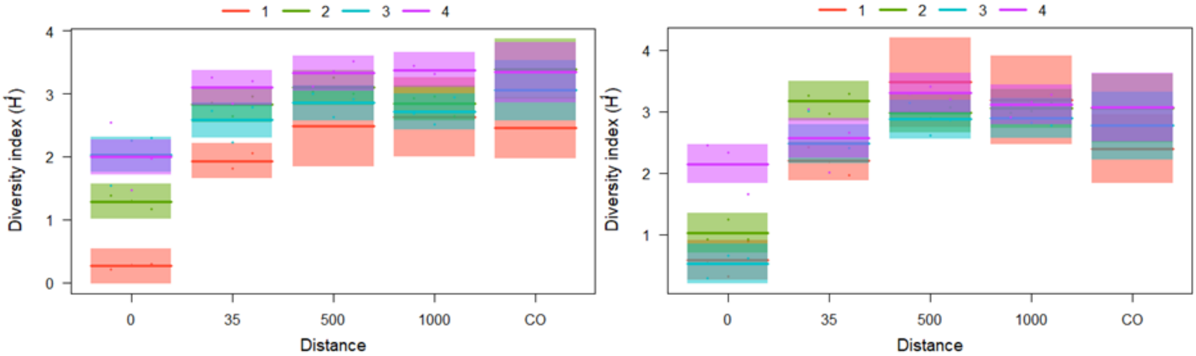


Figure 5.3-25 Species diversity index (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases.

Storm Bay 1. Species diversity index (H') was again lower closer to the cage (particularly at the 0 m cage site) compared to the other transect distances in all four surveys (Figure 5.3-24), with distance exerting a significant effect across all four surveys (Table 5.3-1). Although species diversity was lower at the 35 m site compared to the outer control sites in 3 of the 4 surveys, the differences were not significant. Other planned comparisons across the 4 surveys were not significant (Table 5.3-1). There was a significant survey effect at the 0 and 35 m sites but not at the more distant sites (500 m, 1000 m and CO; Figure 5.3-25, Table 5.3-2).

Multivariate analysis

The multivariate GLM showed that faunal assemblages changed with distance from the cages in all four surveys at both Trumpeter and Storm Bay 1. For the species that showed a significant effect of distance, the fourth corner coefficients were plotted using a LASSO penalty for each survey (see Appendix A for plots of abundance by distance for individual species).

Trumpeter. Of the more common species in survey 1 the polychaetes *Capitella* sp. and *Onuphid* sp.1, showed the strongest positive relationship with proximity to the cage, and the anemone *Edwardsia* sp. and isopod *Anthurid* sp.4 showed a positive relationship with distance from the cage (Figure 5.3-26). In survey 2, the polychaete *Capitella* sp. was again the most common species and showed a positive relationship with proximity to the cages (Figure 5.3-27). Other species to show positive relationships with proximity to the cages included the sea slug *Pleurobranchaea maculata*, the nereid polychaete *Perinereis* sp., tanaid *Leptochelid* sp.1 and the crab *Halicarcinus rostratus*. Species to show negative relationships with proximity to the cages included the amphipods *Bybilus mildura, Aorid* sp.1, *Ampelisca cf. australis* and *Isaeid* sp.3, the isopod *Anthurid* sp.2, the fan worm *Euchone varibilis*, spionid polychaetes *Prionospio kulin* and *Spionid* sp.4, the scalibregmatid polychaete *Scalibregma inflatum*, and polychaetes *Arabella* sp. and *Ampharetid* sp.1.

In survey 3, the species showing a positive relationship with proximity to cage included the polychaetes *Capitella* sp., *Pectinaria antipoda, Mediomasutus* sp., *Glycera tridactyla, Dipolydora gairdi,* gastropods *Nassarius nigellus, Anachis atkinsoni, Syrnola bifasciata* and *Turbonilla fusca,* the nemertean sp.1 and the amphipod *Tethygeneia* sp. Species that showed a negative relationship with proximity to the cages (and/or positive relationship with distance from the cage) included the polychaetes *Arabella* sp. and *Ampharetid* sp.1. *Euchone varibilis*, amphipods *Tipimegus thalerus*, *Bybilus mildura* and *Ampelisca cf. australis* and *Isaeid* sp.3, the isopod *Anthurid* sp.3-4, bivalves, *Placamen placidum* and *Callista diemenensis* and the anemone *Edwardsia* sp.

In survey 4 there was a large increase from survey 3 in the number of species (27 → 47) that showed a significant relationship with distance (Figure 5.3-28). The polychaete *Capitella* sp. again topped the list of species showing a positive relationship, but other species included the bivalves *Mytilus galloprovincialis*, *Fulvia tenuicostata* and *Raeta pulchella*, gastropods *Nassarius nigellus* and *Pyramidellid* sp.1, nemertean sp.1, amphipod *Isaeid* sp.3, and polychaetes *Perinereis* sp., and *Pectinaria antipoda.* Similarly, there was an increase in the number of species identified to have negative relationships with proximity to the cages; these species included amphipods *Tipimegus thalerus*, *Phoxocephalid* sp.3, *Bybilus mildura* and *Ampelisca cf. australis*, the isopod *Anthurid* sp.1-4, bivalves *Pratulum thetidis*, *Placamen placidum, Myadora sp.1* and *Callista diemenensis*, and polychaetes *Arabella* sp. and *Ampharetid* sp.1.

The unconstrained ordination from the LVM showed a separation of samples based on their species composition and relationship with distance from the cage (i.e. along latent variable 1;

Figure 5.3-32). In survey 1, species composition at 0 m was distinct from the assemblages found at the other distances. The 35 m and 200 m assemblages were also distinct from each other with the 100 m assemblage overlapping both distances. Of the ten most important species identified in the LVM, the polychaete *Capitella* sp. was characteristic of the 0 m sites (~4280 ind. m^{-2}), but it was also reasonably common out to 200 m (~380, 250 and 20 ind. m^{-2} at 35, 100 and 200 m sites respectively). The polychaetes *Onuphid* sp.1 and *Dipolydora gairdi* and gastropod *Turbonilla fusca* were most common at 35 m and distances further from the cages; *Onuphid* sp.1 and *Turbonilla fusca* were most common at 35 m and *Dipolydora gairdi* at 100 m. The bivalve *Placamen placidum*, the anemone *Edwardsia sp.*, amphipods *Isaeid* sp.1 *and Tipimegus cf. thalerus,* isopods *Anthurid sp.4* and *Nemertean sp.1* were identified as most characteristic of sites more distant from the cages; none of these species were present in the 0 m samples. *Tipimegus cf. thalerus* was only found at 100 and 200 m sites and *Anthurid* sp.4 at 200 m only.

In survey 2 there was clear separation in species composition according to distance, with the assemblage at 0, 35, 100 and 200 m becoming more distinct from each other and the 500 and 1000 m sites which overlapped each other (Figure 5.3-32). The polychaete *Capitella* sp. was again characteristic of the community at 0 m, but in much lower abundances compared to survey 1 and only found at 0 m (~1900 ind. m^{-2}) and 35 m (~10 ind. m^{-2}). Other species characteristic of sites closer to the cages included the sea slug *Pleurobranchaea maculata*, the tanaid *Leptocheliid* sp.1, the crab *Halicarcinus rostratus* and the polychaete *Perinereis* sp. The polychaetes *Mediomastus* sp. and *Dipolydora gairdi* were not found at the 0 m sites but were most common between 35 and 200 m. The amphipod *Isaeid* sp.3, isopod *Anthurid* sp.4 and polychaetes *Euchone variabilis* and *Ampharetid* were characteristic of the more distant sites from the cages.

The separation in species composition according to distance was again evident in survey 3, but with greater overlap in community composition across the 100-1000 m sites relative to survey 2 (Figure 5.3-32). *Capitella* sp. was again characteristic of the community at 0 m, but in lower abundance (~1300 ind. m^{-2}) compared to surveys 1 (~4280 ind. m^{-2}) and 2 (~1900 ind. m^{-2}). The amphipod *Tethygeneia* sp., gastropod *Syrnola bifasciata*, polychaete *Glycera tridactyla* and *Nemertean* sp.1 were all characteristic of the assemblages at sites closer to the cages. The polychaete *Dipolydora gairdi* was most abundant at the 35 and 100 m sites. The amphipods *Isaeid* sp.3 and *Aorid* sp.1 and the polychaete *Scalibregma cf. inflatum* were characteristic of the more distant sites from the cages.

The separation in assemblages according to distance was more evident in survey 4 compared to the previous surveys (Figure 5.3-32). *Capitella* sp., *Perinereis* sp. and *Mytilus galloprovincialis* were characteristic of the 0 m sites. The gastropod *Pyramidellid* sp.1, cumacean *Gynodiatylid* sp.1 and polychaete *Oweniid* sp. were characteristic of the intermediate distances with *Pyramidellid* most abundant at 35 and 100 m, *Oweniid* sp. at 100 m, and *Gynodiatylid* sp.1 at 100 and 200 m. Species characteristic of the more distant sites from the cages were the polychaete *Ampharetid* sp.1, cumacean *Litogynodiastylis* sp., amphipod *Isaeid* sp.3 and bivalve *Myadora* sp.1.

Storm Bay 1. Of the more common species in survey 1 the polychaetes *Capitella* sp. and *Onuphid* sp.1, and amphipod *Jassa* sp.1 showed the strongest positive relationship with proximity to the cage, but *Onuphid* sp.1 was more common at 35 m rather than 0 m (Figure 5.3-30). The species that showed a negative relationship with proximity to the cages were the polychaete *Ampharetid* sp.1, isopod *Anthurid* sp.4 and tanaid *Apseudid* sp.1, with all three species most common at the 100 and 200 m sites.

In survey 2, the polychaete *Capitella* sp. was again the most common species and showed a positive relationship with proximity to the cages. Other species to show positive relationships with proximity to the cages included the amphipod *Jassa* sp.1, the caprellid *Caprella penatis,* the heart urchin *Echinocardium cordatum,* the polychaete *Perinereis* sp. and Nemertean sp.1. Aside from *Caprella penatis* and *Jassa* sp.1, peak abundance was found at 35 m rather than 0 m sites (Figure 5.3-31). There was a group of species more positively correlated with intermediate distances; these included the amphipods *Tipimegus cf. thalerus, Hirsutonuphis intermedia,* and *Hippomedon cf. Hippolyte* and the gastropod *Anabathrid* sp.2. Species showing negative relationships with proximity to the cages (and positive relationships with more distant sites) included the amphipods *Bybilus mildura, Aorid* sp.1, *Ampelisca cf. australis,* and *Phoxocephalid sp.3,* the isopod *Anthurid* sp.4, the tanaid *Apseudid* sp.1, the anemone *Edwardsia* sp., polychaetes *Paraprionospio coora* and *Scalibregma inflatum*, and the gastropod *Maoricolpus roseus.*

In survey 3, the species showing a positive relationship with proximity to the cages included polychaetes *Capitella* sp., *Pectinaria antipoda, Perinereis* sp., *Cirratulid* sp.2, the gastropod *Anachis atkinsoni,* the amphipod *Jassa* sp.1, the caprellid *Caprella penatis* and the tanaid *Leptocheliid* sp.1 (Figure 5.3-33). Species positively correlated with the intermediate distances included the ostracod *Euphilomedes* sp., the cumacean Gynodiatylid sp.1, the amphipod *Hirsutonuphis intermedia,* the nemertean sp.1 and the gastropod *Turbonilla fusca.* Species showing negative relationships with proximity to the cages (and positive relationships with more distant sites) included the amphipods *Bybilus mildura, Aorid* sp.1, *Ampelisca cf. australis*, the isopod *Anthurid* sp.4, the tanaid *Apseudid* sp.1, the anemone *Edwardsia* sp., and the gastropod *Pagurixus handrecki.*

In survey 4, the species showing a positive relationship with proximity to the cages included the polychaetes *Capitella* sp., *Phoxocephalid* sp.2 and *Perinereis* sp., the bivalve *Mytilus galloprovincialis* and the gastropod *Nassarius nigellus* (Figure 5.3-34). Species positively correlated with the intermediate distances included the isopod *Anthurid* sp.4, the tanaid *Kalliapseudes struthi,* amphipods *Tipimegus cf. thalerus* and *Lysianassid* sp.1, the polychaete *Spionid* sp.4 and the bivalve *Pratulum thetidis.* Species showing negative relationships with proximity to the cages (and positive relationships with more distant sites) included the amphipod *Bybilus mildura,* the tanaid *Apseudid* sp.1, and the spionid polychaete *Prionospio kulin.*

The unconstrained ordination from the LVM showed a separation of samples based on their species composition and relationship to distance from the cage (i.e. along latent variable 1; Figure 5.3-35). In survey 1, species composition at the 0 m and to a lesser extent at 35 m were distinct from the assemblages found at 100 m and 200 m from the cages. Of the ten most important species identified in the LVM, the polychaete *Capitella* sp. was characteristic of the 0 m sites (~8,450 ind. m^{-2}), but it was also reasonably common out to 200 m (~1140, 180 and 220 ind. m^{-2} at 35, 100 and 200 m sites respectively). The amphipod *Jassa* sp.1 was also characteristic of the assemblages found near the cages but was rare in comparison to *Capitella* sp. The polychaete *Onuphid* sp.1 was most abundant at 35 m. The species identified as most characteristic of sites more distant from the cages were the polychaetes *Prionospio kulin* and *Ampharetid* sp.1, the isopod *Anthurid* sp.4 and the tanaid *Apseudid* sp.1.

In survey 2 there was clearer separation in species composition according to distance, with the assemblages at 0 m and 35 m becoming more distinct and separation emerging between 100 m and 200 m sites and the 500 and 1000 m sites (Figure 5.3-35). The polychaete *Capitella* sp. was again a characteristic species of the community at 0 m, but in much lower abundances compared to survey 1 and only found at 0 m (~2150 ind. m^{-2}), 35 m (~100 ind. m^{-}

^{2}) and 100 m (~2 ind. m^{-2}). The amphipod *Jassa* sp.1 and the caprellid *Caprella penatis* were also characteristic of the assemblages found near the cages, with *Jassa* sp.1 in much higher abundance at the 0 m site in survey 2 (~440 ind. m^{-2}) compared to survey 1 (~40 ind. m^{-2}). The heart urchin *Echinocardium cordatum* was most abundant at 35 m sites and the onuphid polychaete *Hirsutonuphis intermedia* at 35 and 100 m sites. The amphipods *Bybilus mildura*, *Isaeid* sp.3 and *Aorid* sp.1, the screw shell *Maoricolpus roseus* and the polychaete Sabellid sp.2 were characteristic of sites with increasing distance from the cages; *Isaeid* sp.3 and *Maoricolpus roseus* were notably more abundant at the 1000 m sites compared to other distances.

The separation in species composition according to distance emerged further in survey 3; the assemblages at 0 m, 35 m, 100, 200 m were distinct from each other and the 500 and 1000 m sites (Figure 5.3-35). *Capitella* sp. abundance increased in survey 3 and it was again characteristic of the 0 m sites (~6,600 ind. m^{-2}), but also reasonably common out to 200 m (~770, 180 and 30 ind. m^{-2} at 35, 100 and 200 m sites respectively). The amphipod *Jassa* sp.1, the polychaete *Cirratulid* sp.2, the tanaid *Leptocheliid* sp.1 and the caprellid *Caprella penatis* were all characteristic of the assemblages at sites closer to the cages. In contrast the amphipods *Byblis mildura*, *Isaeid* sp.3, bivalves *Pratulum thetidis* and *Myadora* sp.1, and the spionid polychaete *Paraprionospio coora* were characteristic of sites with increasing distance from the cages. In survey 4 there was greater overlap in assemblages at 0 and 35 m sites and at 100 and 200 m sites, while the 500 m and 1000 m sites were distinct from each other and the other distances (Figure 5.3-35). *Capitella* sp. returned to lower abundances in survey 4 at the 0 m sites (~1290) but remained in high abundances at the 35 m sites (~690 ind. m^{-2}). The bivalve *Mytilus galloprovincialis* and polychaetes *Perinereis* sp. and *Phyllodocid* sp.2. were also characteristic of sites closer to the cages; a similar pattern was observed for these species in the other surveys, but they were not in the top ten most significant species identified by the LVM. Of these three species, *Perinereis* sp. was the most common with an average abundance of ~100 and 210 ind. m^{-2} at 0 and 35 m, respectively. The polychaete *Spionid* sp.4 was found at all distances except the 0 m sites and was most common at the 200 and 500 m sites. The amphipod *Lysianassid* sp.1 while particularly common was more common (~20 ind. m^{-2}) at the 100 and 200 m sites. The amphipods *Isaeid* sp.3, *Phoxocephalid* sp.3, the polychaete *Scalibregma cf. inflatum* and the bivalve *Myadora* sp.1 were only found at the 500 and 1000 m sites and in all cases, they were far more abundant at 1000 m.

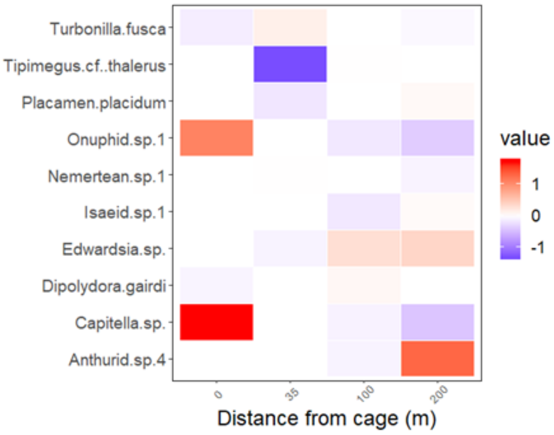


Figure 5.3-26 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Trumpeter lease in survey 1.

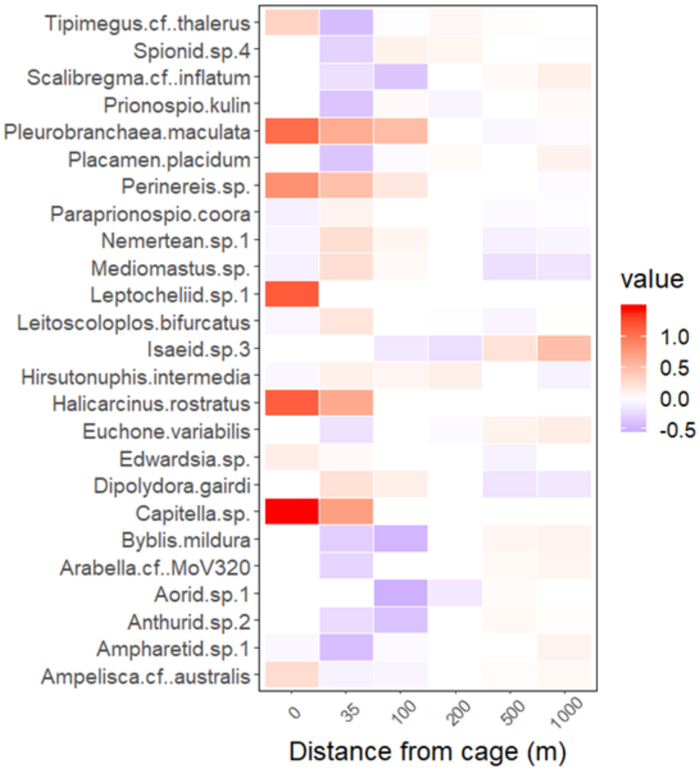


Figure 5.3-27 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Trumpeter lease in survey 2.

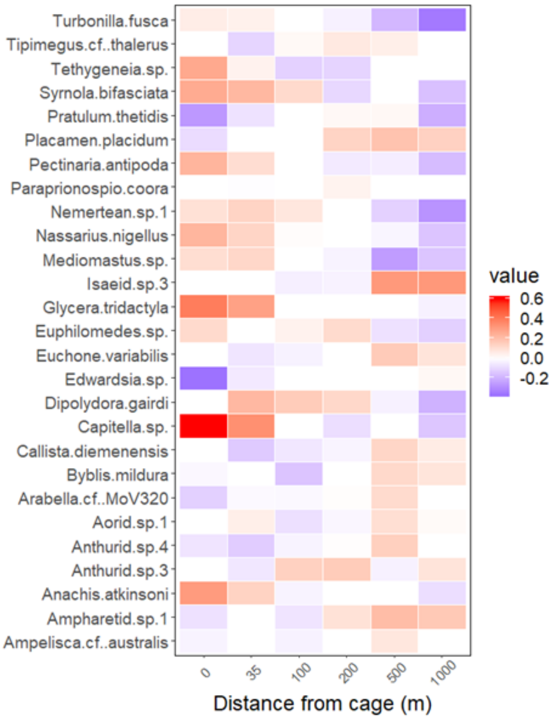


Figure 5.3-28 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Trumpeter lease in survey 3.

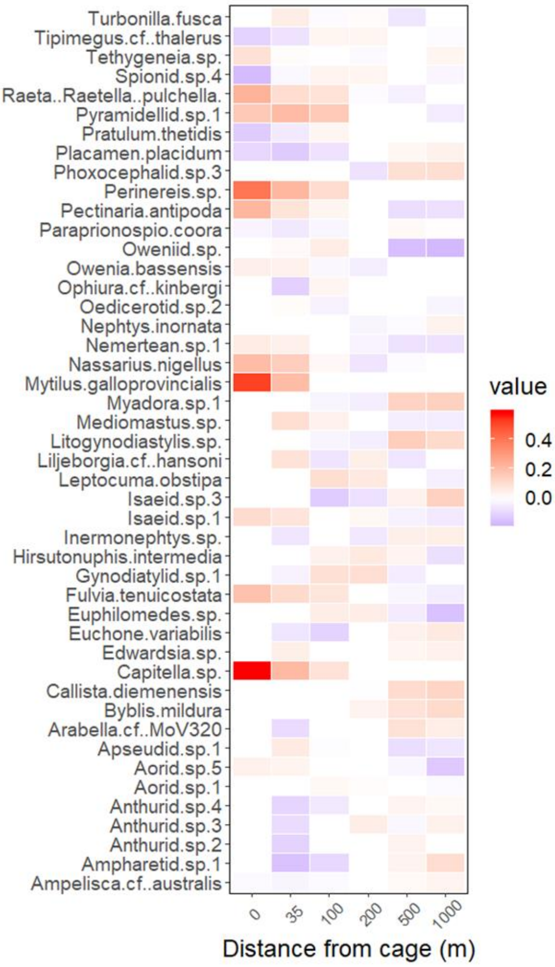


Figure 5.3-29 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Trumpeter lease in survey 4.

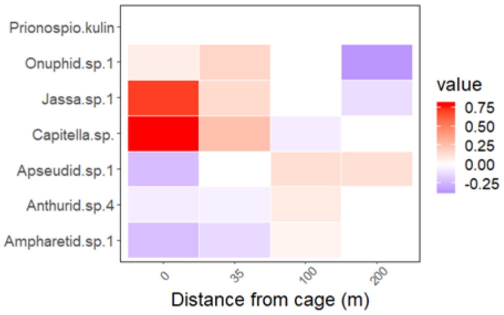


Figure 5.3-30 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Storm Bay 1 lease in survey 1.

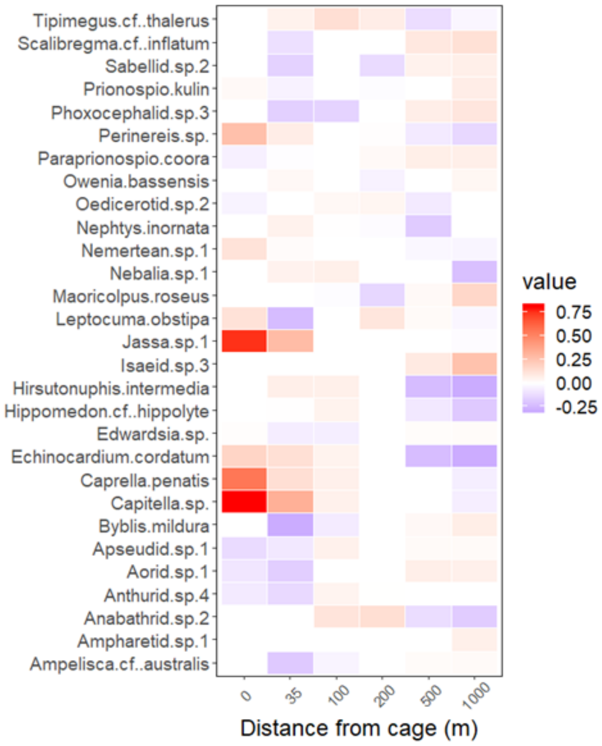
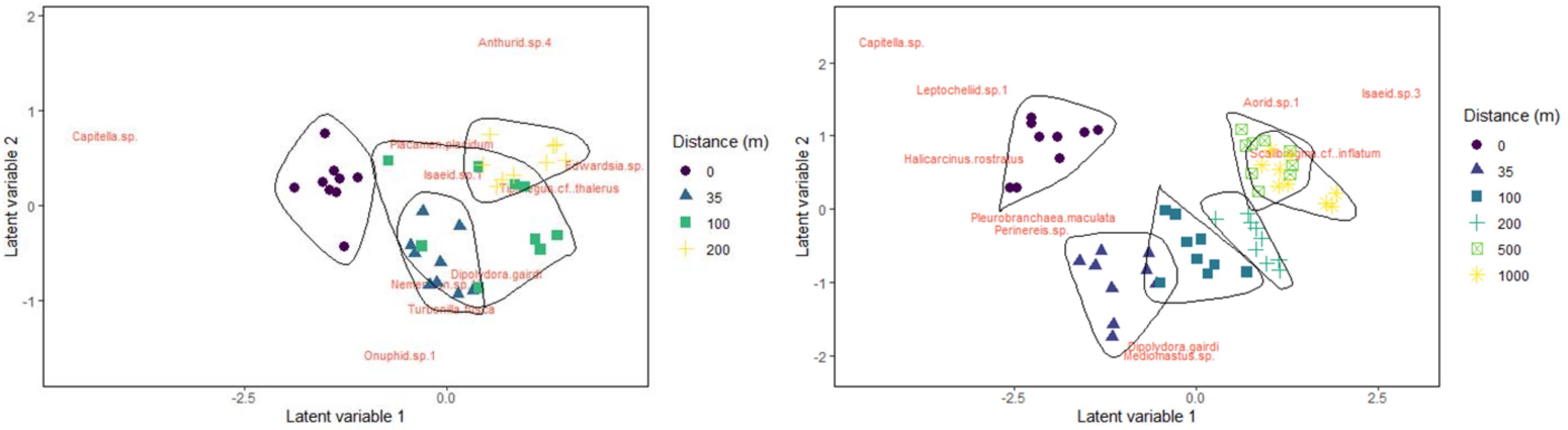


Figure 5.3-31 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Storm Bay 1 lease in survey 2.

a. Survey 1

b. Survey 2



c. Survey 3

d. Survey 4

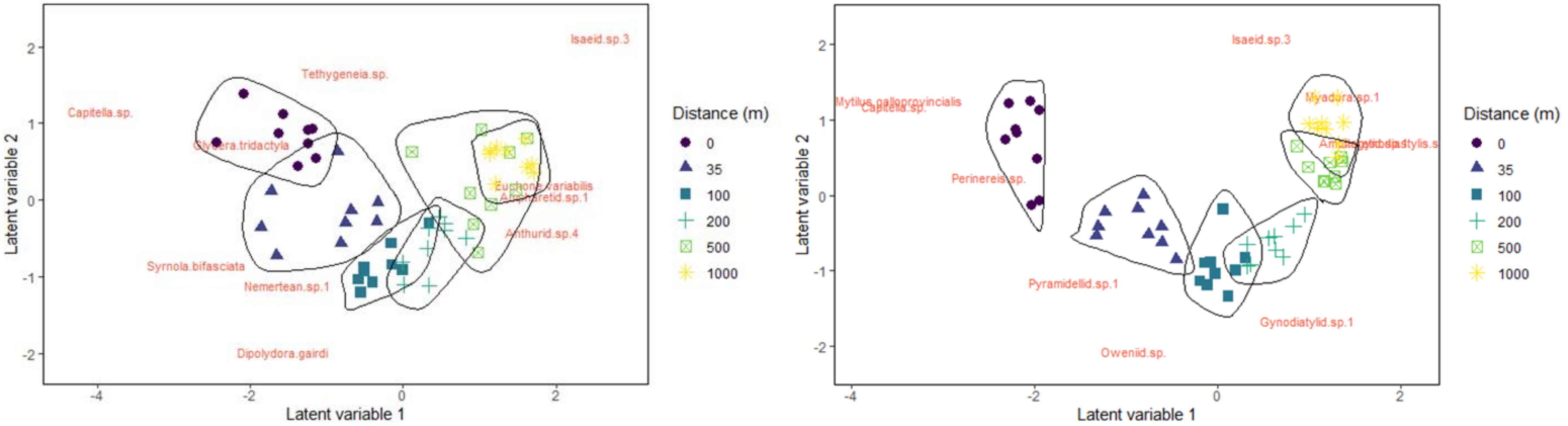


Figure 5.3-32 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1 -4 at the Trumpeter lease. Samples collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and with similar distance from the origin are more correlated.

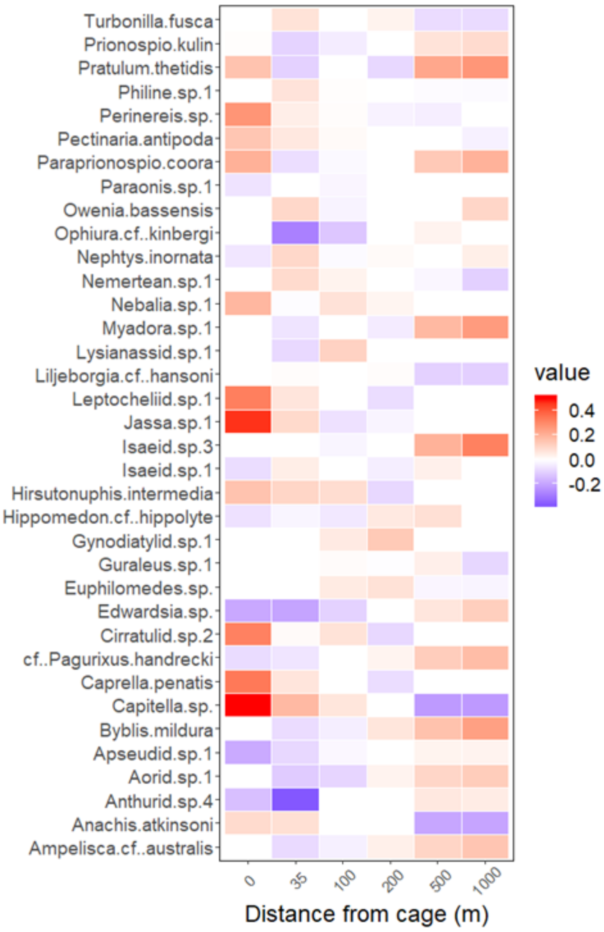


Figure 5.3-33 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Storm Bay 1 lease in survey 3.

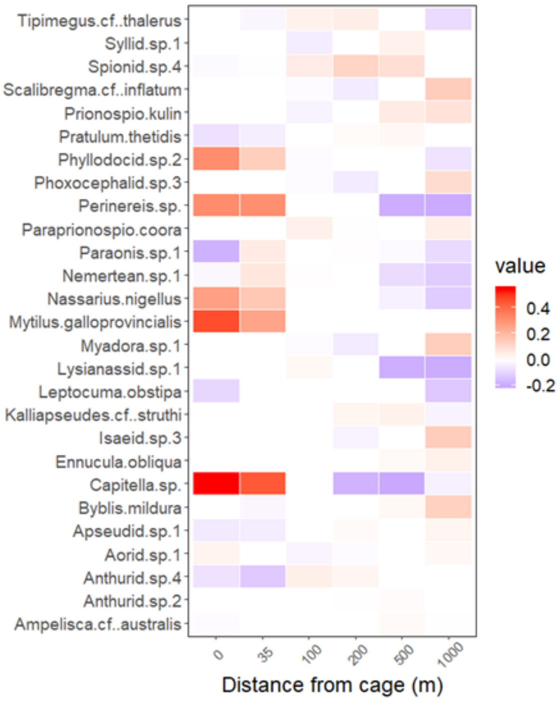
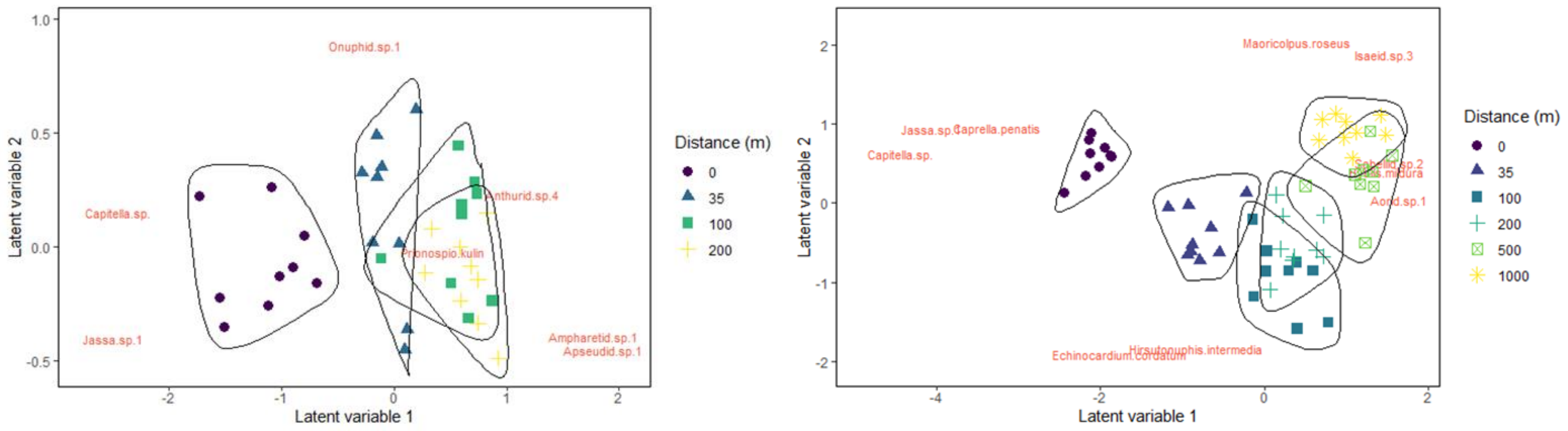


Figure 5.3-34 Species contribution to community composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Storm Bay 1 lease in survey 4.

a. Survey 1

b. Survey 2



c. Survey 3

d. Survey 4

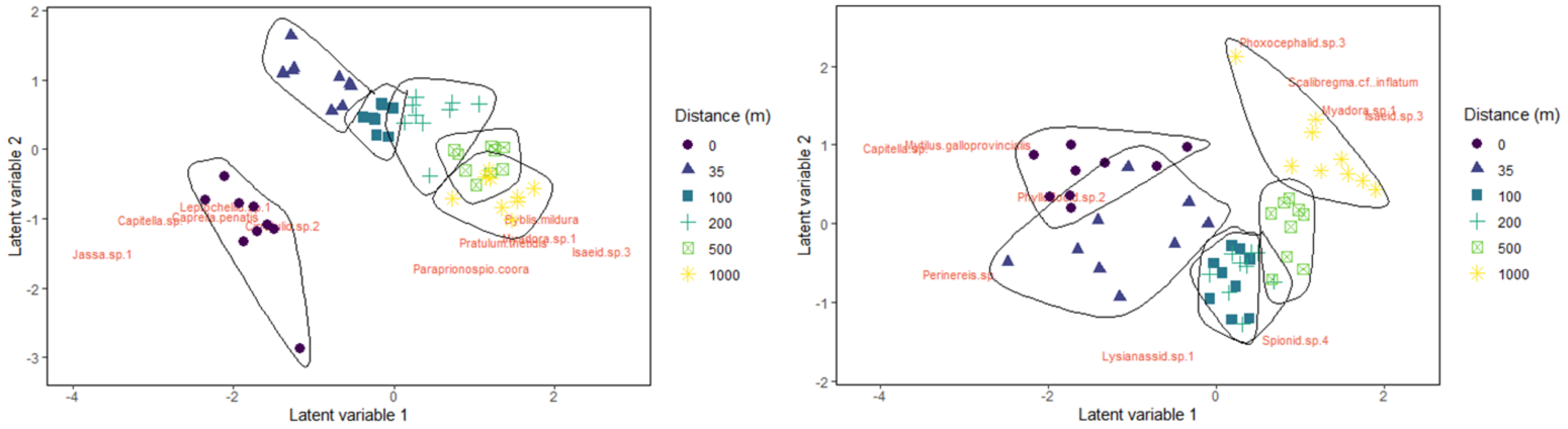


Figure 5.3-35 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1 -4 at the Strom Bay 1 lease. Samples collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and with similar distance from the origin are more correlated.

5.3.1.3 Visual

Trumpeter. There was a significant effect of distance on ROV health scores, with 0 m sites clearly having the lowest scores in each of the surveys (Figure 5.3-36;Table 5.3-1). There was a significant difference in ROV scores between the 35 m and outer controls in survey 2, with ROV scores lower at the 35 m sites, but there was no significant difference in scores between the 200 m transect sites (35 m for lease boundary) and outer controls or between the inner and outer control sites in each of the surveys (Table 5.3-1). There was a significant effect of survey at the cages (0 m) but not at 35 m or the more distant sites (500 m, 100 m and CO; Figure 5.3-37, Table 5.3-2).

Storm Bay 1. The 0 m cage sites had the lowest ROV health scores in each of the surveys, but the effect of distance was not significant in survey 4 (Figure 5.3-36;Table 5.3-1). There was no significant difference detected in ROV scores between the 35 m and outer controls, 200 m and outer controls or between the inner and outer control sites in the 2 surveys (survey 2 & 3) where these tests could be conducted (Table 5.3-1). There was a significant effect of survey at the 0 m and 1000 m sites but not at the other sites (35 m, 500 m and CO; Figure 5.3-37, Table 5.3-2).

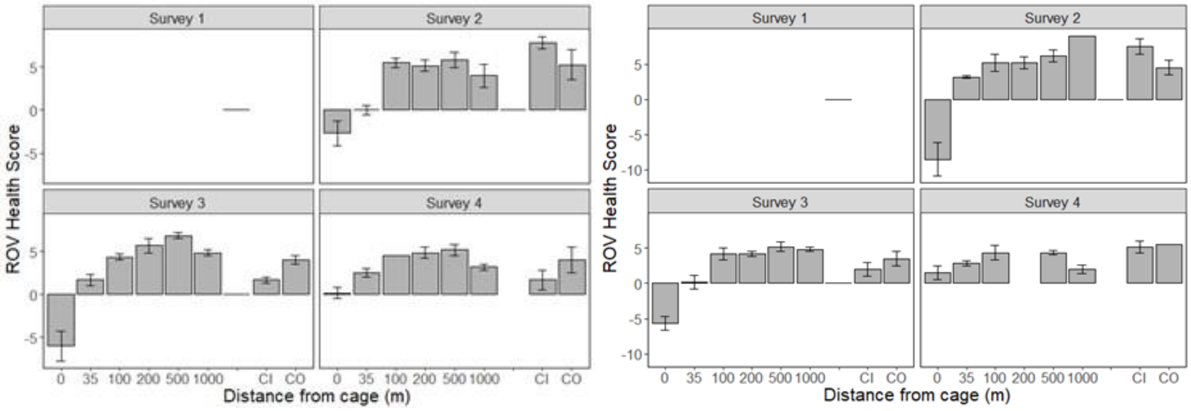


Figure 5.3-36 ROV health score (average ±SE) with distance from the cages (on transects) and at inner (CI) and outer controls (CO) for each survey at the Trumpeter (left) and Storm Bay 1 (right) leases. Data was not available for Survey 1.

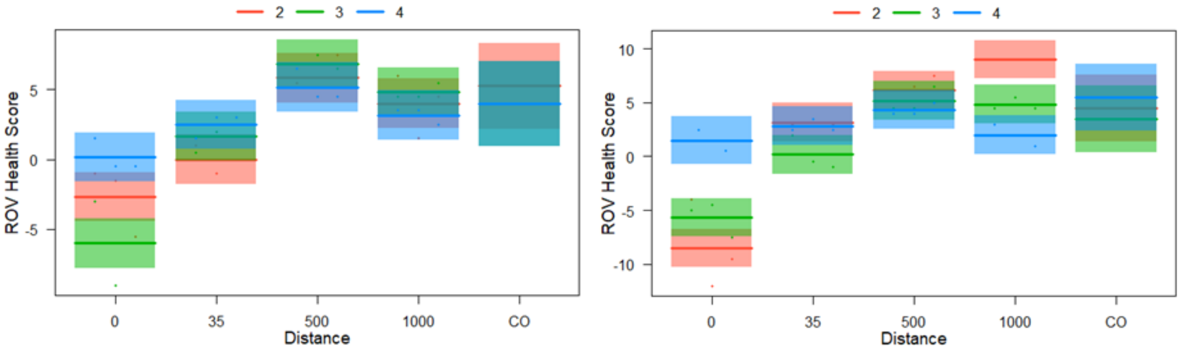


Figure 5.3-37 ROV health score (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 35 m) and at the distances furthest from the cages (500, 1000 m and CO) at the Trumpeter (left) and Storm Bay 1 (right) leases. Data was not available for Survey 1.

5.3.1.4 Sediment Processes

Trumpeter. Sediment respiration measured via dissolved oxygen consumption decreased as a function of distance from cage in both surveys (Figure 5.3-38, Table 5.3-3). Similarly, respiration measured as dissolved inorganic carbon production (i.e. principally carbon dioxide produced during respiration) decreased as a function of distance from cage in both surveys (Figure 5.3-38, Table 5.3-3). Rates of respiration measured by both oxygen consumption and dissolved inorganic carbon production were markedly higher in survey 3 at the 0 m cage sites (~7500 O_2 µmol m^{-2} $h^{-1}$and ~ 9000 DIC µmol m^{-2} h^{-1}) compared to survey 4 at the 0 m cage sites (~2000 O_2 µmol m^{-2} $h^{-1}$and ~ 3500 DIC µmol m^{-2} h^{-1}). At the other distances respiration rates were generally higher in survey 4 compared to survey 3.

Fluxes of ammonium also decreased as a function of distance from cage in both surveys (Figure 5.3-39, Table 5.3-3), with much higher fluxes recorded at 0 m in survey 3 compared to 4. Again, at the other distances, ammonium fluxes were generally higher in survey 4 compared to survey 3. Nitrate + nitrite fluxes were directed into the sediment (i.e. uptake) at several sites across both surveys (Figure 5.3-39). In survey 3, there was a significant effect of distance, with the flux into the sediment decreasing with distance from the cage, and from 200 m there was a release of nitrate + nitrite from the sediment (Table 5.3-3). In survey 4, there was also a general trend of nitrate + nitrate shifting from net uptake to net release as a function of distance from the cage; however, the fluxes were quite variable, and the effect of distance was not significant in survey 4 (Table 5.3-3). The rate of nitrate + nitrate uptake was higher in survey 3 compared to survey 4 at the 0 m cage sites, but there was no clear difference in rates at the other distances.

Fluxes of phosphate were elevated at the cage site and the effect of distance was significant in both surveys (Figure 5.3-40, Table 5.3-3). Like the other nutrients, phosphate release at 0 m was higher in survey 3 compared to survey 4, but at the other distances, the release was generally higher in survey 4 compared to 3.

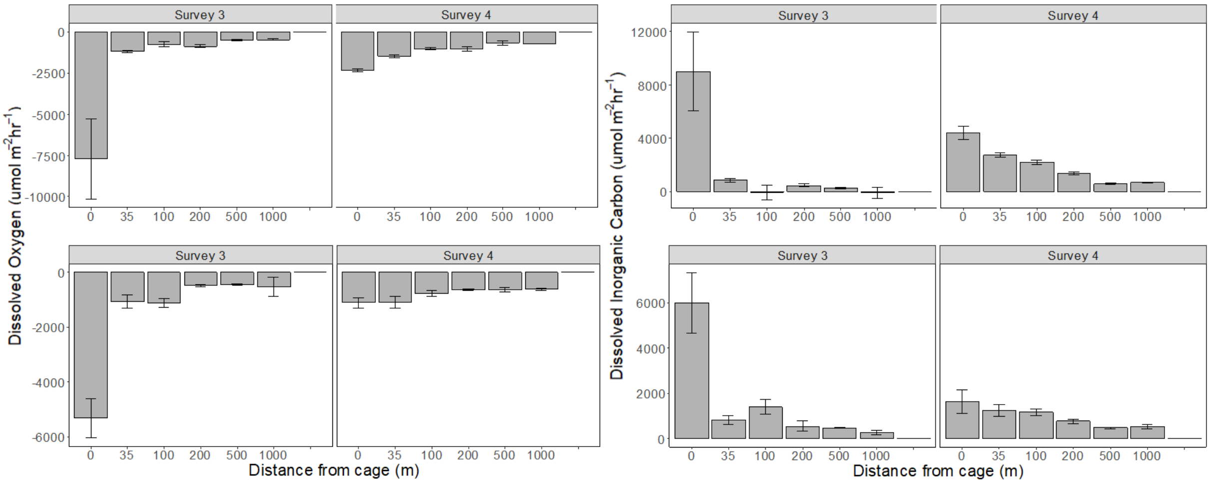


Figure 5.3-38 Sediment respiration: dissolved oxygen (left) and dissolved inorganic carbon right) fluxes (umol m^{-2} h^{-1}) (± SE) with distance from the cages on transects in survey 3 and 4 at the Trumpeter (top) and Storm Bay 1 (bottom) leases. Sediment flux assessments were not conducted in surveys 1 and 2 or at inner and outer controls.

*Storm Bay 1***.** The patterns with distance and between surveys were like those seen at Trumpeter. Sediment respiration measured via dissolved oxygen consumption and dissolved inorganic carbon production (i.e. principally carbon dioxide produced during respiration) decreased as a function of distance from cage in both surveys (Figure 5.3-38, Table 5.3-3).

Rates of respiration measured by both oxygen consumption and dissolved inorganic carbon production were considerably higher in survey 3 at the 0 m cage sites (~5500 O_2 µmol m^{-2} $h^{-1}$and ~ 6000 DIC µmol m^{-2} h^{-1}) compared to survey 4 at the 0 m cage sites (~800 O_2 µmol m^{-2} $h^{-1}$and ~ 1500 DIC µmol m^{-2} h^{-1}). At the other distances respiration rates also appeared marginally higher in survey 4 compared to survey 3.

Fluxes of ammonium also decreased as a function of distance from the cage in both surveys (Figure 5.3-39, Table 5.3-3), with much higher fluxes recorded at 0 m in survey 3 compared to 4. At the other distances, amffmonia fluxes were generally higher in survey 4 compared to survey 3. Nitrate + nitrite fluxes were directed into the sediment (i.e. uptake) at several sites in both surveys (Figure 5.3-39). In survey 3, there was a significant effect of distance, with the flux into the sediment decreasing with distance from the cage, and from 100 m there was a release of nitrate + nitrite from the sediment (Table 5.3-3). In survey 4, there was also a general trend of nitrate + nitrate shifting from net uptake to net release as a function of distance from the cage; however, the effect of distance was not significant (Table 5.3-3). The net flux of nitrate + nitrate was more positive in survey 4. The uptake was lower in survey 4 compared to survey 3 at the 0 m and 35 m from cage sites, and the rate of release was higher at 100 m and 200 m in survey 4 compared to 3. Fluxes of phosphate were elevated at the cage site in survey 3 and the effect of distance was significant (Figure 5.3-40, Table 5.3-3). In contrast there was no clear pattern of response with distance from the cage in survey 4.

Table 5.3-3 Results of the generalised linear model testing for the effect of distance for sediment nutrient fluxes for surveys 3 and 4 at the Trumpeter and Storm Bay 1 leases. Because nutrient fluxes were only measured on 2 transects at Trumpeter and 1 transect at Storm Bay 1, planned comparisons between sites and surveys were not conducted.

	Trumpeter		**Storm Bay 1**	
	F ratio	**p**	**F ratio**	**p**
Dissolved Oxygen				
Survey 3	21.635	**0.010**	25.872	**0.007**
Survey 4	165.56	**<0.001**	14.412	**0.019**
Dissolved Inorganic Carbon				
Survey 3	18.65	**0.012**	23.809	**0.008**
Survey 4	107.74	**<0.001**	31.312	**0.005**
Ammonium				
Survey 3	14.822	**0.018**	24.039	**0.008**
Survey 4	14.843	**0.018**	55.388	**0.002**
Nitrate + Nitrite				
Survey 3	27.501	**0.006**	45.425	**0.003**
Survey 4	4.390	0.104	5.171	0.085
Phosphate				
Survey 3	14.246	**0.020**	15.201	**0.018**
Survey 4	59.890	**0.002**	4.078	0.114

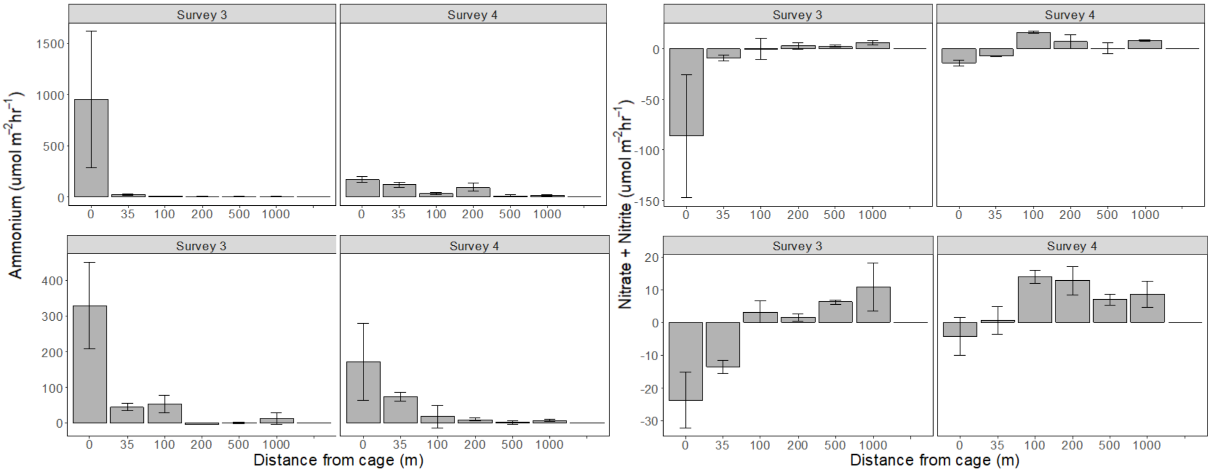


Figure 5.3-39 Sediment ammonium (left) and nitrate + nitrite (right) fluxes (umol m^{-2} h^{-1}) (± SE) with distance from the cages on transects in survey 3 and 4 at the Trumpeter (top) and Storm Bay 1 (bottom) leases. Sediment flux assessments were not conducted in surveys 1 and 2 or at inner and outer controls.

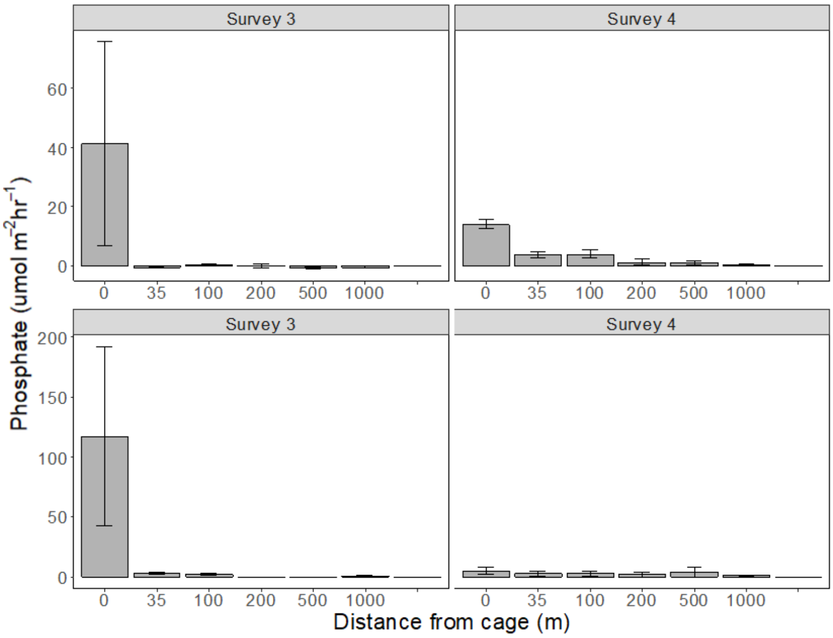


Figure 5.3-40 Sediment phosphate flux (umol m^{-2} h^{-1}) (± SE) with distance from the cages on transects in survey 3 and 4 at the Trumpeter (top) and Storm Bay 1 (bottom) leases. Sediment flux assessments were not conducted in surveys 1 and 2 or at inner and outer controls.

5.3.1.5 Historical comparison

Trumpeter. Redox potential values (mV) at each of the control sites used from the 2014 (control x) and 2016 (control y) baselines varied between an average of 286 mV and 149 mV respectively (Table 5.3-4). In the IMAS surveys, the average varied between 224 and 451 mV at control x and 193 and 346 mV at control y. Not surprisingly the average redox value at the farm sites was substantially lower (3 -109 mV) except in the final survey when it was higher (219 mV) and within the range recorded at the controls. Sulphide concentrations at the control sites were negligible (<2 µM) during the 2014 and 2016 baselines. Average sulphide concentrations across three IMAS surveys were negligible at control x whilst at control y they ranged from 8 - 68 µM. As expected, farm sites showed the highest sulphide concentrations (ranging from 22 - 124 µM) and the lowest redox potential (ranging from 33-202 mV) relative to the control sites across the three recent surveys (Table 5.3-4).

Macrofaunal abundance (ind. m^{-2}) was quite variable across the control sites in the two baseline surveys prior to farming, ranging from an average of 583 ind. m^{-2} at control x in 2014 to 2449 ind. m^{-2} at control site y in 2016. In the four IMAS surveys, average abundance was similarly variable at both control sites. Control site x ranged from 1215-2247 ind. m^{-2} and control y from 430-1812 ind. m^{-2}. Average abundance at the compliance sites displayed a similarly large range in abundance (1182-2742 ind. m^{-2}), but typically fell within the range observed at the control sites in each of the surveys. As expected, the average abundance at the farm sites was higher than at the control and compliance sites ranging from an average of 2788 ind. m^{-2} to 4440 ind. m^{-2}. The average number of macrofaunal families was highly variable across the sites used from the two baseline surveys prior to farming, ranging from 18 at control y in 2014 to 49 at control site x in 2016. In the IMAS surveys, the number of families recorded was also variable, ranging from 13-39 at control sites and 24-43 at the compliance sites. Except for survey 3, the number of families at the farm sites was lower than at the control and compliance sites. Macrofauna diversity ranged from 2.4 to 3.4 at the baseline survey sites, and from 2.2 to 3.2 across the more recent surveys at the control and compliance sites. Diversity was lower at the farm compared to the compliance and control sites across all surveys. Overall, and as expected, farm sites generally showed the highest abundance, lowest diversity and lowest number of families compared to the compliance and control sites.

The unconstrained ordination from the LVM showed a separation of samples based on their family composition and relationship with sample location (i.e. CI, CO and 0 m) and survey (Figure 5.3-41). Not surprisingly, there was a clear separation of farm (0 m) sites from the compliance and control sites with Capitellidae (mostly *Capitella* sp.) the key indicator family of farm sites. The families Mytilidae (bivalves), Phoxichlidiidae (sea spiders), Palaemonidae (shrimp) and Janiridae (isopods) while far less common were typically found at the farm sites when present. There was no clear or consistent separation in family composition between compliance and control sites across the surveys pre- or post- the commencement of farming. The 2016 baseline control site clustered in the middle of the distribution of control and compliance sites from the IMAS surveys, and there was general overlap between the 2014 compliance sites and the IMAS sites. Control site x in 2014 sits distinct in the top right of the plot, and while there is no overlap with the same site in the IMAS surveys, the cluster of IMAS control sites at the top right represent the same site. Although rare, the family Arcturidae (isopods) was characteristic of this site in both 2014 and the IMAS surveys, while the family Whiteleggiidae (tanaids) was only found in the IMAS samples and not the 2014 samples at this site.

Storm Bay 1. Figure 5.3-42 shows the lease and sites surveyed in the 2015 baseline and by IMAS from 2016-18. For the comparison of compliance sites, the following sites were used across the two data sets: 2015 (5,6,7,8,9 & 10) and for 2016-18 (100 m {SBU3, SBM3, SBL3} sites and CI {SBCI-4, SBCI-5 SBCI-6 sites). For the comparison at control sites, there were two sites with overlap from the 2015 baseline (C2 & C6) and the IMAS surveys (SBCO3 & SBCO4).

Redox potential values (mV) at the compliance and control sites in 2015 varied between an average of 230 mV and 268 mV respectively (Table 5.3-5). Average values at the control sites were typically higher in the more recent IMAS surveys, ranging from 218 – 423 mV. Redox values at the compliance sites remained high but were lower at the compliance sites compared to the control site in the IMAS surveys, most notably in the first 2 surveys. Sulphide concentrations at the compliance and control sites were negligible (<3 µM) in the 2015 baseline. Across the three recent surveys in which sulphide was measured, average sulphide concentrations ranged from 0 – 14 µM at the control sites. Sulphide concentrations were variable but higher at the compliance sites relative to the controls in the IMAS surveys, notably higher in survey 4. As expected, farm sites showed the highest sulphide concentrations (ranging from 28 - 173 µM) and the lowest redox potential (ranging from 7-169 mV) relative to the compliance and control sites across the IMAS surveys.

Macrofaunal abundance (ind. m^{-2}) at the compliance and control sites in 2015 varied between an average of 1087 and 2225 ind. m^{-2}, respectively. A similar range in abundances was observed across the IMAS surveys, ranging from an average of 1575 ind. m^{-2} to 1998 ind. m^{-2} at the control sites, and from 1513 ind. m^{-2} to 2300 ind. m^{-2} at the compliance sites. The average number of macrofaunal families at the compliance and control sites in 2015 varied between 27 and 28, respectively. In the IMAS surveys, the numbers of families ranged from an average of 22 to 33 at the control sites, and from 30-35 at the compliance sites. Macrofauna diversity at the compliance and control sites in 2015 varied between an average of 2.7 and 2.8, respectively. In the IMAS surveys, diversity ranged from an average of 2.3 to

2.9 at the control sites, and from 2.5 to 3.1 and 0.7 to 2.6 at the compliance sites in 2004 and 2016 respectively, and from 1.9 to 2.9 across the more recent surveys. In comparison, farm sites had the highest abundance, lowest diversity and lowest number of families compared to the compliance and control sites across all the surveys.

The unconstrained ordination from the LVM showed a separation of samples based on their family composition and relationship with sample location (i.e. CI, CO and 0 m) and survey (Figure 5.3-42). There was a clear separation of farm (0 m) sites from the compliance and control sites and again Capitellidae (mostly *Capitella* sp.) was the most common key indicator family of farm sites. The families Semelidae (the introduced bivalve *Theora lubrica*), Mytilidae (bivalves), Ischyroceridae (amphipods), Amphilochidae (amphipods), Leptochelidae (tanaids) were far less common but more characteristic of the farm sites. There was no clear or consistent separation in family composition between control sites between the 2015 baseline and IMAS surveys. For the compliance sites, there was some indication of a separation in family composition between the baseline and IMAS compliance sites which were typically sitting to the left of the baseline sites in the ordination.

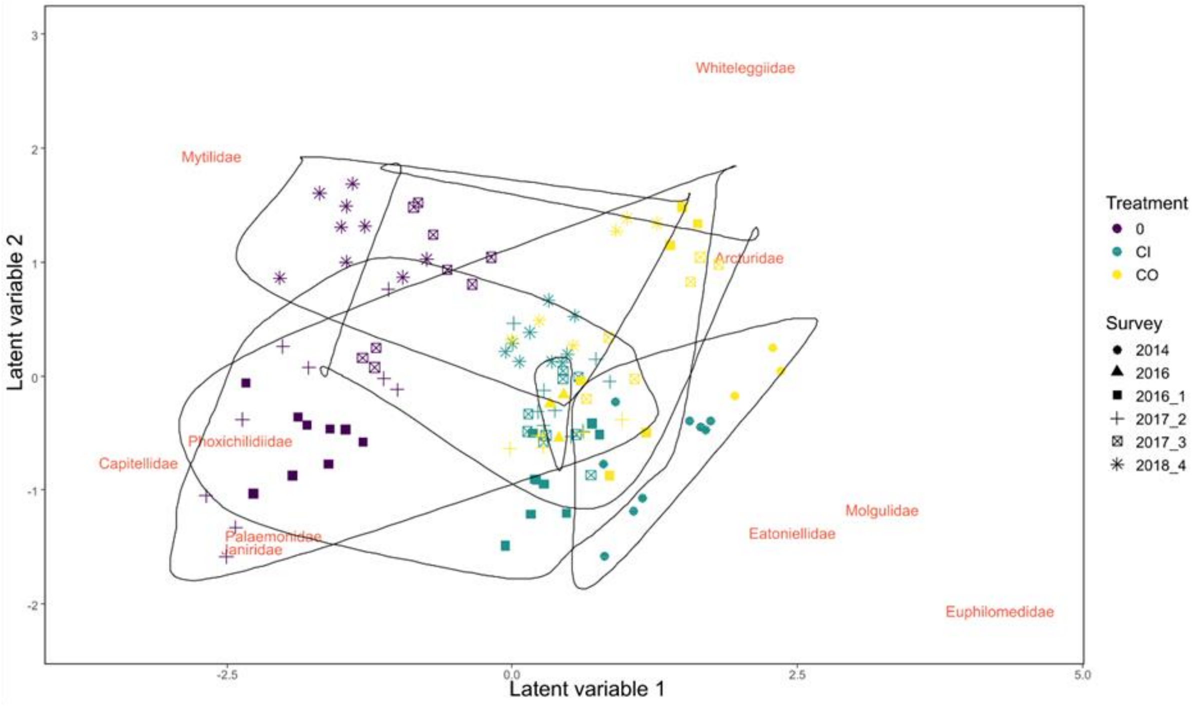


Figure 5.3-41 Biplot showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for inner and outer control sites from the baseline surveys in 2014, 2016 and IMAS surveys 1 -4 conducted over 2016-18 at Trumpeter. For comparison with the effects of farming, sites sampled directly adjacent to the cages (0 m) in the surveys conducted since the commencement of farming are also included. Survey and sample location are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

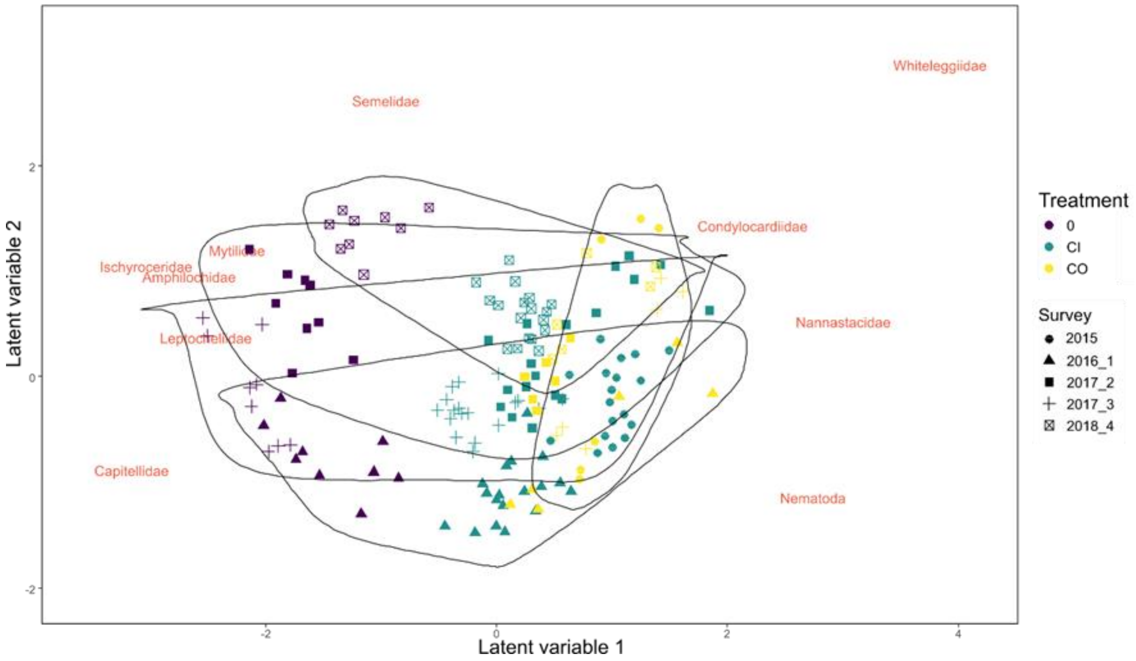


Figure 5.3-42 Biplot showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for inner and outer control sites from the baseline survey in 2015 and IMAS surveys 1 -4 conducted over 2016-18 at Strom Bay 1. For comparison with the effects of farming, sites sampled directly adjacent to the cages (0 m) in the surveys conducted since the commencement of farming are also included. Survey and sample location are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

Table 5.3-4 Historical comparison of physiochemical and macrofaunal parameters at the Trumpeter lease between the 2014 and 2016 baseline surveys and the four IMAS surveys conducted since farming commenced. Sites chosen were based on overlap across surveys to minimise any confounding due to spatial variation. Note, control site x refers to site C1 in the 2014 baseline and site CO2 in the IMAS surveys, while control site y refers to site 12 in the 2016 baseline and site CO1 in the IMAS surveys.

	Sulphide			**Redox**			**Number of Families**			**Abundance (indiv. m^{-2})**			**Diversity (H')**		
Surveys	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT
2014															
compliance	0	(0)	9	291	(56)	9	21	(4)	9	700	(166)	9	2.7	(0.2)	9
control x	0	(0)	3	286	(84)	3	18	(7)	3	583	(202)	3	2.4	(0.4)	3
2016															
control y	2	(2)	3	149	(42)	3	49	(3)	3	2449	(252)	3	3.4	(0.1)	3
2016_1															
compliance							24	(6)	9	1182	(482)	9	2.5	(0.3)	9
control x				451	(9)	3	23	(4)	3	1215	(291)	3	2.6	(0.4)	3
control y				193	(201)	3	13	(8)	3	430	(320)	3	2.2	(0.8)	3
farm				50	(85)	5	8	(4)	9	4440	(3115)	9	0.3	(0.2)	9
2017_2															
compliance							33	(8)	9	1317	(479)	9	3.0	(0.3)	9
control x	0	(0)	3	400	(45)	3	36	(2)	3	1240	(31)	3	3.3	(0.1)	3
control y	10	(10)	3	243	(25)	3	36	(2)	3	1812	(185)	3	3.1	(0.1)	3
farm	33	(33)	5	5	(49)	5	18	(4)	9	2788	(1630)	9	1.3	(0.3)	9
2017_3															
compliance							31	(5)	9	1361	(386)	9	3.0	(0.1)	9
control x	0	(0)	3	371	(50)	3	27	(3)	3	1407	(297)	3	2.9	(0.2)	3
control y	8	(8)	3	227	(83)	3	23	(4)	3	741	(207)	3	2.9	(0.2)	3
farm	173	(173)	5	109	(52)	5	25	(5)	9	3129	(2098)	9	2.0	(0.5)	9
2018_4															
compliance							43	(8)	9	2742	(877)	9	3.2	(0.2)	9
control x	0	(0)	3	224	(111)	3	37	(3)	3	2247	(177)	3	3.0	(0.1)	3
control y	68	(68)	3	346	(119)	3	39	(3)	3	2074	(121)	3	3.2	(0.2)	3
farm	202	(202)	5	219	(92)	5	24	(4)	9	3040	(1798)	9	2.0	(0.5)	9

Table 5.3-5 Historical comparison of physiochemical and macrofaunal parameters at the Storm Bay 1 lease between the 2015 baseline survey and the four IMAS surveys conducted since farming commenced. Sites chosen were based on overlap across surveys to minimise any confounding due to spatial variation.

	Sulphide			**Redox**			**Number of Families**			**Abundance (indiv. m^{-2})**			**Diversity**		
Surveys	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT	AVG	SD	COUNT
2015															
compliance	3	(4)	18	230	(94)	18	27	(5)	18	1087	(322)	18	2.8	(0.3)	18
control	1	(1)	6	268	(32)	6	28	(7)	6	2225	(1435)	6	2.7	(0.2)	6
2016_1															
compliance				182	(147)	5	30	(7)	18	2300	(623)	18	2.5	(0.4)	18
control				423	(10)	6	22	(7)	6	1649	(1159)	6	2.3	(0.3)	6
farm				13	(57)	5	18	(8)	9	9184	(7477)	9	0.6	(0.4)	9
2017_2															
compliance	9	(5)	5	160	(62)	5	32	(10)	18	1513	(740)	18	3.0	(0.3)	18
control	0	(0)	6	306	(67)	6	33	(13)	6	1575	(1102)	6	2.9	(0.5)	6
farm	28	(21)	5	7	(72)	5	15	(3)	9	2910	(763)	9	1.0	(0.3)	9
2017_3															
compliance	18	(10)	5	212	(58)	5	35	(5)	18	1567	(412)	18	3.1	(0.2)	18
control	5	(7)	6	218	(93)	6	26	(9)	6	1602	(1110)	6	2.6	(0.2)	6
farm	110	(85)	5	39	(110)	5	12	(5)	9	7180	(4866)	9	0.5	(0.2)	9
2018_4															
compliance	127	(107)	5	306	(97)	5	35	(4)	18	2020	(525)	18	3.1	(0.1)	18
control	14	(25)	6	361	(49)	6	31	(7)	6	1998	(1011)	6	2.9	(0.2)	6
farm	173	(143)	5	169	(190)	5	23	(6)	9	2899	(1800)	9	2.1	(0.4)	9

5.3.2 Discussion

The benthic communities at the control sites at both the Storm Bay 1 and Trumpeter leases were highly diverse with 187 and 198 different taxa recorded respectively. Crustaceans were the most abundant taxa, making up 41-59% of the community, followed by polychaetes (18-25%), cnidarians (4-22%) molluscs (10-15%) and echinoderms (1-2%). The most abundant taxa were the anemone *Edwardsia* sp., amphipods *Ampelisca cf. australis*, *Tethygeneia* sp., *Isaeid* sp.2, *Aorid* sp.1, and *Tipimegus cf. thalerus*, the ostracod *Euphilomedes* sp., polychaetes *Prionospio kulin* and *Pectinaria antipoda*, and the bivalve *Pratulum thetidis.*

Response to enrichment

The biodiversity metrics, species richness (S), total abundance (N), and Shannon-Weaver diversity index (H') showed response patterns consistent with prior research and the guidelines established for southern Tasmania (Macleod and Forbes 2004). While total abundance peaked at the cage sites, species diversity and richness were typically reduced before increasing with distance from the cage. Previous research has shown faunal assemblages at impacted sites are less diverse but more abundant due to the dominance of a few opportunistic species (Keeley et al. 2015; Macleod and Forbes 2004; Pearson and Rosenberg 1978). These faunal metrics also generally responded to farm management as expected. At the Trumpeter lease the first two surveys followed periods of continuous stocking (20 and 27 months respectively from when farming first began on the lease in late 2014). The faunal metrics at the cage sites in both surveys were consistent with moderate (diversity index H' >1<2) to major (diversity index H' <1, No spp. <50% ref) impact based on the guidelines produced for sandy and more exposed sites by Macleod and Forbes (2004). The third survey followed a five-to-six-month fallow period, and the metrics were indicative of improved conditions and minor impacts at the cage sites (i.e., diversity index H' ≈ 2, No spp. ≈ ref, total abundance x 2 ref). Seven months after the lease was restocked, the faunal metrics at the cage site in survey 4 were again indicative of minor to moderate impacts depending on the metric used (diversity index H' ≈ 2, No spp. ≈ 50% ref). At the Storm Bay 1 lease, species richness and diversity were reduced at the cage sites relative to the more distant sites in all four surveys, but total faunal abundance was far more elevated at the cage sites relative to the more distant sites in surveys 1 and 3 compared to 2 and 4. 1. Because the transects were across two grids that had different stocking regimes the response to farming was more difficult to interpret at Storm Bay. Nonetheless, feed inputs were lower in the months preceding surveys 2 and 4 on the southern grid that included two or three transects. The only time the diversity index was >2 at the cage site was in survey 4 when both grids had been fallowed for 1-2 months. Importantly, at Storm Bay 1, like Trumpeter, the change in these faunal metrics observed at 35 m from the cage did not exceed the minor impact category across any of the surveys.

Changes in the taxonomic and functional ecology provided greater insight to the ecological and functional changes in response to farm derived organic enrichment. The opportunistic polychaete *Capitella* sp. was by far the most dominant (67% and 81% of the total abundance at Trumpeter and Storm Bay 1 respectively) and characteristic species of the community at the cage sites at both leases. Similarly, the bivalve *Mytilus galloprovincialis* and the amphipod *Jassa* sp., though not particularly abundant, were more common at the cage sites. The presence of *Mytilus galloprovincialis* was most likely a result of net cleaning, given it is a common fouling species on net infrastructure. The change in species composition at the cage sites at Trumpeter was also consistent with farm stocking regimes. *Capitella* sp. were less common, and a wide range of other species were far more abundant at the cage sites relative to the other surveys, during survey 3 when the lease had been fallowed for five-to-six months. The species that became more abundant at the cage sites over this period included species such as polychaetes *Pectinaria antipoda* and *Perinereis* sp., the gastropod *Nassarius nigellus* and amphipods from the family Phoxocephalidae. These species were likely responding opportunistically to the low to moderate levels of organic enrichment and the fact other environmental conditions had improved. The gastropod *Nassarius nigellus* and amphipods from the family Phoxocephalidae are both known to be indicative of recovering sediments and minor/moderate impacts (Macleod et al., 2008).

More broadly, there was a shift in species composition with increasing distance from the cage evident across both leases (Figure 5.3-44, Figure 5.3-45). Several taxa increased in abundance at intermediate distances (35-200 m from the cage). Many of these species were sessile suspension feeders or surface deposit feeders taking advantage of the increased food supply but were more sensitive to the highly enriched conditions directly under and adjacent to stocked pens than their more mobile and tolerant deposit feeding counterparts. Species such as the heart urchin *Echinocardium cordatum*, and polychaetes *Mediomastus* sp. and *Nemertean* sp.1 were generally most common closer to the source of enrichment at 35 m, while other species such as polychaetes *Spionid* sp.4 and *Hirsutonuphis intermedia*, amphipods *Tipimegus cf. thalerus* and *Hippomedon cf. hippolyte,* the ostracod *Euphilomedes* sp., and the brittle star *Ophiura cf. kinbergi* were more common at distances beyond100-200 m. Again, this shift most likely reflects differences in their sensitivity to environmental conditions. A number of these species (or families) are known to be characteristic of the transitory community between heavily impacted and unimpacted communities (e.g., Pearson & Rosenberg 1978; Macleod et al., 2008; Edgar., 2005; 2010). This includes Nassarius, Echinocardium, Phoxocephalidae, Euphilomedes, Nemertea, Spionidae, Pectinaria and Tethygeneia. At the transect sites most distant from the cage (500-1000 m), several of these species remained common, but the increased presence of species such as the amphipods *Ampelisca cf. australis*, *Bybilus mildura, Isaeid* sp.3 and *Aorid* sp.1, and polychaetes *Paraprionospio coora*, *Prionospio kulin, Ampharetid* sp.1 and *Scalibregma cf. inflatum* characterised a community shift that reflects little to no evidence of any particular adverse or increased impacts of organic enrichment. This is consistent with the findings of Macleod et al., (2008) who found crustaceans were more dominant and typical of relatively unimpacted conditions at the sandy exposed site, and Hall (1994) who reported crustaceans as often being particularly sensitive to organic enrichment. In this study crustaceans made up 6-12% (15-30% at 35 m) and polychaetes 77-88% (37-58% at 35 m) of the community at 0 m from the cages at both leases compared to 42-47% for crustaceans and 22 -31% for polychaetes at the outer control sites. Ampeliscid amphipods have been reported to be particularly sensitive to organic enrichment (Macleod et al., 2008), with *Ampelisca* characteristic of unimpacted sediments at an exposed/sand site in that study.

It is also important to acknowledge the potential challenge of spatial confounding when interpreting the patterns in the context of enrichment. At the Trumpeter lease and to some extent at the Storm Bay 1 lease, there were differences in the communities at 500 and 1000 m from the cage; however, the sediments were naturally finer at the 1000m position (greater percentages of 0.063 mm particles), which may help explain the increased presence of several *Ampelisca* and other amphipod species at this site. Whilst there were also subtle differences in the sediment types at the inner and outer controls relative to the other sites, this was not actually reflected in a clear or constant difference in the community assemblage. Thus, establishing baseline conditions through time prior to farming and monitoring is clearly an important tool to distinguish farm induced change from natural variability. This also highlights the importance of using a suite of indicators, both biotic and physico-chem when establishing cause and effect as discussed below.

Redox potential and sulphide concentrations appeared to be the most useful changes in sediment chemistry to detect the gradients of enrichment with distance from cages. Redox potential was more depleted and sulphide concentrations elevated in proximity to the cages across all surveys at both Storm Bay leases. However, changes in redox appeared to be more sensitive to changes in farm management than sulphide concentrations. In survey 3 at Trumpeter and survey 4 at Storm Bay 1, redox potential was higher at the cage site consistent with fallowing/lower stocking preceding the surveys. Sulphide concentrations on the other hand varied significantly between surveys and did not appear to respond to farm management. Macleod and Forbes (2004) also noted the challenges with probe stability and comparability across surveys for sulphide, suggesting it is more useful to describe spatial patterns of enrichment rather than temporal change. The sulphide levels and the ascribed impact stages based on Macleod and Forbes (2004) are also inconsistent with the response determined using faunal assemblages. Sulphide concentrations were 100-300 µM from the cage out to 100 - 200 m in surveys 3 and 4 at both leases, consistent with major effects, yet changes in the faunal community data indicated only minor impacts extended beyond 35 m from the cage. The C and N signatures (content, ratio and isotopes) were indicative of enrichment effects, but typically only at the 0 m cage site, with δ^{15}N the most sensitive measure.

As both sulphide and redox are measured at depth in the sediments, they may measure the longer term, persistent effects of farming relative to the measures that integrate the sediment surface, where sediment re-oxygenation by resuspension, bioturbation and diffusion occur more rapidly. Both % C and N of the sediments were typically far more enriched at the cage site, but otherwise there was no consistent discernible difference that could be attributed to distance at either lease. The C and N contents of the cage sediments were lower in survey 3 at Trumpeter following the period of fallowing and it was notable the content of both C and N were lower at the cage sites in surveys 2 and 4 at the Storm Bay 1 lease. This may indicate the sediment conditions had improved in response to farm management strategies, given two of the three transects were on the southern grid that had experienced two-to-three months of low feed inputs prior to these surveys. Regardless, it was only the δ^{15}N signature of the sediments that appeared sensitive enough to distinguish these more subtle effects of farming. This is an improvement on previous investigations of fish farm impacts in southeastern Tasmania which suggested isotope ratios, including δ^{15}N, lacked discriminatory power to detect farm effect changes due to the background and input isotope signatures being too similar in their isotopic signatures (Edgar et al., 2005; Macleod et al., 2004). Interestingly the improved sensitivity reported here likely reflects the recent shift in protein sources used in fish feeds. Historically, fishmeal was the dominant protein source, and as such it was not readily distinguishable from the background signature δ^{15}N typical of marine organic matter. Now, however, terrestrially derived sources of protein (e.g., plant proteins), which typically have more depleted δ^{15}N values, make up an increasing fraction of the protein used in fish feeds and so are more discernible.

Sediment respiratory activity proved highly sensitive to the more subtle effects of enrichment. Rates of organic matter mineralisation were clearly elevated at the 0 m cage sites relative to the other distances. These results are consistent with findings for the physico-chemical and macrofaunal response parameters; the bulk of organic matter deposition and processing that lead to more significant benthic changes occurred under and within very close proximity (< 35 m) to the cages. Importantly though, the process measurements documented the more extensive spatial input and influence of organic matter deposition. Elevated respiration rates and ammonia production were evident, albeit at much lower levels than at the cage, out to 100 -200 m from the cages (Figure 5.3-43). This is consistent with the more subtle changes in benthic communities and the more sensitive abiotic parameters (i.e., redox, sulphide and $\delta^{15}N$) evident at the same spatial scale. An important finding from other studies (e.g., Keeley et al. 2019; Valdemarsen et al. 2015) is the high level of responsiveness that process rates provide in near cage sediments to farm inputs.

At Storm Bay 1, oxygen consumption and dissolved inorganic carbon production (principally carbon dioxide) at the cage was > 5-fold higher in survey 3 when biomass was close to its peak compared to survey 4 when the grid had been fallow for a few months. Although not to the same extent, this was also reflected in the rates of ammonia and phosphate production and nitrate uptake. In contrast, the change between surveys at the Trumpeter lease was not consistent with expectations based on stocking of the cage grid; process rates were higher at the cage site in survey 3 compared to survey 4, despite an extended period of fallowing/low stocking before survey 3. Importantly though, despite the greater stocking prior to survey 4 at the grid scale, the cage at the origin of the middle transect (i.e., process measurements were only conducted on this transect) was stocked at the time of sampling in survey 3 and not survey 4. This suggests process rates may be highly sensitive to both the timing of, and proximity to, the source of enrichment. It is also notable the process rates at the intermediate distances (35-200 m) were more elevated in survey 4 compared to survey 3. This may reflect the footprint further from the cage reflects the overlap of multiple cages, and thus, the response is indicative of grid scale inputs relative to the measurements taken directly adjacent to the cages.

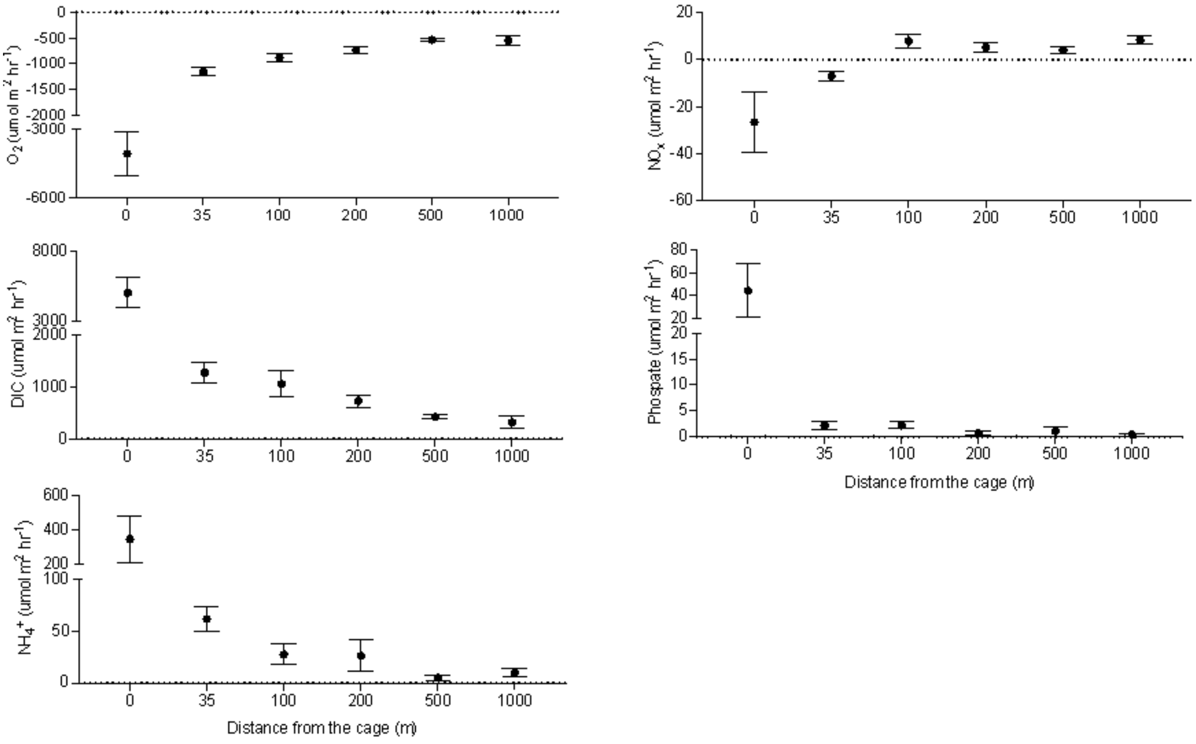


Figure 5.3-43 Summary of sediment fluxes at the Storm Bay leases. Fluxes of oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages averaged (± SE) across leases and surveys.

The strong benthic effect gradient near the cages was reflected in the visual health score. The cage sites clearly had the lowest health scores and the 35 m from cage sites were also clearly lower than the more distant sites. The more subtle effects revealed by the faunal communities and process rates were not as distinguishable from the visual health score. From a regulatory monitoring perspective, the visual assessment method could discriminate what would be considered moderate to major impacts but was less able to distinguish across the range from no evidence of impacts to minor impacts. Importantly, the scoring categories were consistent with those proposed by Macleod and Forbes (2004) for exposed/sandy sites; scores at the cage site were consistent with major to moderate impacts (typically negative and always < 2.5), minor to moderate impacts at 35 m (typically <2.5 and always <5) and minor to no evidence of impacts at the more distant sites (typically >2.5). However, the variability of scores in both space and time was notable, highlighting the importance of a time series at cage, compliance, and reference sites for optimal interpretation.

Implications for monitoring and management

The results at the Storm Bay leases were consistent with those reported elsewhere for southeast Tasmania. Benthic faunal evaluation was the most robust and sensitive measure of sediment condition and response to enrichment. Macleod and Forbes (2004) highlighted regional differences in the key biotic and abiotic indicators and their thresholds based on substrate type from more exposed sandy to sheltered muddier sites in southeast Tasmania. The results described here for both Storm Bay leases were broadly consistent with the indicators and their depiction of different impact stages for the more exposed sandy site. The key biotic and abiotic indicators were all indicative of moderate to major impacts at the cage sites and minor to moderate impacts at the 35 m sites. This is consistent with the changes in faunal community composition and the gradient depicted by the benthic process rates; both measures were able to distinguish more subtle effects that extended further from the cage.

The performance of the physico-chemical parameters as indicators of enrichment was quite variable; most parameters could discern the major effects near the cage, but only redox, $\delta^{15}N$, and to a lesser extent sulphide, appeared capable of reliably distinguishing moderate to minor effects. Biodiversity metrics, total abundance clearly depicted highly enriched conditions, but species richness appeared far more sensitive for discerning moderate to minor effects. Importantly, the results did validate the performance and utility of visual indicators and the ROV health scoring for monitoring major to moderate sediment impacts. However, the results highlighted the importance of using other measures of sediment condition (e.g. faunal evaluation) in conjunction with visual assessment when a more detailed understanding of farm effects is required.

Finally, it is important to note while the results at Storm Bay were in broad agreement with the criteria proposed by Macleod and Forbes (2004) based on the exposed/sandy environment, there were inevitable site-specific differences, particularly regarding the indicator species. For example, although *Capitella* appeared to be largely ubiquitous in its response to highly enriched conditions at all leases in southern Tasmania, there were also inherent differences in the species composition and the response to enrichment between leases and regions. Thus, establishing monitoring locations that minimise background environmental variability as well as establishing baseline conditions over time are necessary and important initial steps for the effective management of these systems.

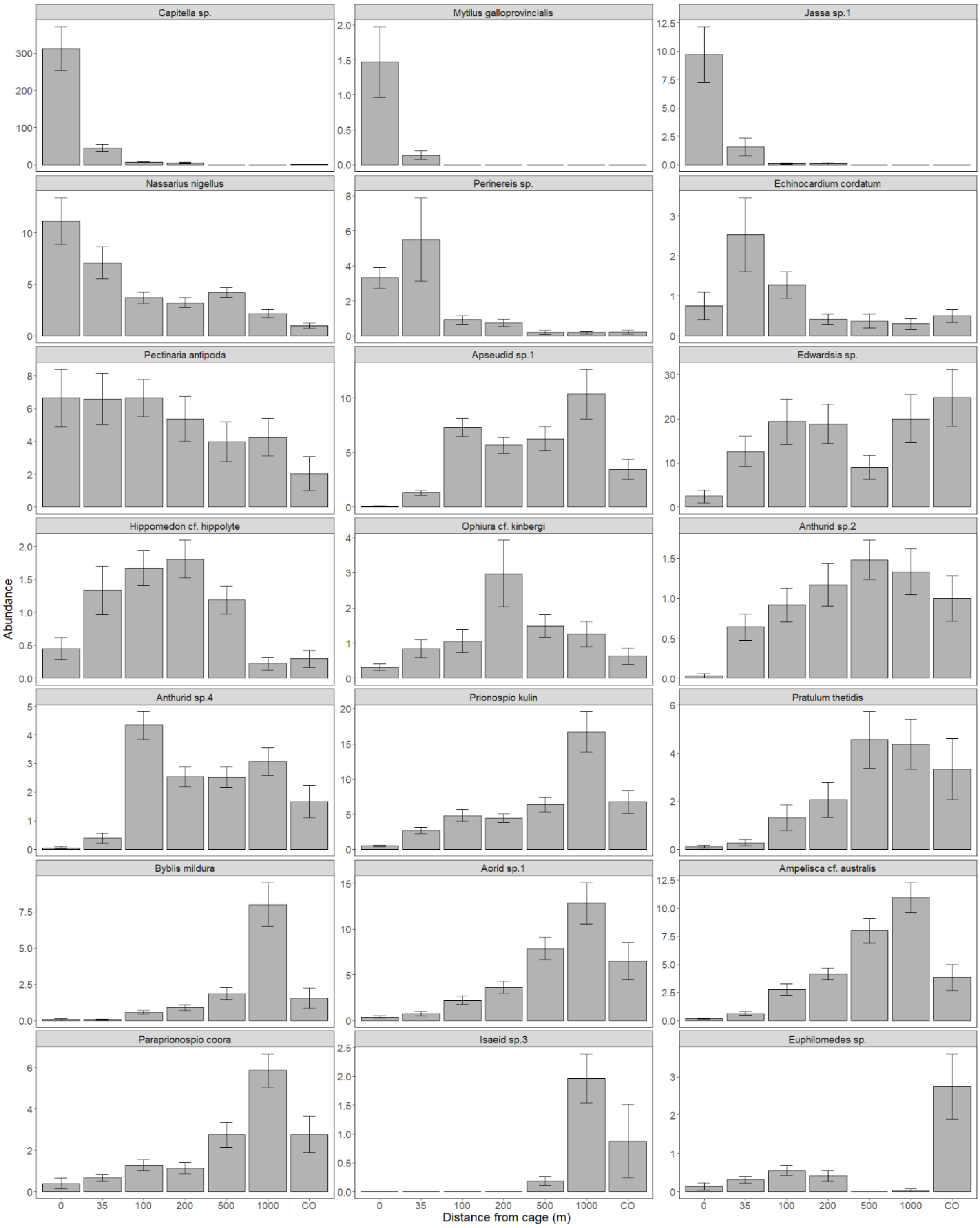


Figure 5.3-44 Average abundance (ind. per grab) of key taxa with distance from the cages (on transects) and at the outer controls (CO) sites across surveys at Storm Bay lease.

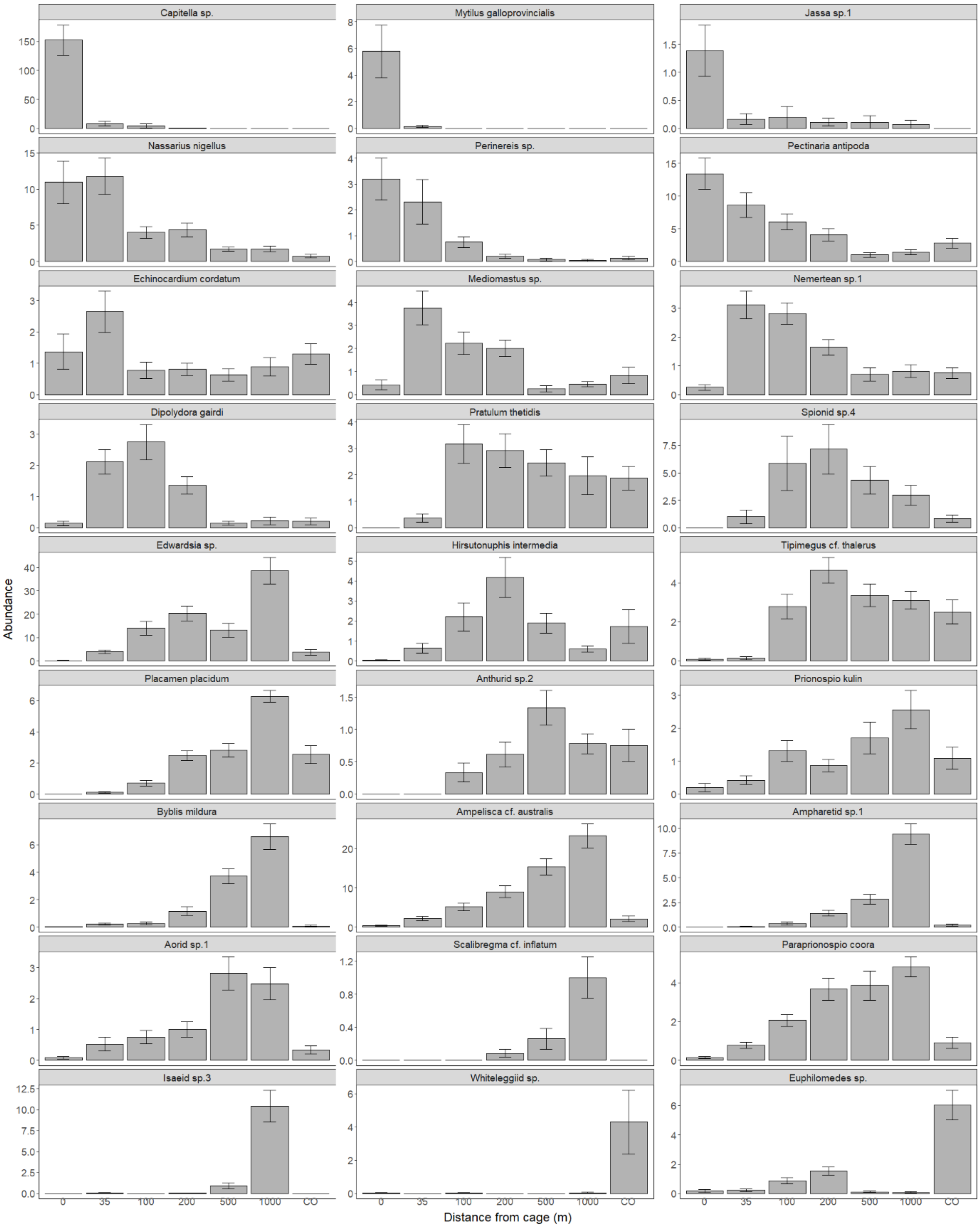


Figure 5.3-45 Average abundance (ind. per grab) of key taxa with distance from the cages (on transects) and at the outer controls (CO) sites across surveys at the Trumpeter lease.

5.3.3 References

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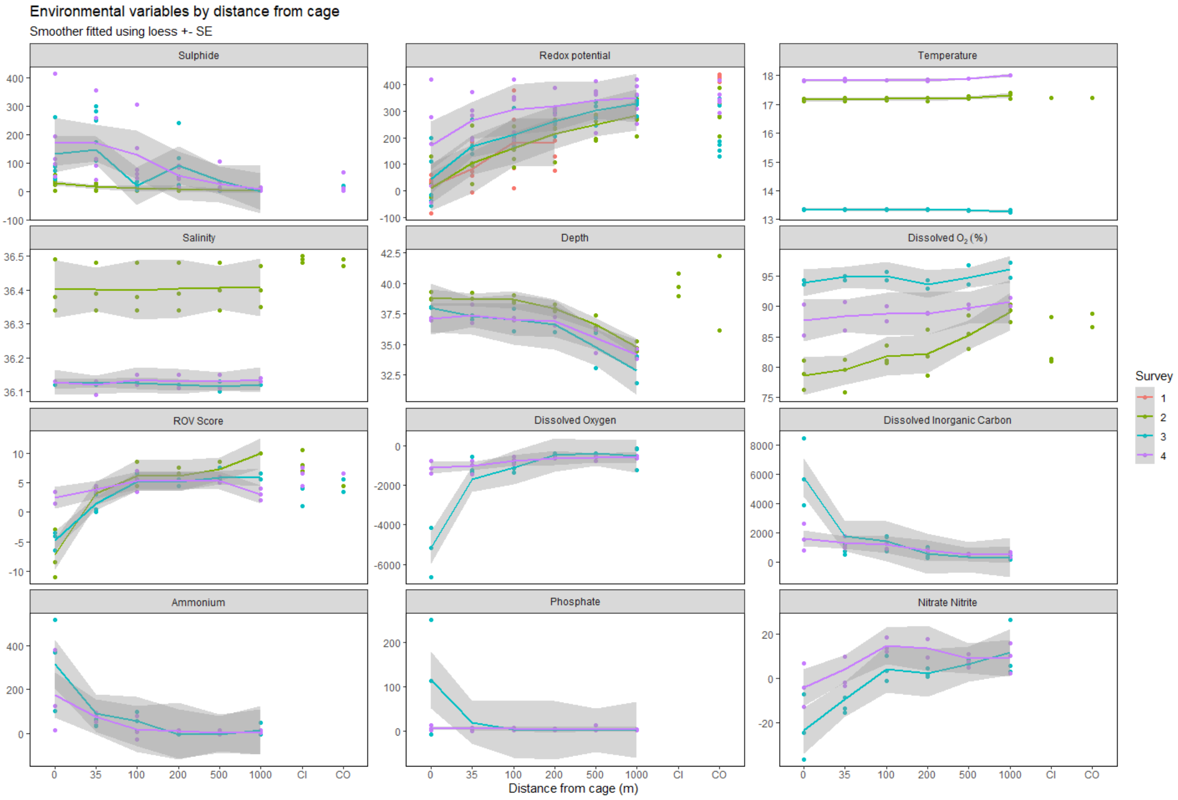
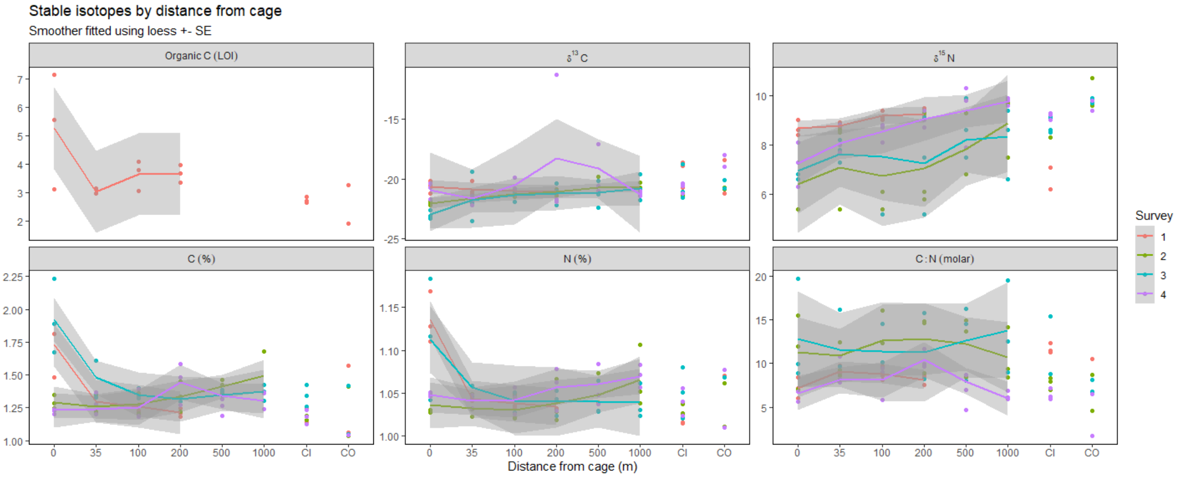
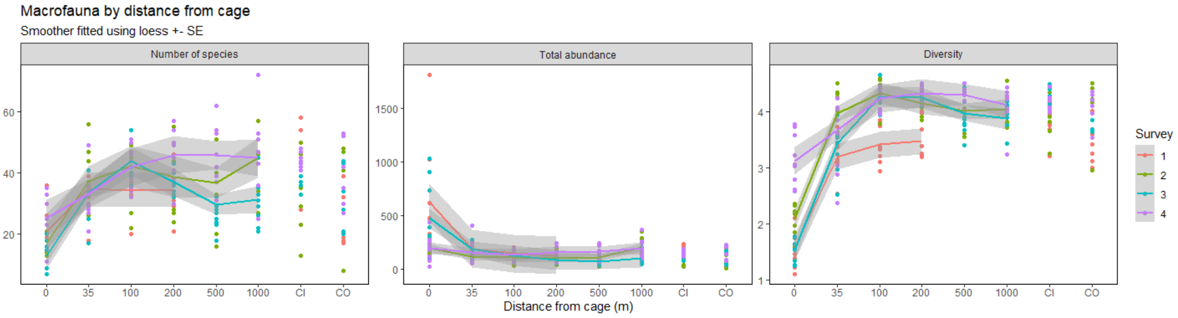
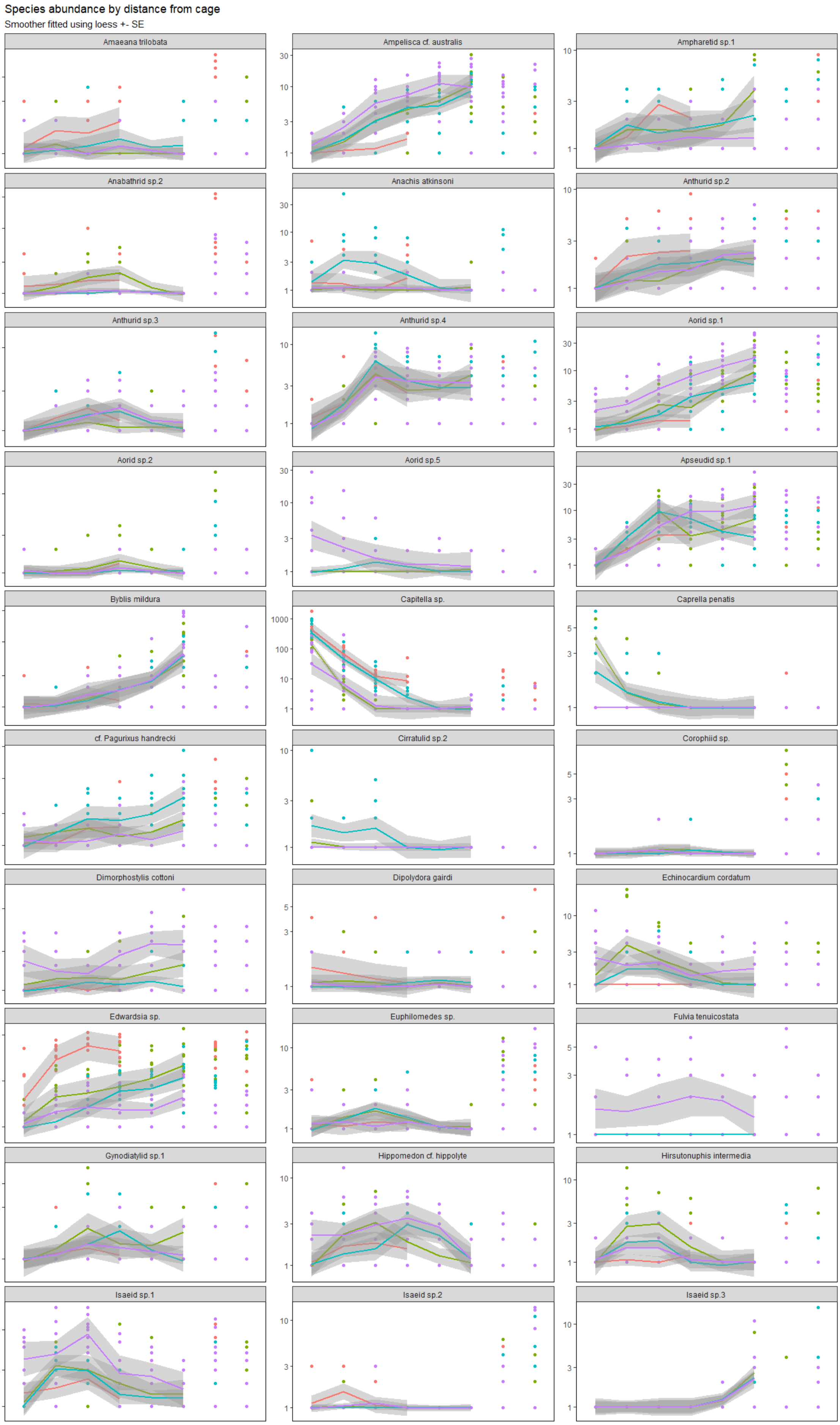
5.3.4 Appendix

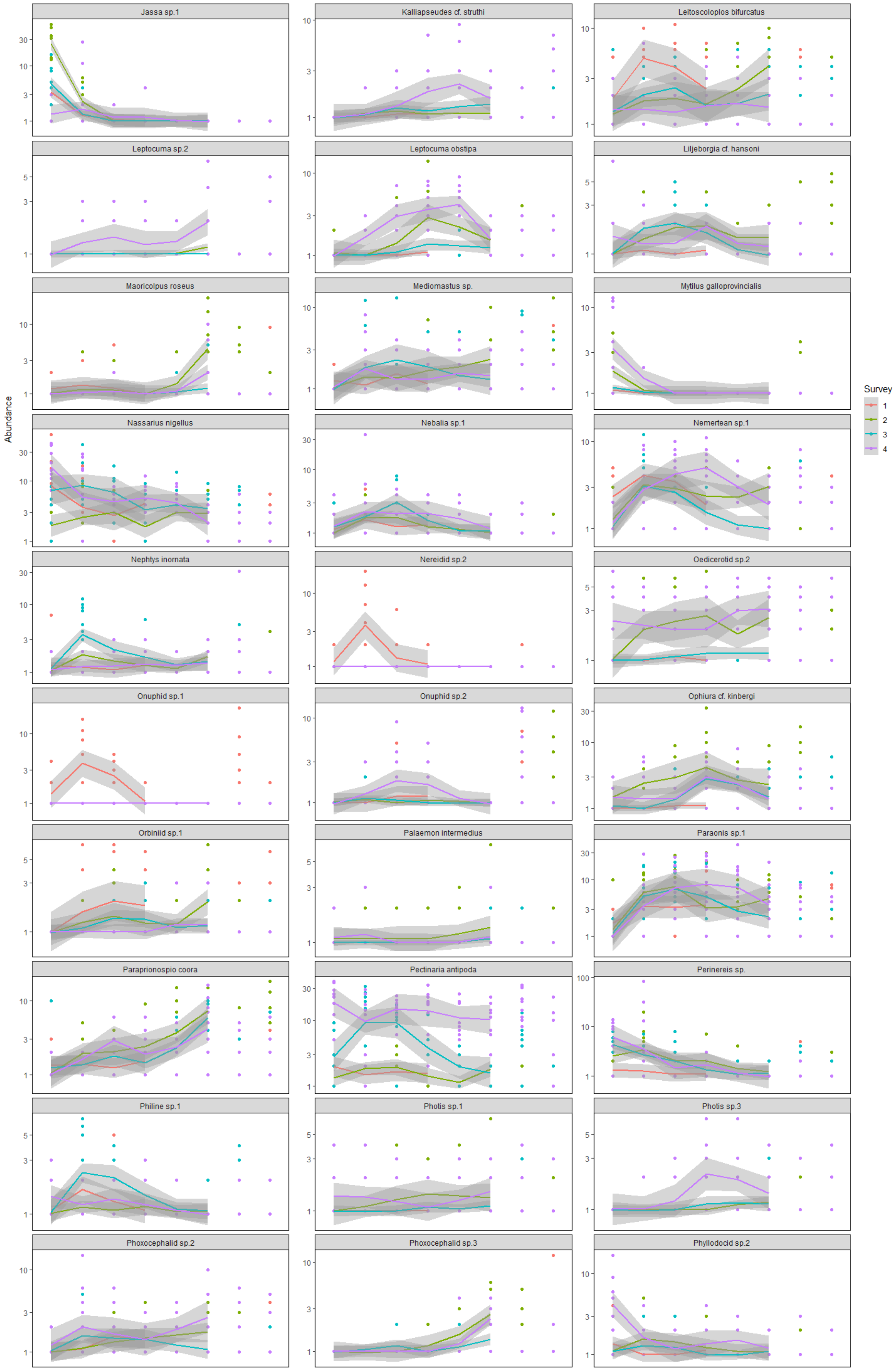
5.3.4.1 Storm Bay 1

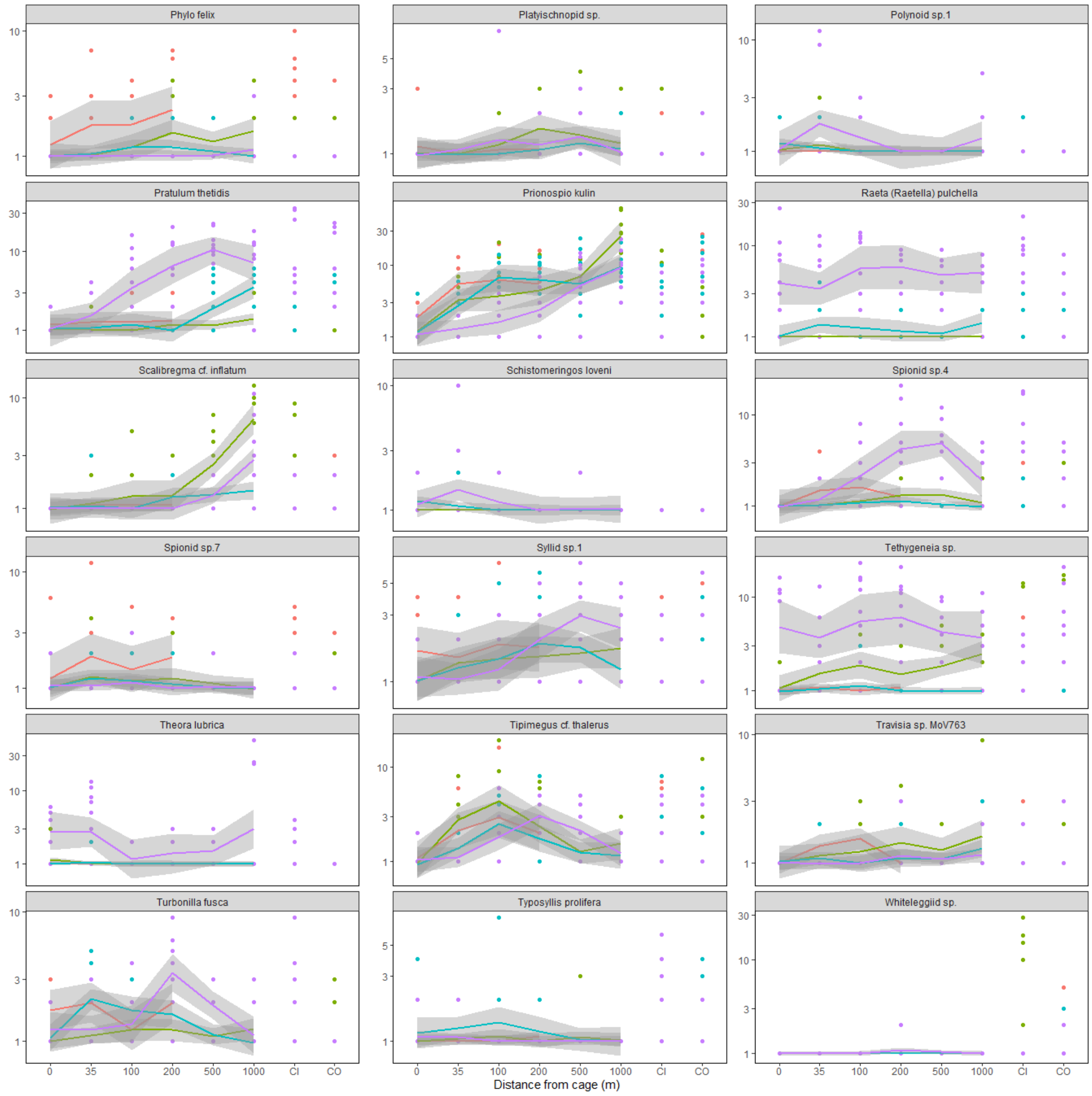




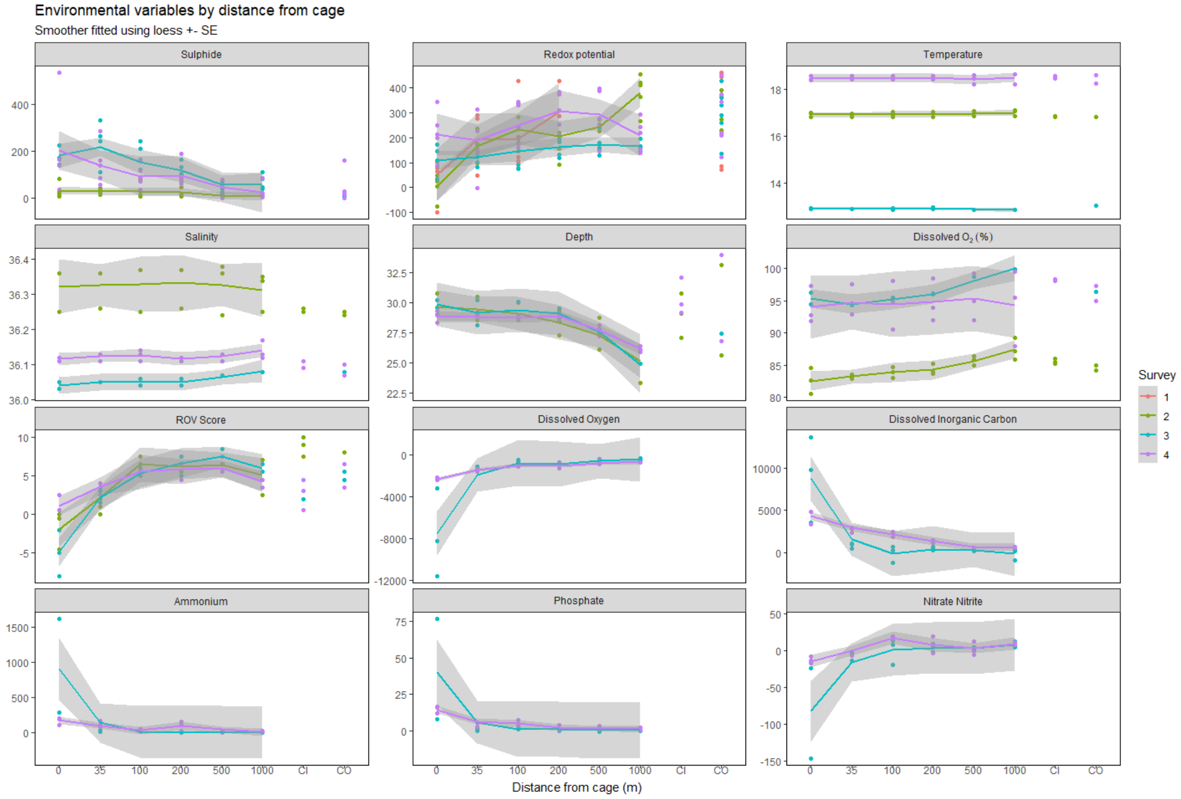


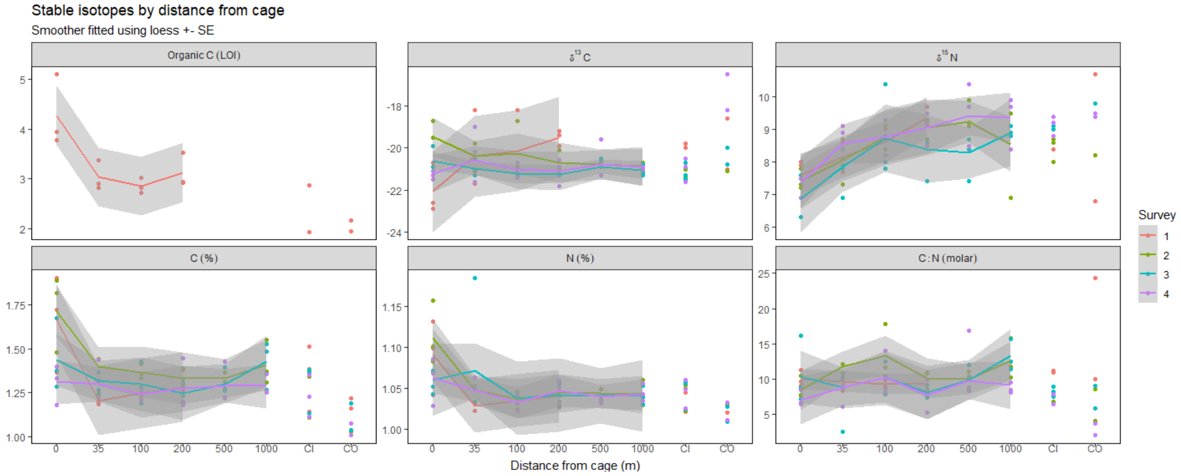

Species abundance by distance from cage
Smoother fitted using loess +- SE

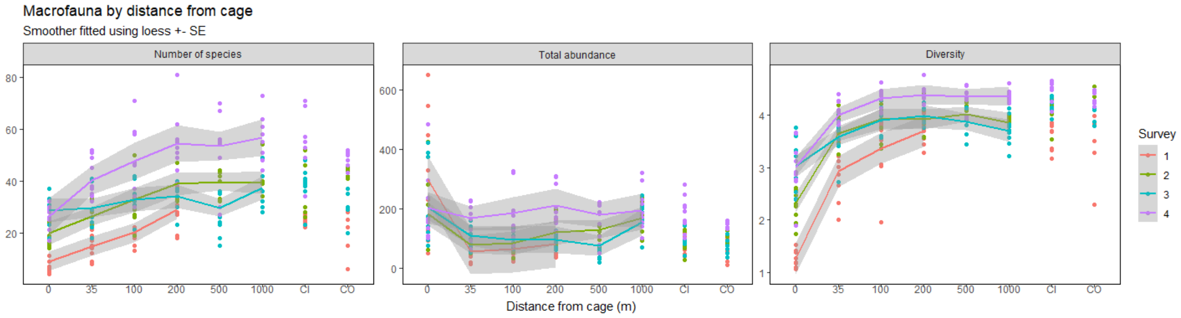

Jassa sp.1
Kalliapseudes cf. struthi
Leitoscoloplos bifurcatus
Leptocuma sp.2
Leptocuma obstipa
Liljeborgia cf. hansoni
Maoricolpus roseus
Mediomastus sp.
Mytilus galloprovincialis
Nassarius nigellus
Nebalia sp.1
Nemertean sp.1
Nephtys inornata
Nereidid sp.2
Oedicerotid sp.2
Onuphid sp.1
Onuphid sp.2
Ophiura cf. kinbergi
Orbiniid sp.1
Palaemon intermedius
Paraonis sp.1
Paraprionospio coora
Pectinaria antipoda
Perinereis sp.
Philine sp.1
Photis sp.1
Photis sp.3
Phoxocephalid sp.2
Phoxocephalid sp.3
Phyllodocid sp.2
Abundance
Survey
1
2
3
4

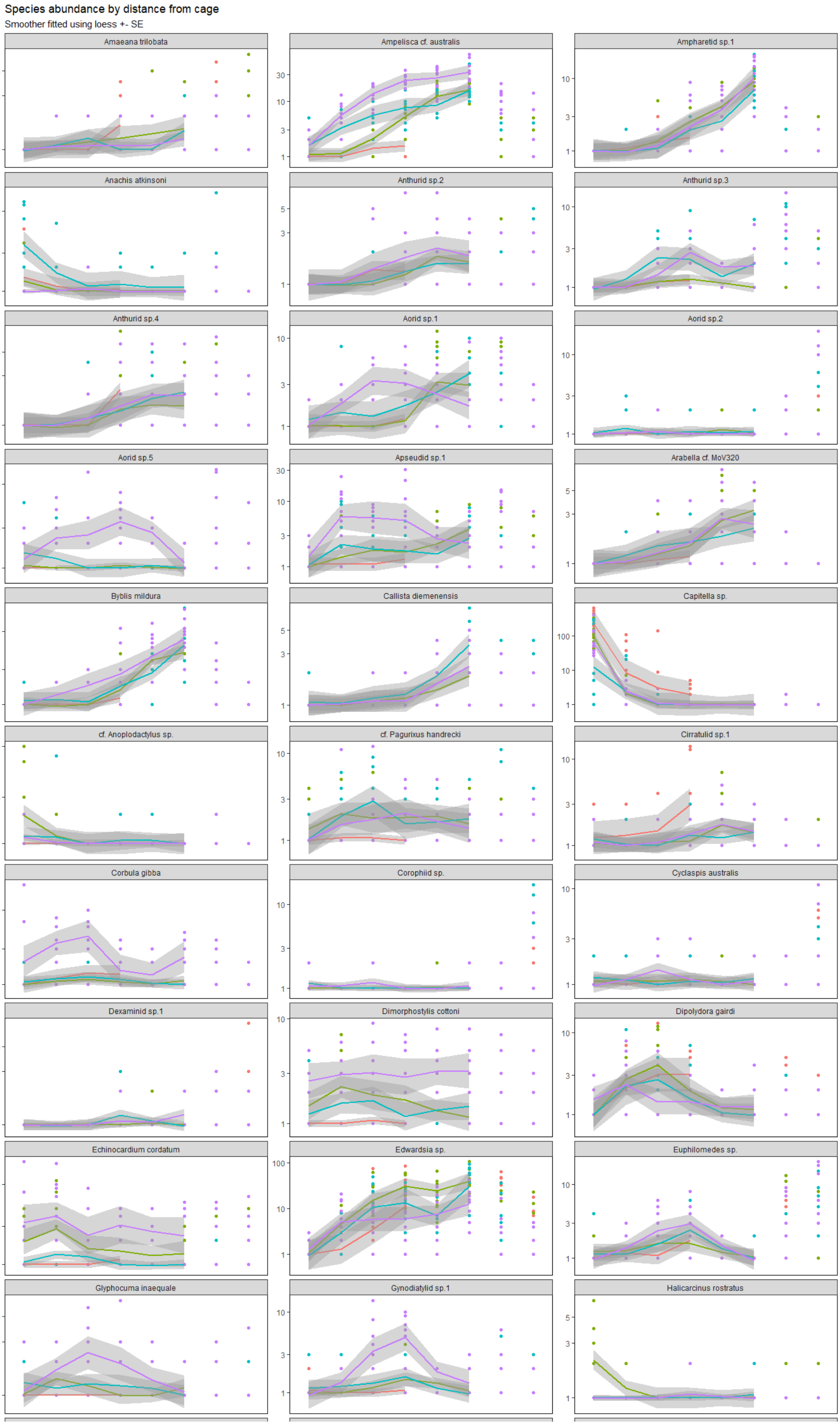

Phylo felix
Platyischnopid sp.
Polynoid sp.1
Pratulum thetidis
Prionospio kulin
Raeta (Raetella) pulchella
Scalibregma cf. inflatum
Schistomeringos loveni
Spionid sp.4
Spionid sp.7
Syllid sp.1
Tethygeneia sp.
Theora lubrica
Tipimegus cf. thalerus
Travisia sp. MoV763
Turbonilla fusca
Typosyllis prolifera
Whiteleggiid sp.
1
3
5
10
30
0
35
100
200
500
1000
CI
CO
Distance from cage (m)

5.3.4.2 Trumpeter



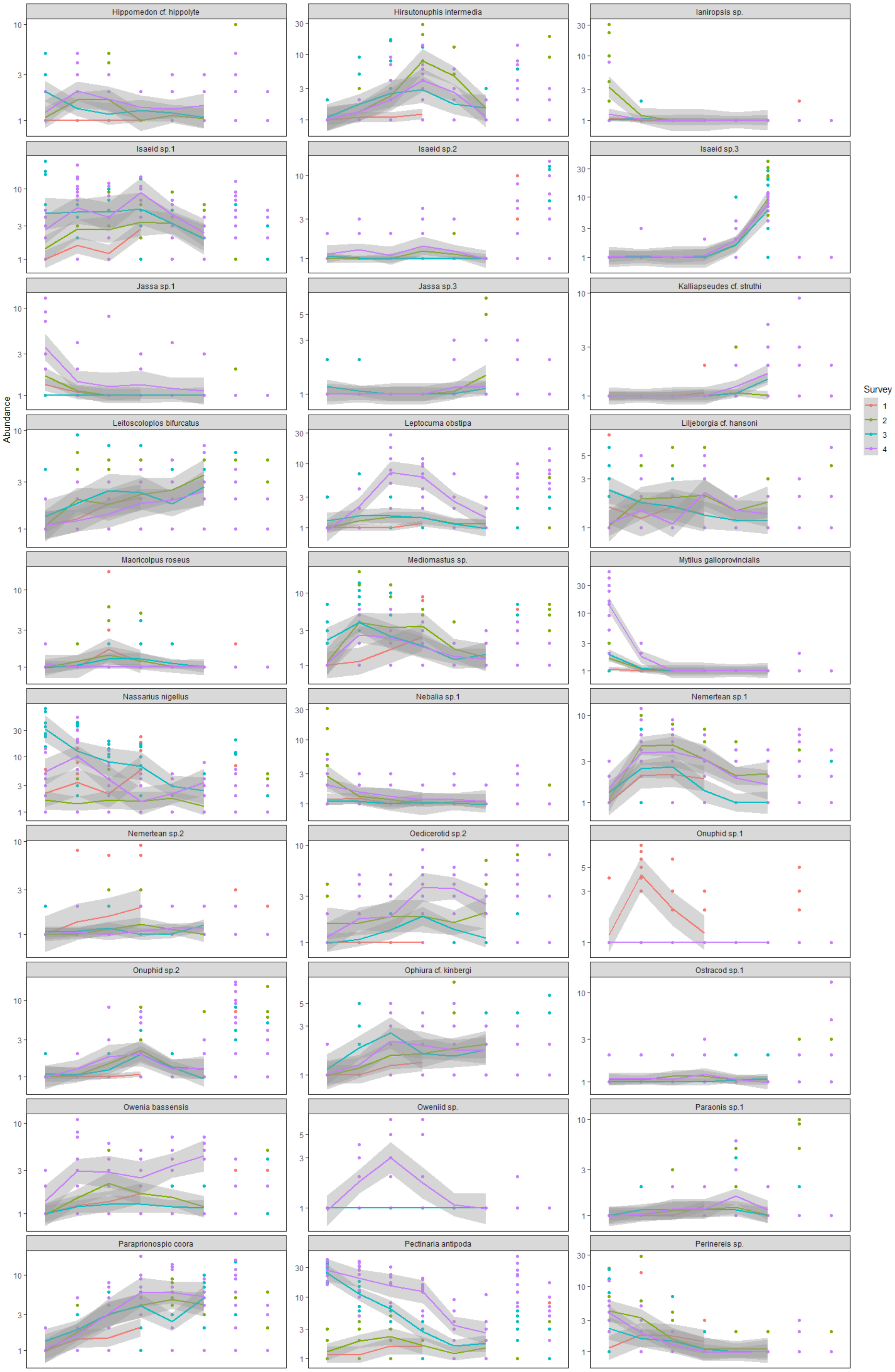


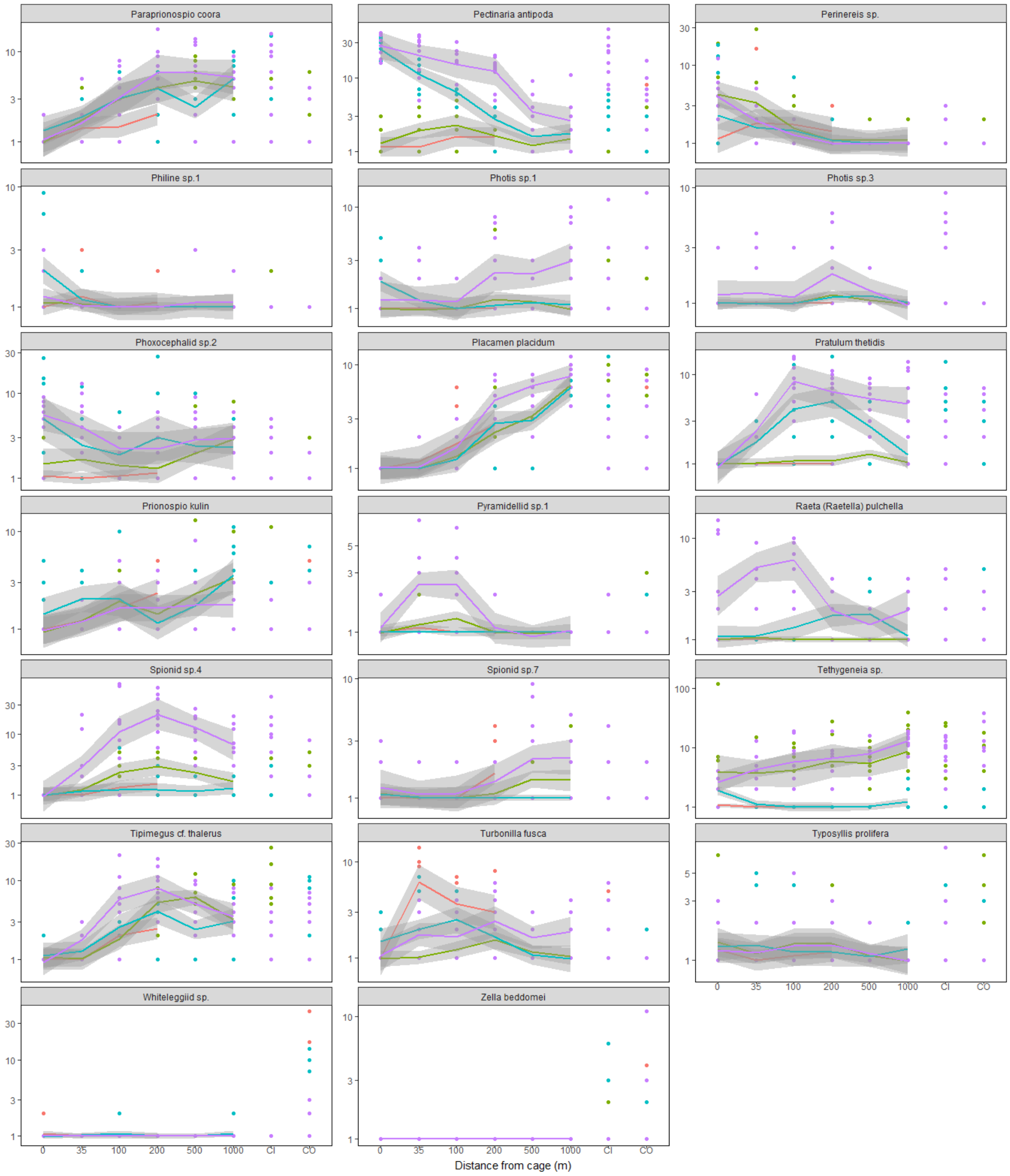




Species abundance by distance from cage

Smoother fitted using loess +- SE


Hippomedon cf. hippolyte
Hirsutonuphis intermedia
Ianiropsis sp.
Isaeid sp.1
Isaeid sp.2
Isaeid sp.3
Jassa sp.1
Jassa sp.3
Kalliapseudes cf. struthi
Leitoscoloplos bifurcatus
Leptocuma obstipa
Liljeborgia cf. hansoni
Maoricolpus roseus
Mediomastus sp.
Mytilus galloprovincialis
Nassarius nigellus
Nebalia sp.1
Nemertean sp.1
Nemertean sp.2
Oedicerotid sp.2
Onuphid sp.1
Onuphid sp.2
Ophiura cf. kinbergi
Ostracod sp.1
Owenia bassensis
Oweniid sp.
Paraonis sp.1
Paraprionospio coora
Pectinaria antipoda
Perinereis sp.
Abundance
1
3
5
10
30
Survey
1
2
3
4


Paraprionospio coora
Pectinaria antipoda
Perinereis sp.
Philine sp.1
Photis sp.1
Photis sp.3
Phoxocephalid sp.2
Placamen placidum
Pratulum thetidis
Prionospio kulin
Pyramidellid sp.1
Raeta (Raetella) pulchella
Spionid sp.4
Spionid sp.7
Tethygeneia sp.
Tipimegus cf. thalerus
Turbonilla fusca
Typosyllis prolifera
Whiteleggiid sp.
Zella beddomei
0
35
100
200
500
1000
CI
CO
1
3
5
10
30
100
Distance from cage (m)

5.4 Macquarie Harbour

5.4.1 Results

5.4.1.1 Physico-chemical

Redox

Table Head Central. Redox potential generally increased with distance but was often highly variable between replicate measurements, sites, and surveys; the effect of distance was only significant in survey 4 (Table 5.4-1). Redox was only measured on one transect, precluding planned comparisons between controls and the 50 m from cage or 100 m (~35 m from the lease boundary) from cage sites; however, there was no evidence of a consistent difference between the control and 50 or 100 m from cages sites across the surveys. Similarly, there was no test for the effect of survey, and Figure 5.4-5 highlights the variability within and between surveys at all distances.

Gordon. There was a clear effect of distance on redox potential at the Gordon lease. Interestingly redox potential was lower in closer proximity to the cages in surveys 1 and 4, but higher in surveys 5 and 6; the effect of distance was significant in all surveys with the exception of survey 5 (Figure 5.4-2, Table 5.4-1). There was no consistent difference between the control and 50 or 100 m sites across the surveys. In surveys 1 and 4 redox was lower at the 50 and 100 m sites compared to the control sites. In survey 5 it was higher at the 50 m site compared to 100 m and the controls, and in survey 6 there was no clear difference between these sites. Redox varied more between surveys at the 0 and 50 m sites compared to the more distant 500 m and control sites (Figure 5.4-5).

Strahan. With the exception of survey 6 when redox potential decreased with distance from the cages there was no clear pattern with distance (Figure 5.4-4, Table 5.4-1). There was no evidence of a consistent difference between the control and 50 or 100 m sites across the surveys. There was perhaps more variability across surveys at the cage site, but otherwise there was no clear difference in the change across surveys at the sites closer to, or more distant from the cages (Figure 5.4-5).

Franklin. Like at the Strahan lease, redox potential was typically negative at all transect and control sites. Apart from survey 1, redox potential was markedly lower at the cage site compared to the other distances, but the effect of distance was only significant in survey 4 (Figure 5.4-3, Table 5.4-1). There was no evidence of a consistent difference between the control and 50 or 100 m sites across the surveys. Consistent with the other leases, Figure 5.4-5 highlights the variability within and between surveys at all distances, with the most notable change occurring at the cage site between the survey 1 and the subsequent surveys.

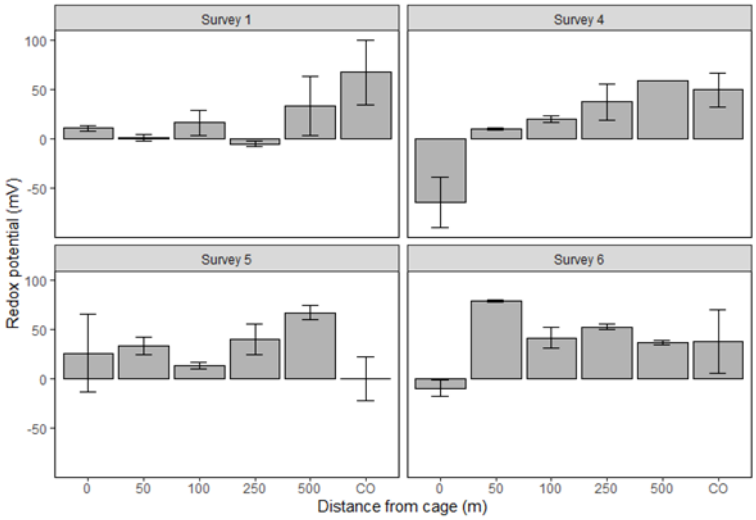


Figure 5.4-1 Average (±SE) redox potential (mV) at 3cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease. Redox was not measured at the Table Head Central lease in surveys 2 and 3.

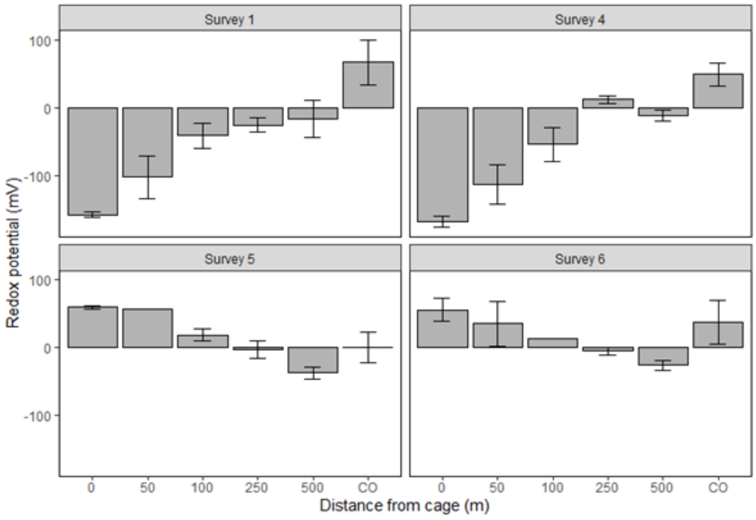


Figure 5.4-2 Average (±SE) redox potential (mV) at 3cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease. Redox was not measured at the Strahan lease in surveys 2 and 3.

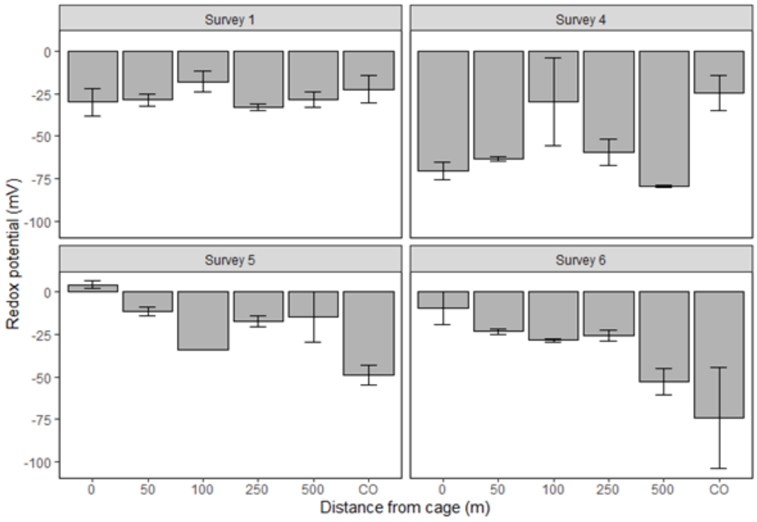


Figure 5.4-3 Average (±SE) redox potential (mV) at 3cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Redox was not measured at the Strahan lease in surveys 2 and 3.

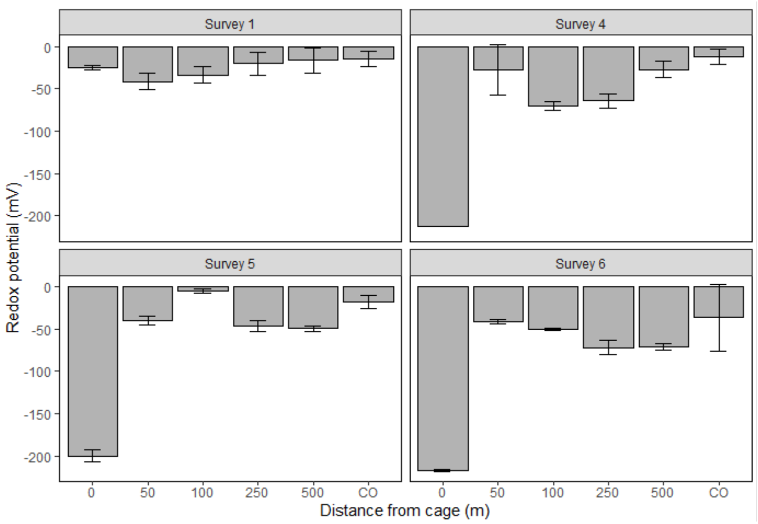


Figure 5.4-4 Average (±SE) redox potential (mV) at 3cm depth in sediment with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease. Redox was not measured at the Franklin lease in surveys 2 and 3.

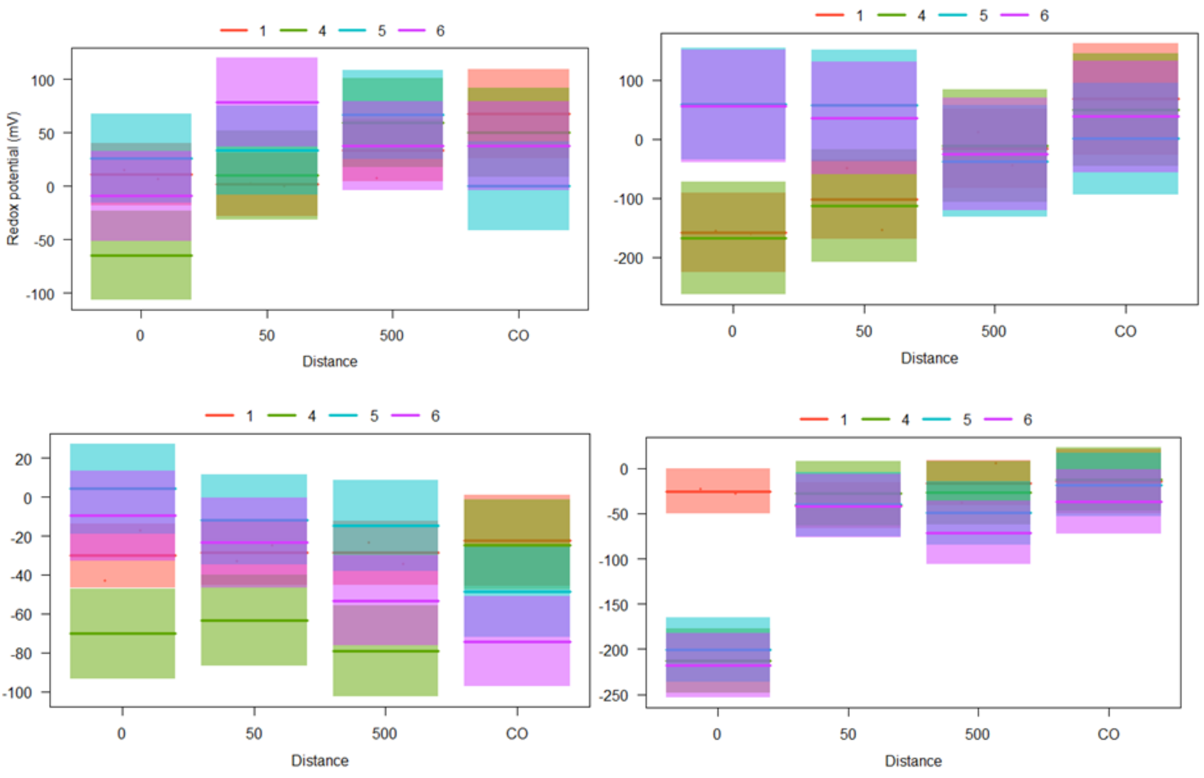


Figure 5.4-5 Redox (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Organic carbon and nitrogen and stable isotopes

Organic carbon content (%)

Table Head Central. Although organic carbon content (%) was often higher at intermediate distances on the transect (50 - 250 m) there was no clear or consistent effect of distance in any of the surveys (Figure 5.4-6, Table 5.4-1). Organic carbon content (%) was often lower at the control compared to the 50 or 100 m sites, but the differences were not significant due to the variation between replicates. There was no clear survey effect on organic carbon content at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-10, Table 5.4-2).

Gordon. In surveys 1 and 2, organic carbon content (%) was higher at the cage site, but there was no clear or constant effect of distance across the surveys (Figure 5.4-7, Table 5.4-1). Organic carbon content (%) was often lower at the control compared to the 50 or 100 m sites; this difference was significant for both 50 and 100 m in surveys 5 and 6 and 100 m in survey 4. There was a significant effect of survey at the sites closest to the cages (0 and 50 m) but not at the more distant sites (500 m and CO; Figure 5.4-10, Table 5.4-2).

Strahan. There was no clear or constant effect of distance on organic carbon content (%) across the surveys (Figure 5.4-8, Table 5.4-1). There was also no clear or consistent difference between the control and 50 or 100 m sites in any of the surveys. There was a significant effect of survey at the 500 m but not that the other sites (500 m and CO; Figure 5.4-10, Table 5.4-2).

Franklin. Although there does appear to be a trend of increasing organic carbon content (%) in closer proximity to the cages, the effect of distance was not significant in any survey (Figure 5.4-9, Table 5.4-1). In four of the five surveys, organic carbon content (%) was lower at the control compared to the 50 and 100 m sites and this difference was significant for both 50 and 100 m in surveys 5 and 6 and 100 m in survey 2. There was a significant effect of survey at the sites closest to the cages (0 and 50 m) but not at the more distant sites (500 m and CO; Figure 5.4-10, Table 5.4-2).

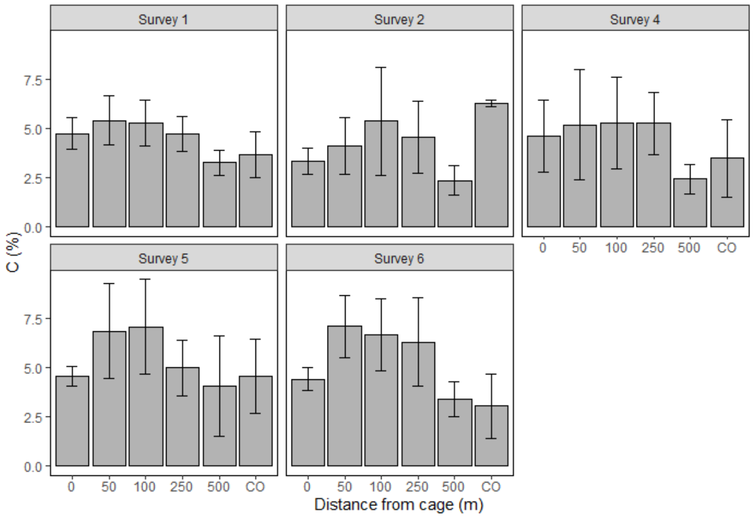


Figure 5.4-6 Average (±SE) of percent carbon (%C) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

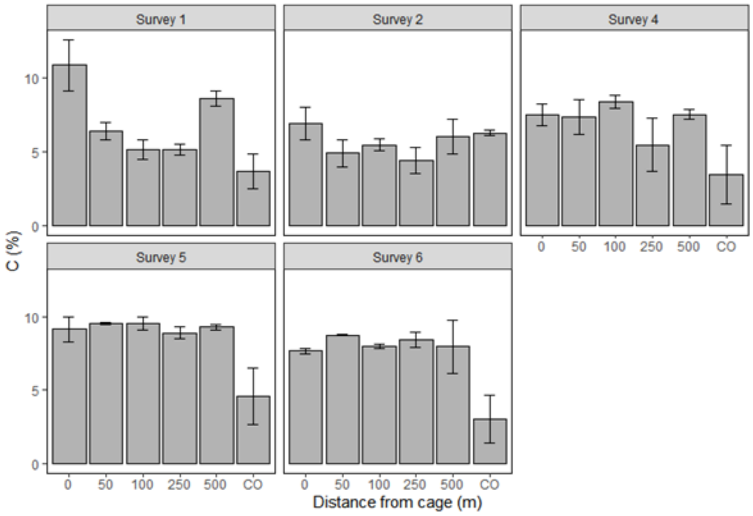


Figure 5.4-7 Average (±SE) of percent carbon (%C) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

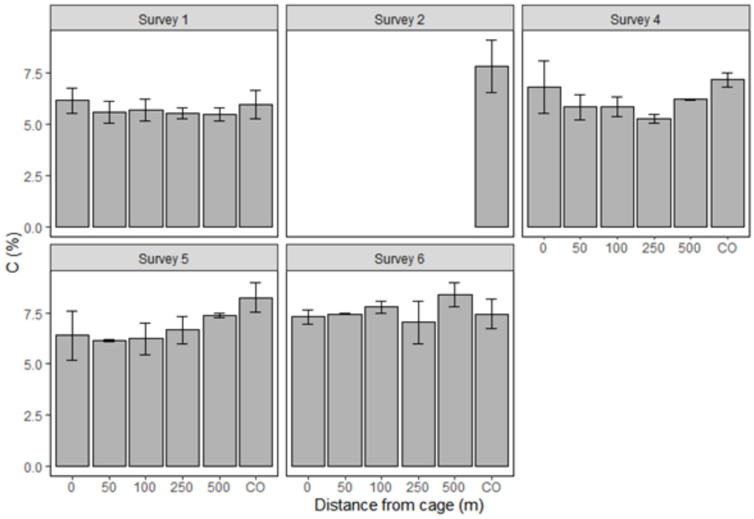


Figure 5.4-8 Average (±SE) of percent carbon (%C) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

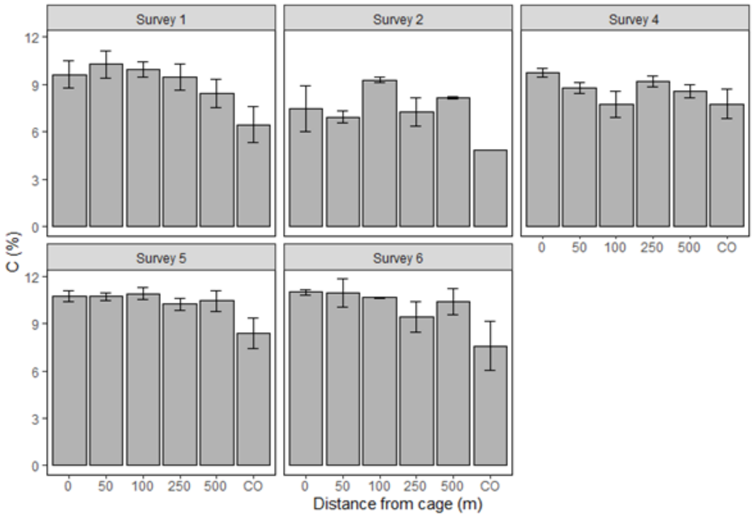


Figure 5.4-9 Average (±SE) of percent carbon (%C) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

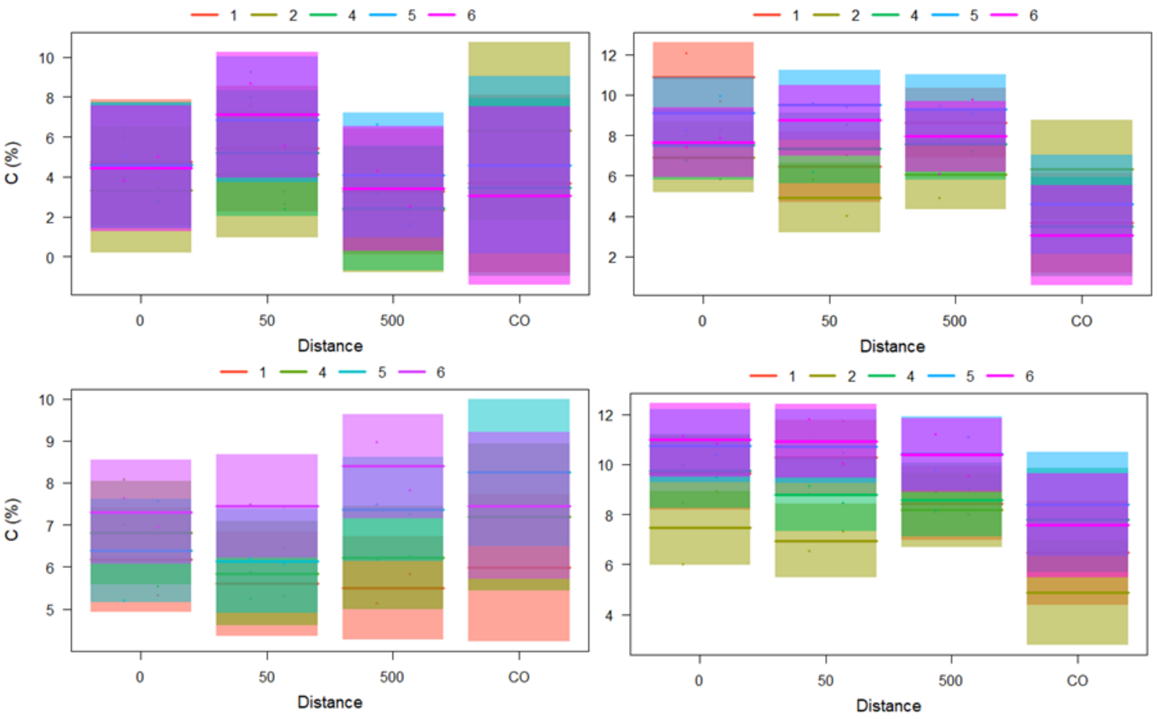


Figure 5.4-10 Percent carbon (%C) (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Organic nitrogen content (%)

Table Head Central. Although organic nitrogen content (%) was often higher at sites in closer proximity to the cages it was quite variable between replicate measurements, sites and surveys; the effect of distance was not significant in any of the surveys (Figure 5.4-11, Table 5.4-1). Similarly, organic nitrogen content (%) was often lower at the control compared to the 50 or 100 m sites, but the differences were not significant. There was no evidence of a survey effect at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-15, Table 5.4-2).

Gordon. There was a very clear effect of distance in organic nitrogen content (%) with higher values at the cage site; the effect of distance was significant in each of the surveys (Figure 5.4-12, Table 5.4-1). Although organic nitrogen content (%) was typically lower at the control compared to the 50 or 100 m from cages sites, the differences were not significant in any survey. There was also no evidence of a survey effect at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-15, Table 5.4-2).

Strahan. Organic nitrogen content (%) was typically higher at the cage site and the effect of distance was significant in surveys 4 and 5 (Figure 5.4-13, Table 5.4-1). There was also no clear or consistent difference between the control and 50 or 100 m sites in any of the surveys. There was no evidence of a survey effect at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-15, Table 5.4-2).

Franklin. Organic nitrogen content (%) was higher in closer proximity to the cages in surveys 4, 5 and 6, and the effect of distance was significant (or very close) in each of these surveys (Figure 5.4-14, Table 5.4-1). Although organic nitrogen content (%) was typically lower at

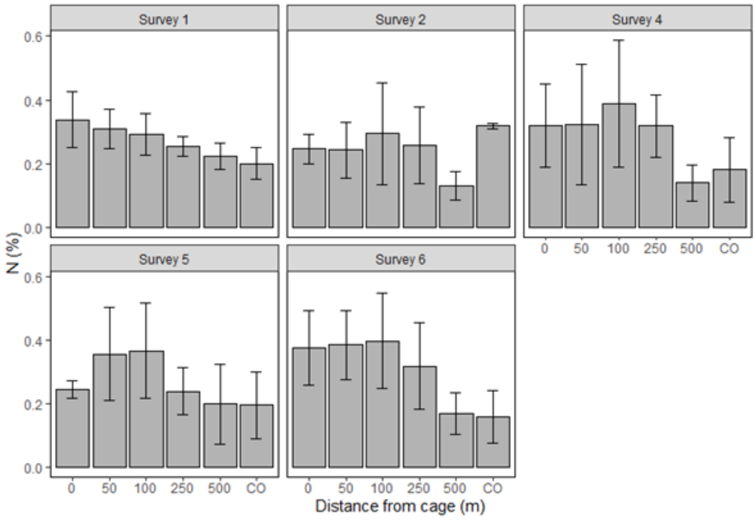


Figure 5.4-11 Average (±SE) of percent nitrogen (%N) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

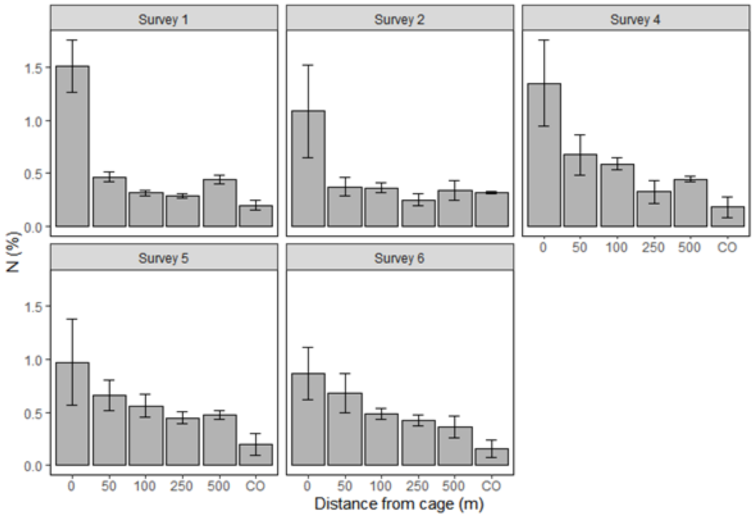


Figure 5.4-12 Average (±SE) of percent nitrogen (%N) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

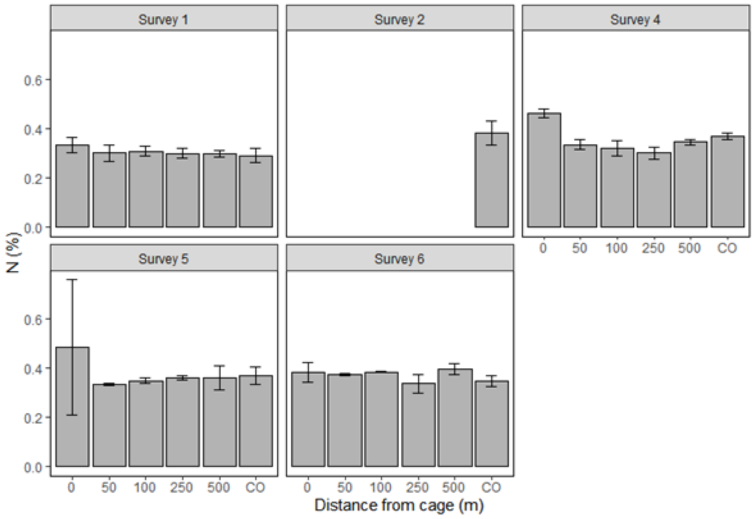


Figure 5.4-13 Average (±SE) of percent nitrogen (%N) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

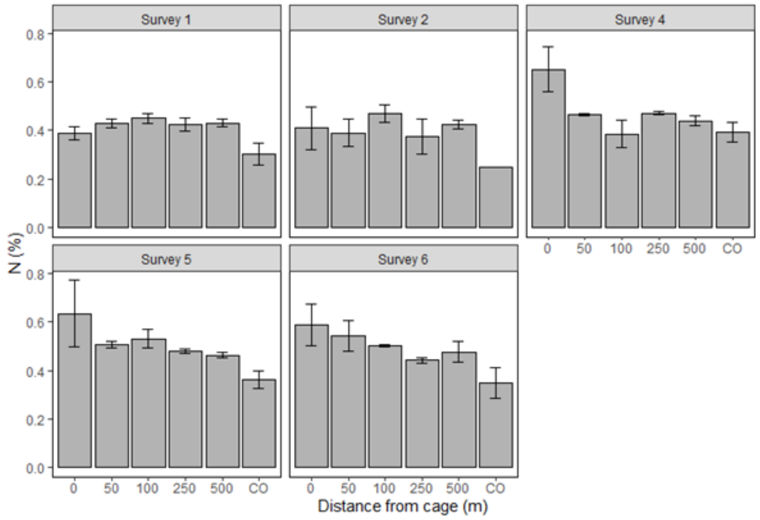


Figure 5.4-14 Average (±SE) of percent nitrogen (%N) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

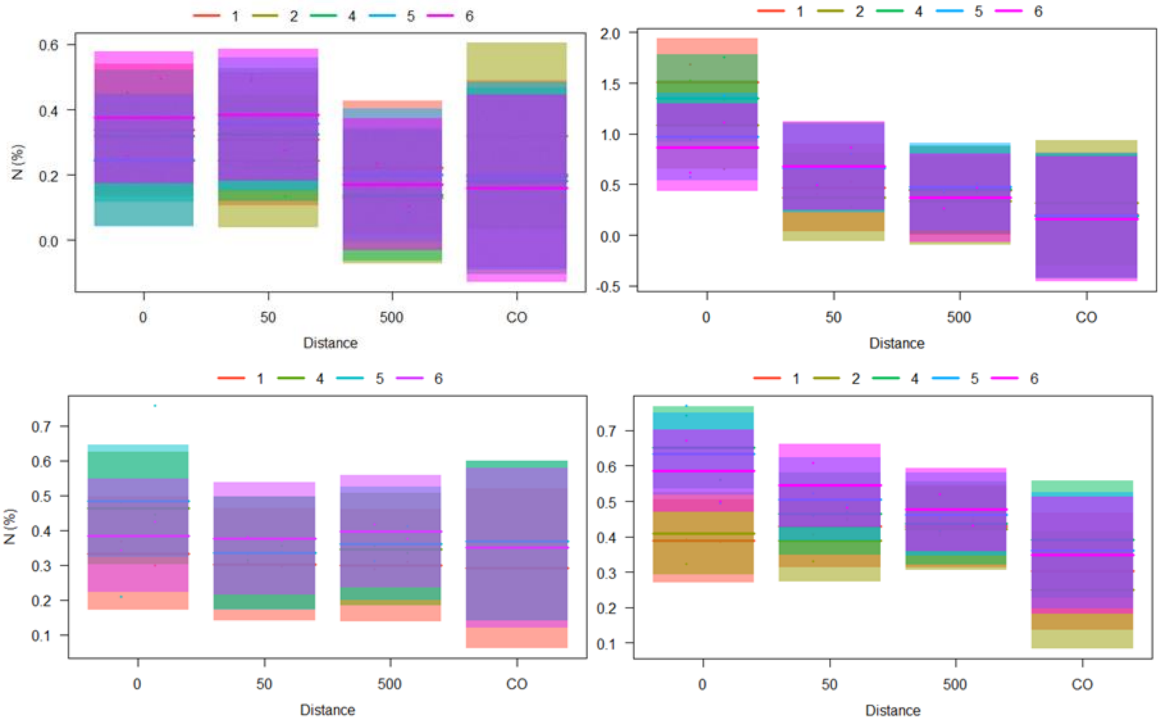


Figure 5.4-15 Percent nitrogen (%N) (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

the control compared to the 50 or 100 m from cages sites, the differences were only significant in survey 1. There was a significant effect of survey at the 0 m cage site but not at 50 m or more distant sites (500 m and CO; Figure 5.4-15, Table 5.4-2).

C:N molar ratio

Table Head Central. There was a trend of an increase in the C:N ratio of the sediments with distance in all five surveys, but the effect of distance was only significant in surveys 2 and 6 (Figure 5.4-16, Table 5.4-1). Although the C:N ratio of the sediments was often higher at the control sites compared to the 50 and 100 m sites, the differences were only significant in survey 5. There was a significant effect of survey at the 0 m, 500 m and the control site but not at 50 m (Figure 5.4-20, Table 5.4-2).

Gordon. There was a very clear effect of distance in the C:N ratio of the sediments with values increasing with distance from the cage; the effect of distance was significant in each of the surveys (Figure 5.4-17, Table 5.4-1). The C:N ratio of the sediments was higher at the control compared to the 50 or 100 m sites, but this difference was only significant at 50 m in surveys 4 and 5. There was no evidence of a survey effect at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-20, Table 5.4-2).

Strahan. The change in the C:N ratio of the sediments was less evident at Strahan, but the ratios were still typically lower in closer proximity to the cages (Figure 5.4-18). This was most evident in survey 4 when the effect of distance was significant (Table 5.4-1). The C:N ratio of the sediments was higher at the control compared to the 50 or 100 m sites, but the differences were not significant in any of the surveys. There was no evidence of a survey effect at the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-20, Table 5.4-2).

Franklin. In survey 1 the C:N ratio of the sediment decreased with distance from the cages, opposite to the trend observed at the other leases and in subsequent surveys at Franklin (Figure 5.4-19, Table 5.4-1). In surveys 4 and 5 the effect of distance was significant, with lower C:N ratios measured in closer proximity to the cages (Table 5.4-1). The comparison between the control and 50 and 100 m sites was not significant in any of the surveys. There was a significant effect of survey at the 0 m cage site but not at 50 m or more distant sites (500 m and CO; Figure 5.4-20, Table 5.4-2). This most likely reflects the markedly higher C:N ratio of the sediments at the 0 m cage site in survey 1 relative to the other surveys.

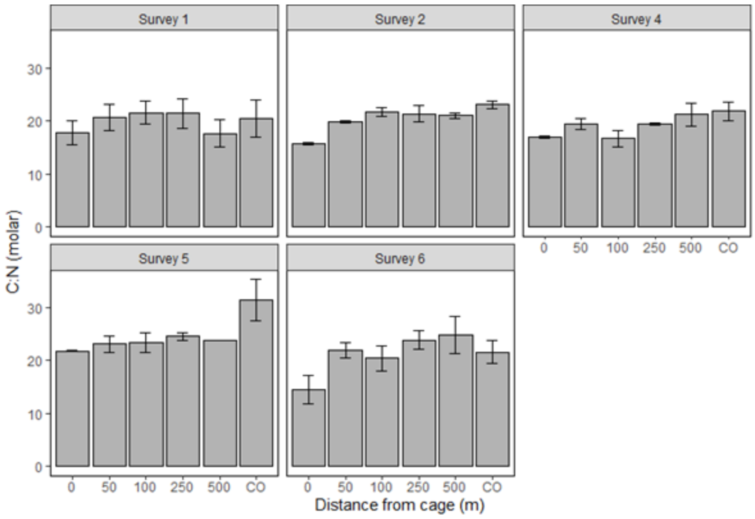


Figure 5.4-16 Average (±SE) C:N molar ratio with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

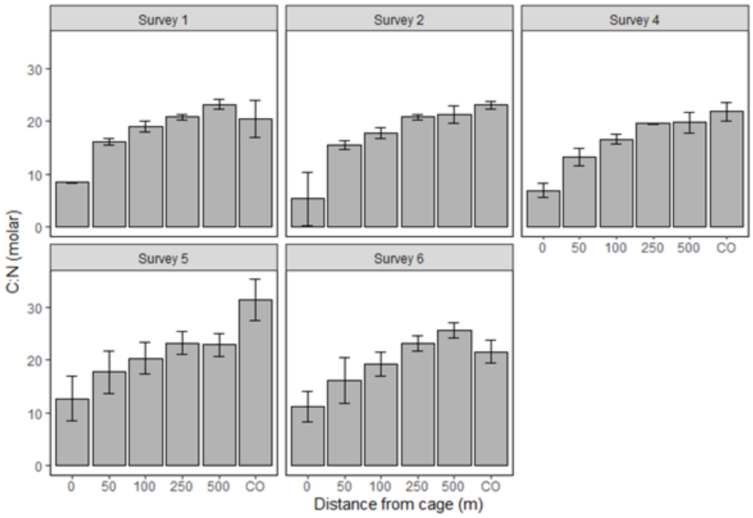


Figure 5.4-17 Average (±SE) C:N molar ratio with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

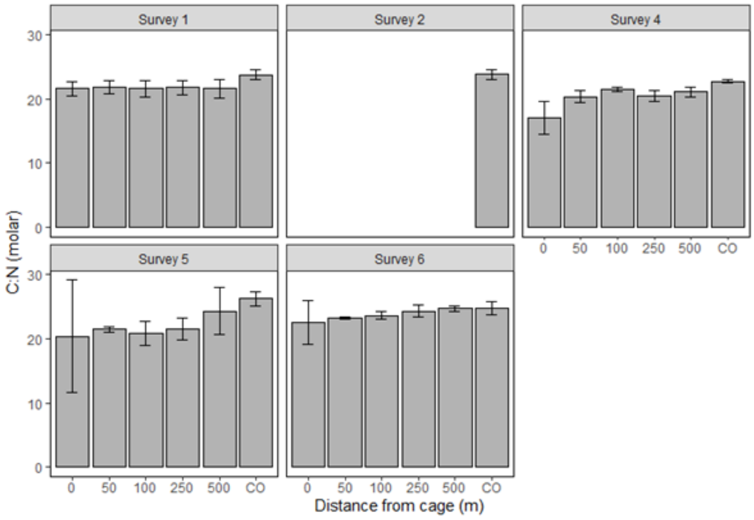


Figure 5.4-18 Average (±SE) of C:N molar ratio with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Data not available for Survey 2.

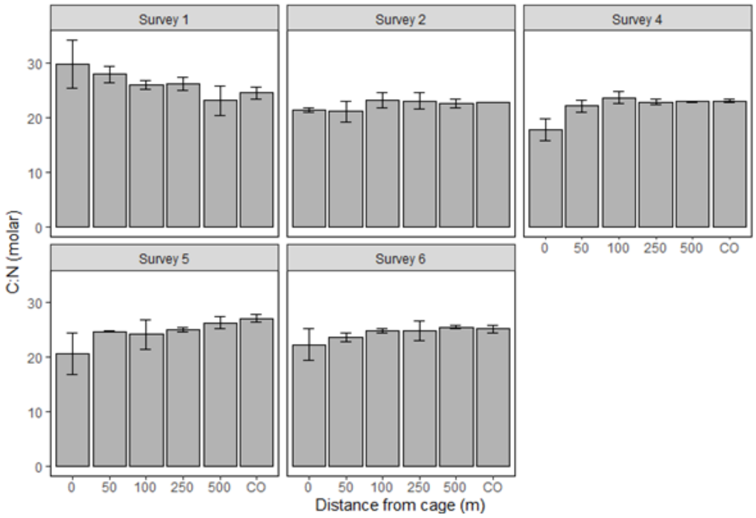


Figure 5.4-19 Average (±SE) C:N molar ratio with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

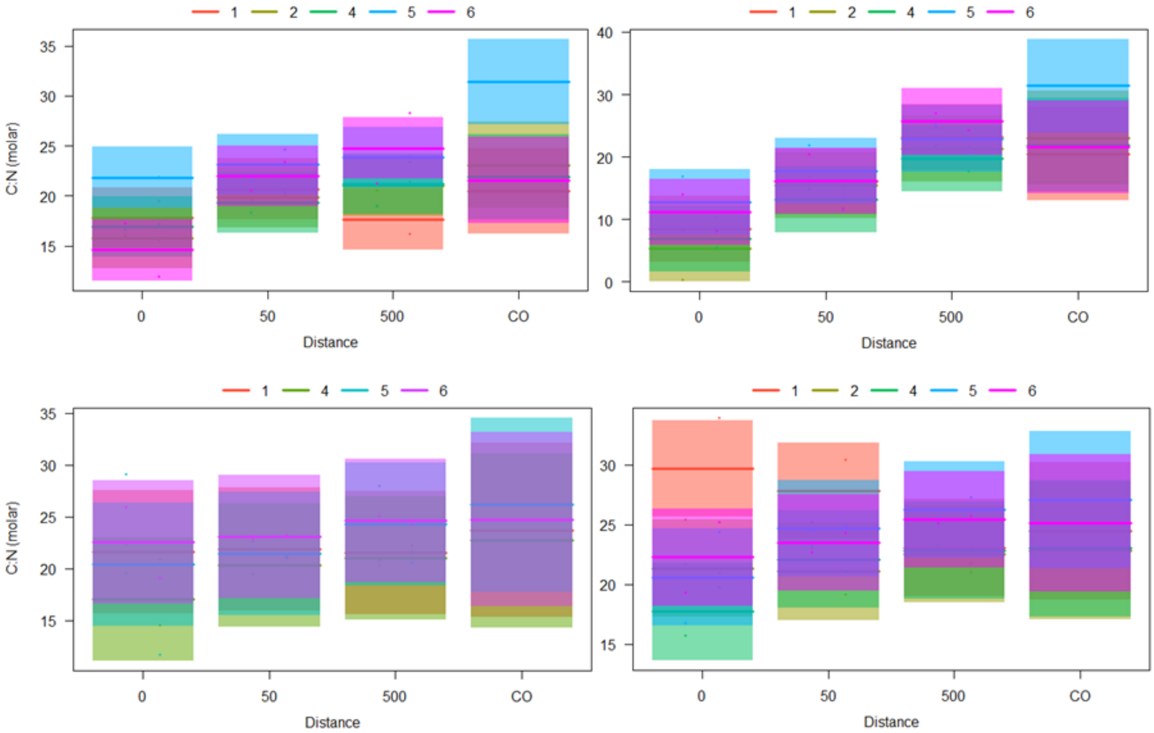


Figure 5.4-20 C:N molar ratio (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Carbon (δ^{13}C)

Table Head Central. The δ^{13}C signature of the sediments was more depleted closer to the cage with the effect of distance significant in surveys 1, 4, 5 and 6, but not in survey 2 (Figure 5.4-21, Table 5.4-1). With the exception of the 50 m site in survey 4, there was no significant difference in δ^{13}C between the 50 and 100 m sites and the controls in any of the surveys. There was a significant effect of survey at both the sites closest to the cages (0 and 50 m) and the more distant sites (500 m and CO; Figure 5.4-25, Table 5.4-2), reflecting the lower values recorded at all sites throughout the harbour in survey 4.

Gordon. The δ^{13}C signature of the sediments was more depleted closer to the cage; the effect of distance was significant in surveys 1, 2, 4 and 6, but not in survey 5 (Figure 5.4-22, Table 5.4-1). However, there was no significant difference in δ^{13}C between the 50 and 100 m sites and the controls in any of the surveys. There was a significant effect of survey at both the sites closest to the cages (0 and 50 m) and the more distant sites (500 m and CO; Figure 5.4-20, Table 5.4-2), reflecting the lower values recorded at all sites throughout the harbour in survey 4.

Strahan. The trend of a more depleted δ^{13}C signature of the sediments closer to the cage was not as evident at Strahan, but the effect of distance was significant in surveys 4 and 6 (Figure 5.4-23, Table 5.4-1). However, there was no significant difference in δ^{13}C between the 50 and 100 m sites and the controls in any of the surveys. There was a significant effect of survey at both the sites closest to the cages (0 and 50 m) and the more distant sites (500 m and CO; Figure 5.4-25, Table 5.4-2), reflecting the lower values recorded at all sites throughout the harbour in survey 4.

Franklin. The trend of a more depleted δ^{13}C signature of the sediments closer to the cage was not evident at Franklin, with the effect of distance only significant in survey 6 (Figure 5.4-24, Table 5.4-1). In surveys 1 and 2, δ^{13}C was significantly higher at the control site compared to both the 50 and 100 m sites. In all other cases the difference was not significant (Table 5.4-1). There was a significant effect of survey at both the sites closest to the cages (0 and 50 m) and the more distant sites (500 m and CO; Figure 5.4-25, Table 5.4-2), reflecting the lower values recorded at all sites throughout the harbour in survey 4.

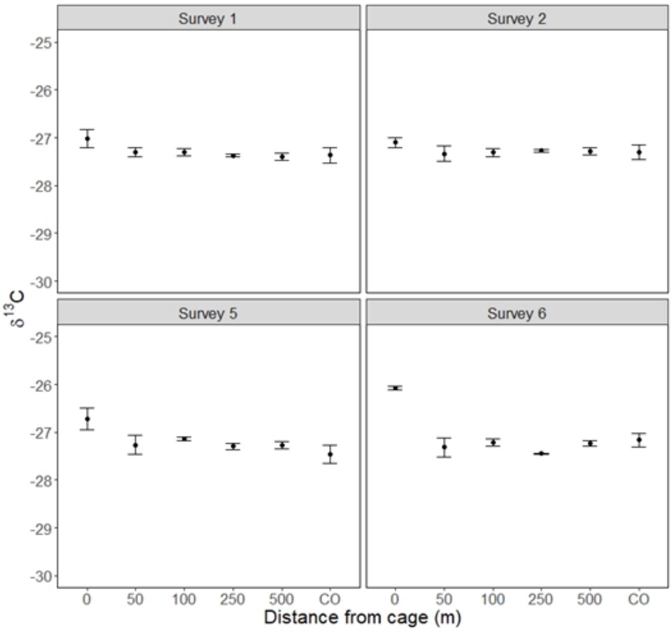


Figure 5.4-21 Carbon ($\delta^{13}C$) stable isotope values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease. Data not available for surveys 3 and 4.

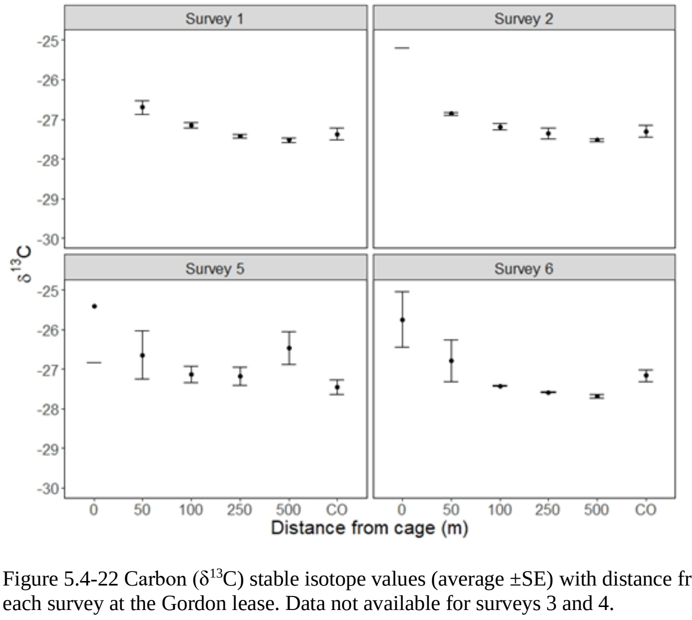


Figure 5.4-22 Carbon ($\delta^{13}C$) stable isotope values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease. Data not available for surveys 3 and 4.

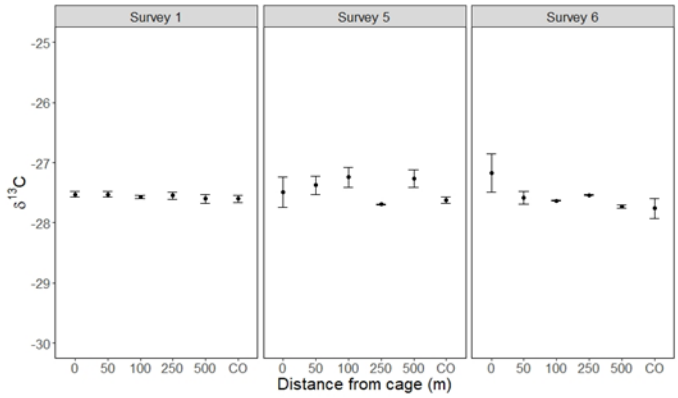


Figure 5.4-23 Carbon ($\delta^{13}C$) stable isotope values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Data not available for surveys 2, 3 and 4.

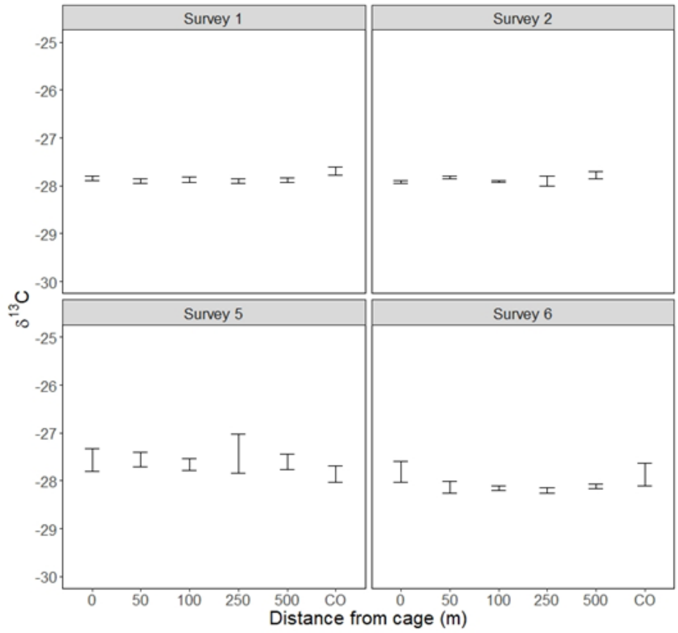


Figure 5.4-24 Carbon ($\delta^{13}C$) stable isotope values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease. Data not available for surveys 3 and 4.

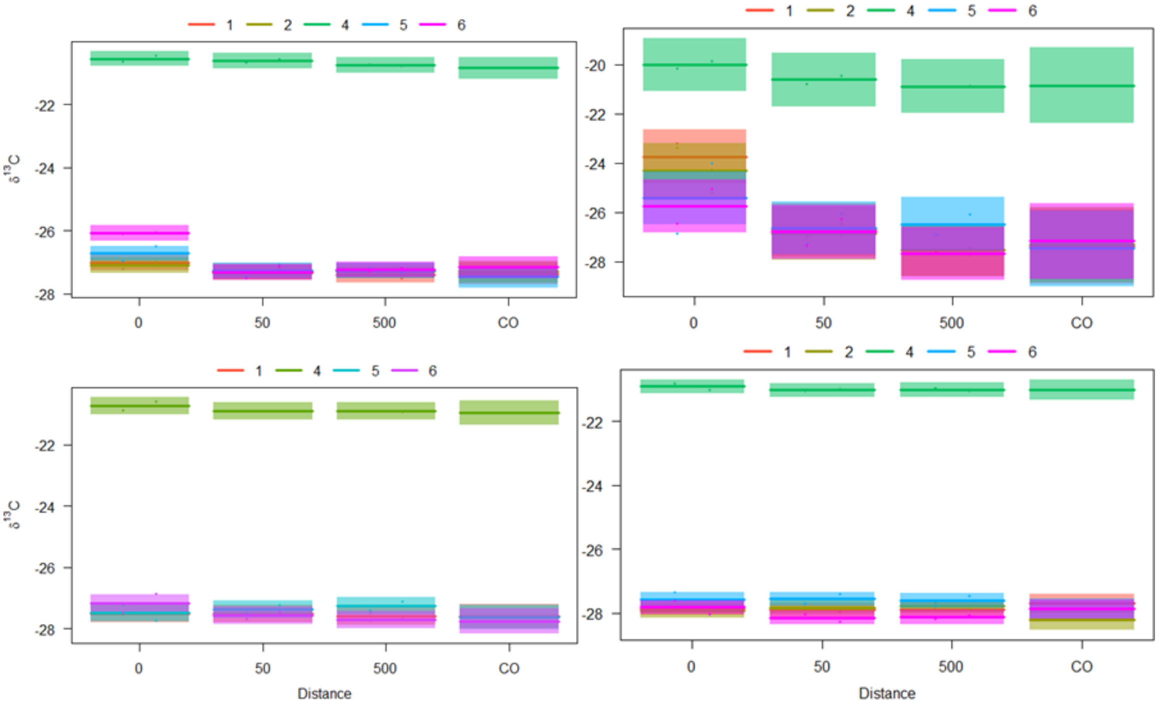


Figure 5.4-25 Carbon ($\delta^{13}C$) stable isotopes value (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Nitrogen ($\delta^{15}N$)

Table Head Central. The more enriched $\delta^{15}N$ signature of the sediments at the cage site was most evident in surveys 2 and 6 and reflected in the statistical tests (Figure 5.4-26, Table 5.4-1). However, there was no significant difference in $\delta^{15}N$ between the 50 and 100 m sites and the controls in any of the surveys. There was also no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-30, Table 5.4-2).

Gordon. There was a very clear effect of distance in the $\delta^{15}N$ signature of the sediments with more enriched values closer to the cage; the effect of distance was significant in each of the surveys (Figure 5.4-27, Table 5.4-1). Although the $\delta^{15}N$ signature of the sediments was often more enriched at the 50 or 100 m sites compared to the controls, the differences were not significant in any of the surveys. There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-30, Table 5.4-2).

Strahan. A more enriched $\delta^{15}N$ signature of the sediments at sites closer to the cage was evident in all four surveys, but the effect of distance was only significant in surveys 4 and 5 (Figure 5.4-28, Table 5.4-1). There was no significant difference in $\delta^{15}N$ between the 50 and 100 m sites and the controls in any of the surveys and there was also no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-30, Table 5.4-2).

Franklin. A more enriched $\delta^{15}N$ signature of the sediments at sites closer to the cage was evident in all five surveys, but the effect of distance was only significant in surveys 4 and 6 (Figure 5.4-29, Table 5.4-1). There was no significant difference in $\delta^{15}N$ between the 50 and 100 m sites and the controls in any of the surveys and there was also no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-30, Table 5.4-2).

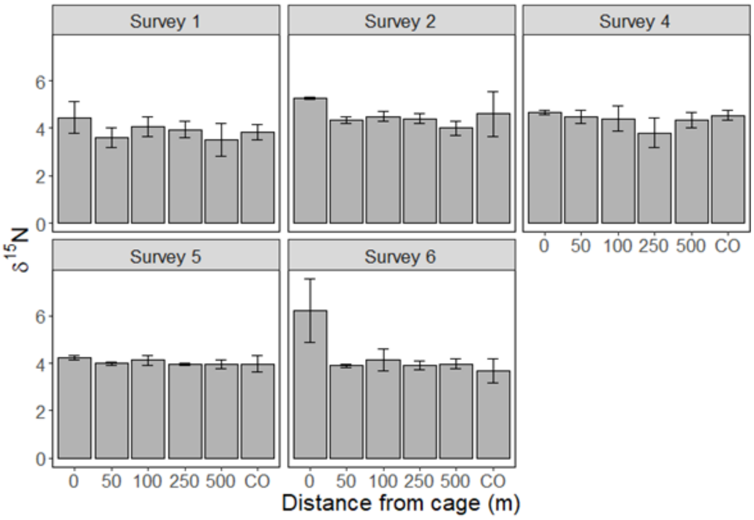


Figure 5.4-26 Nitrogen ($\delta^{15}N$) stable isotopes values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

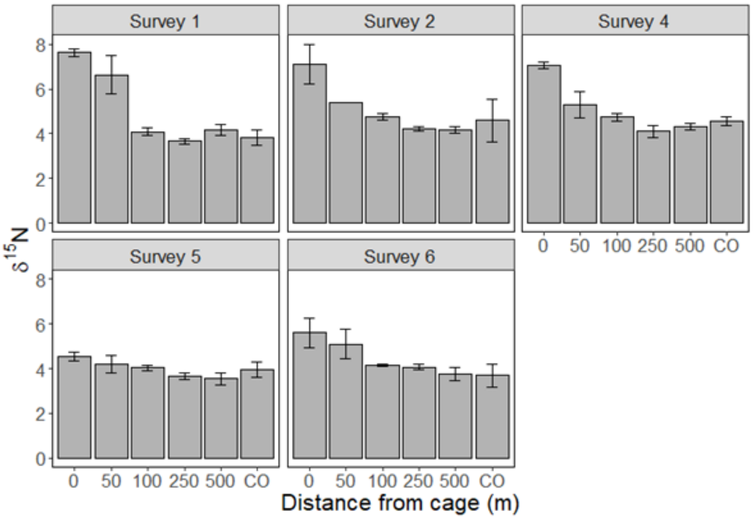


Figure 5.4-27 Nitrogen ($\delta^{15}N$) stable isotopes values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

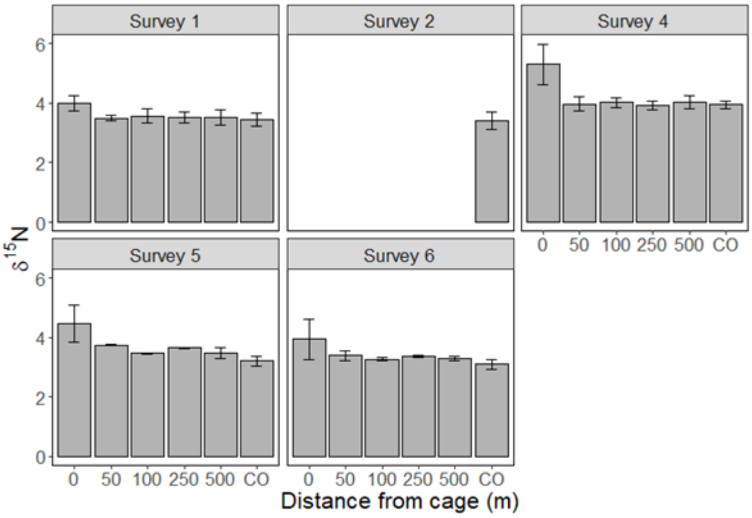


Figure 5.4-28 Nitrogen ($\delta^{15}N$) stable isotopes values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Data not available for Survey 2.

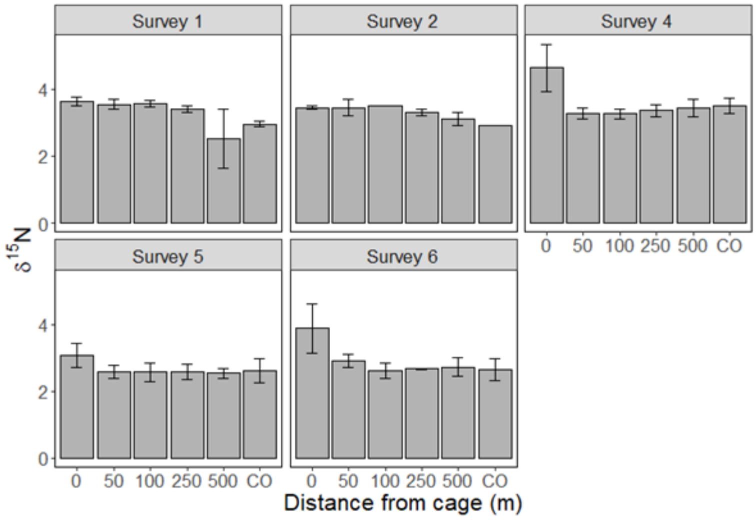


Figure 5.4-29 Nitrogen (δ^{15}N) stable isotopes values (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

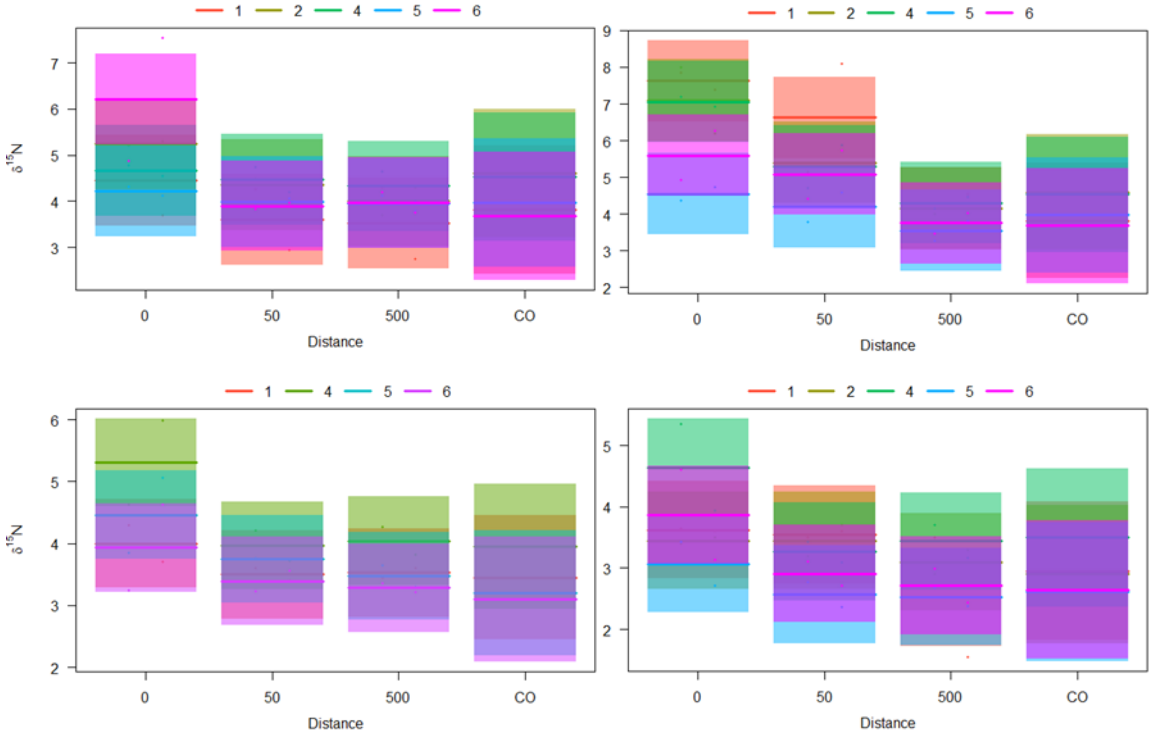


Figure 5.4-30 Nitrogen (δ^{15}N) stable isotopes values (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Dissolved Oxygen

Table Head Central. Dissolved oxygen levels were highly variable between sites, surveys and transects. There was evidence of slightly depleted bottom water oxygen levels at the sites closest to the cages in some but not all surveys (Figure 5.4-31), notably in survey 3 when the effect of distance was significant (Table 5.4-1). Bottom water oxygen levels were often higher at the control compared to the 50 and 100 m sites, and these differences were significant in surveys 3 and 6. The variation in bottom water oxygen levels between surveys was significant at the 0, 50 m and 500 m sites (Figure 5.4-35, Table 5.4-2); bottom water oxygen levels were notably lower at the sites in survey 6.

Gordon. Dissolved oxygen levels were highly variable between sites, surveys and transects. There was evidence of slightly depleted bottom water oxygen levels at the sites closest to the cages in some of the surveys (Figure 5.4-32), but the effect of distance was not significant in any of the surveys (Table 5.4-1). Bottom water oxygen levels were higher at the control compared to the 50 and 100 m sites, and these differences were significant in most cases. The variation in bottom water oxygen levels between surveys was significant at the 0, 50 m and 500 m sites (Figure 5.4-35, Table 5.4-2), with oxygen levels notably lower at the sites in survey 6.

Strahan. Dissolved oxygen levels were highly variable between surveys at Strahan but there appeared to be less variation between sites along the transects. There was some evidence of slightly depleted bottom water oxygen levels at the sites closest to the cages in some of the surveys (Figure 5.4-33), but the effect of distance was not significant in any of the surveys (Table 5.4-1). Although bottom water oxygen levels were quite variable between the control sites, the comparison with levels at the 50 and 100 m sites was significant in surveys 1 and 4. The variation in bottom water oxygen levels between surveys was significant at the 0, 50 m, 500 m and control sites (Figure 5.4-35, Table 5.4-2), with bottom water oxygen levels notably lower in survey 6.

Franklin. Dissolved oxygen levels were highly variable between sites, surveys and transects. However, with the exception of survey 4, there was evidence of depleted bottom water oxygen levels at the sites closest to the cages (Figure 5.4-34), and the effect of distance was significant in surveys 1 and 2 (Table 5.4-1). Bottom water oxygen levels were higher at the control compared to the 50 and 100 m sites in surveys 2, 3, 5 and 6, and the differences were significant in surveys 2 and 6. Consistent with the other leases, bottom water oxygen levels varied significantly between surveys at the 0, 50 m and 500 m sites (Figure 5.4-35, Table 5.4-2), with the lowest bottom water oxygen levels in survey 6.

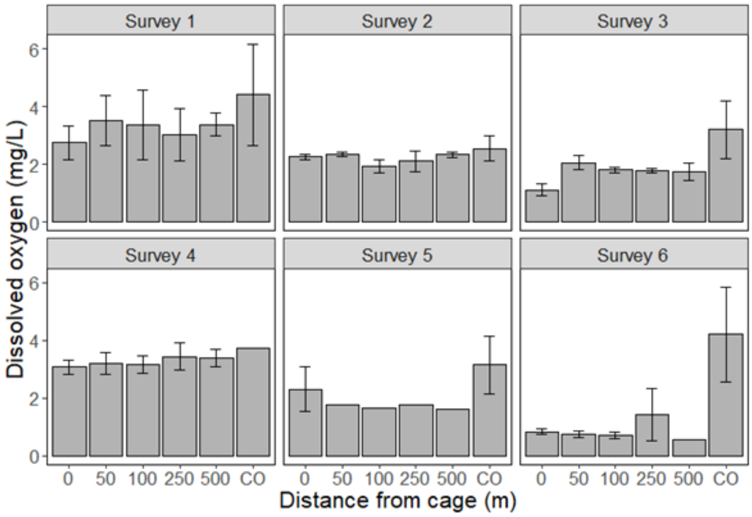


Figure 5.4-31 Bottom water dissolved oxygen concentration (mg/L; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

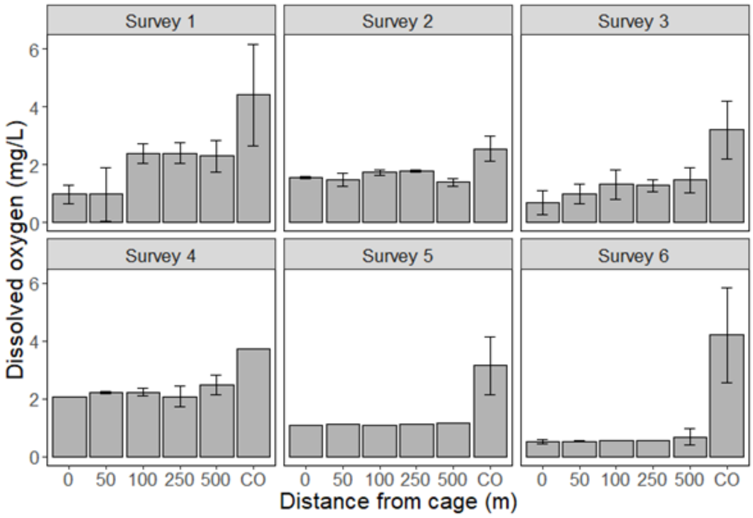


Figure 5.4-32 Bottom water dissolved oxygen concentration (mg/L; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

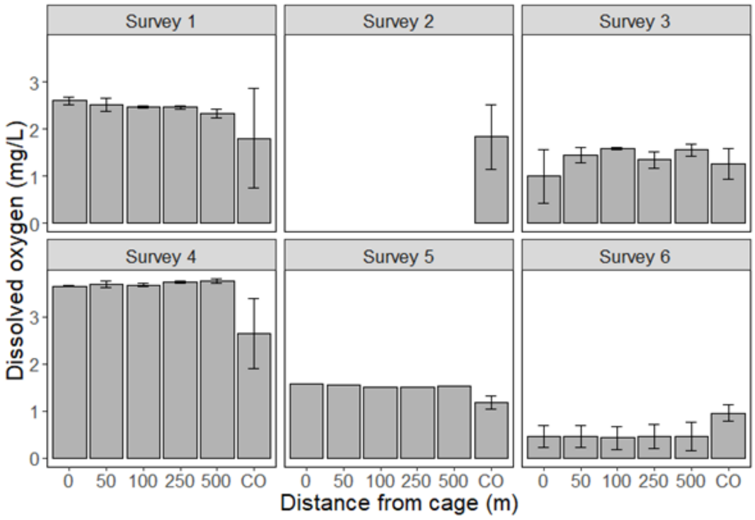


Figure 5.4-33 Bottom water dissolved oxygen concentration (mg/L; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

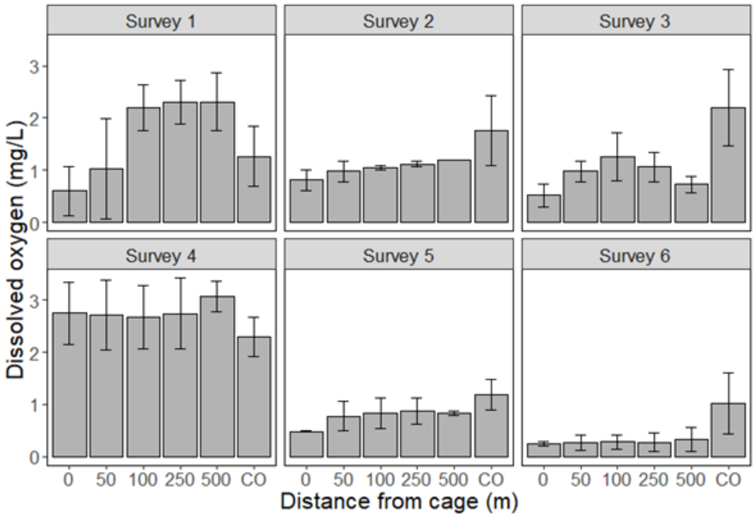


Figure 5.4-34 Bottom water dissolved oxygen concentration (mg/L; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

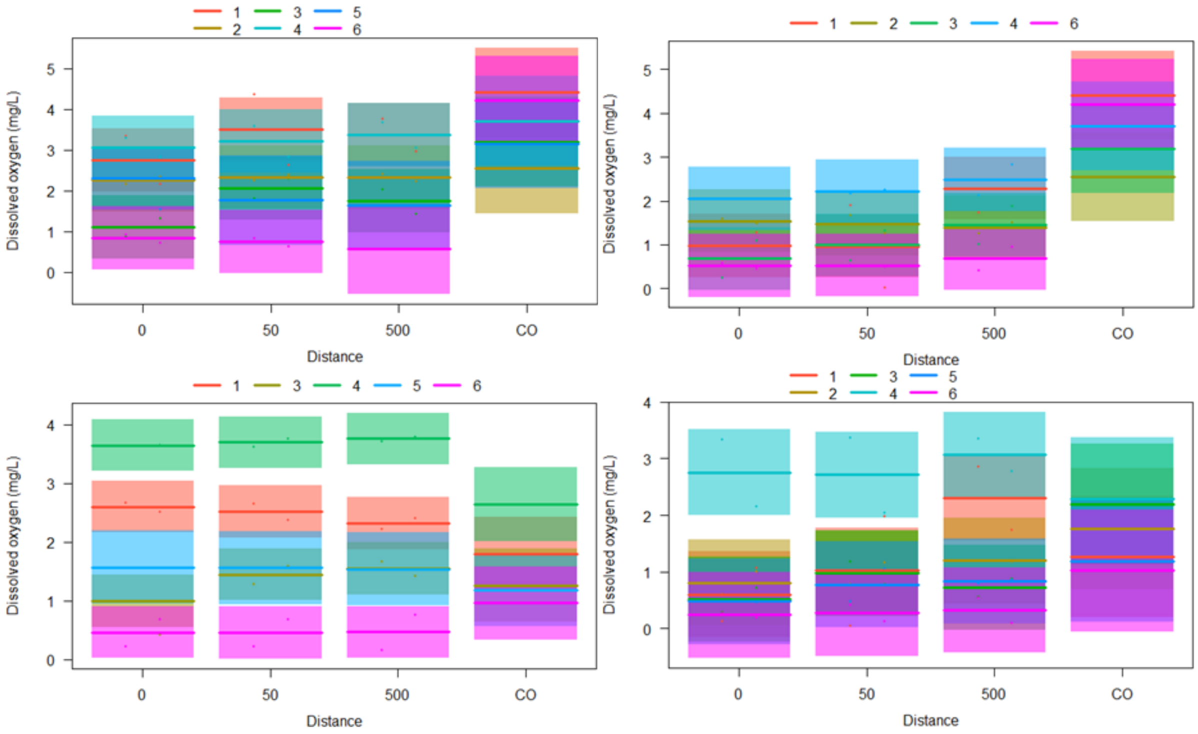


Figure 5.4-35 Bottom water dissolved oxygen concentration (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

5.4.1.2 Macrofauna

Across the six[1] surveys, 13,330 (234 grabs), 10,354 (234 grabs), 7676 (194 grabs) and 5894 (231 grabs) invertebrates were collected at the Table Head Central, Gordon, Strahan and Franklin leases respectively, comprising 65 taxa at Table Head Central, 64 taxa at Gordon, 48 taxa at Strahan and 39 taxa at Franklin. Overall, polychaetes were the most abundant taxon at the Table Head Central lease, making up 85% of the total taxa abundance, followed by crustaceans (8%), molluscs (5%), echinoderms (1%) and others (< 1%). At Gordon polychaetes made up 90% of the total taxa abundance, followed by crustaceans (7%), molluscs (7%), echinoderms (1%) and others (< 1%). Polychaetes were still dominant at Strahan (72%), but crustaceans (23%) and echinoderms (3.5%) made up a greater proportion of the community, followed by molluscs (1%) and others (< 1%). At Franklin polychaetes made up 88% of the total taxa abundance, followed by crustaceans (9%), echinoderms (3%), molluscs (1%) and others (<1%).

At Table Head, polychaetes were present in 94% of the samples, crustaceans in 76%, molluscs in 56%, echinoderms in 38% and the rest of the groups in < 10% of the samples. At Gordon, polychaetes were present in 88% of the samples, crustaceans in 62%, molluscs in 24%, echinoderms in 23% and the rest of the groups in < 10% of the samples. At Strahan, polychaetes were present in 97% of the samples, echinoderms in 56%, crustaceans in 45%, molluscs in 34%, and the rest of the groups in < 10% of the samples. At Franklin, polychaetes were present in 76% of the samples, crustaceans in 42%, echinoderms in 26%, molluscs in 17% and the rest of the groups in < 10% of the samples. At Table Head 8 of 234 samples had no fauna. At Gordon 13 of 231, at Strahan 3 of 194 and at Franklin 38 of 231 samples had no fauna.

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[1] The Strahan lease was only surveyed 5 times; it was not surveyed May 2015 (survey 2).

At Table Head, 40% of the taxa were crustaceans, 38% polychaetes, 8% molluscs, 3% echinoderms, 3% ascidians, 3% anemones, 3% nemerteans, and 2% other. The most abundant taxon at Table Head over the six surveys was the terebellid polychaete *Pista australis*, making up 37% of the total abundance, followed by the dorvilleid polychaete *Schistomeringos loveni* (9.7%), spionid polychaete *Pseudopolydora cf. paucibranchiata* (15.2%), ampharetid polychaete *Samythella* sp. (5.0%), sabellid polychaete *Branchiomma* sp. (4.7%), bivalve *Parathyasira resupine* (4.5%), nebaliid crustacean *Nebalia* sp. (3.0%), and other taxa (2.0% or less).

At Gordon, 42% of the taxa were crustaceans, 36% polychaetes, 8% molluscs, 3% echinoderms, 3% ascidians, 3% anemones, 3% nemerteans and 2% other. The most abundant taxon at Gordon over the six surveys was the dorvilleid polychaete *Schistomeringos loveni* making up 30% of the total abundance, followed by the terebellid polychaete *Pista australis* (23.9%) the spionid polychaete *Pseudopolydora cf. paucibranchiata* (20.4%), ampharetid polychaete *Samythella* sp. (8.3%), sabellid polychaetes *Branchiomma* sp. (3.0%) and *Euchone varibilis* (2.1%), nebaliid crustacean *Nebalia* sp. (2.8%), and other taxa (2.0% or less).

At Strahan, 38% of the taxa were crustaceans, 42% polychaetes, 10% molluscs, 4% echinoderms, 4% anemones and 2% nemerteans. The most abundant taxon at Strahan over the five surveys was the terebellid polychaete *Pista australis*, making up 21% of the total abundance, followed by the nebaliid crustacean *Nebalia* sp. (20.2%), the spionid polychaete *Pseudopolydora cf. paucibranchiata* (20.2%), the dorvilleid polychaete *Schistomeringos loveni* (15%), ampharetid polychaete *Samythella* sp. (6.0%), echinoderm *Echinocardium cordatum* (3.5%), sabellid polychaete *Euchone varibilis* (2.3%), dorvilleid polychaete *Ophryotrocha shieldsi* (2.1%) and other taxa (2.0% or less). However, it is important to note that the abundance of *Nebalia* sp., *Euchone varibilis, Ophryotrocha shieldsi* was highly variable between surveys.

At Franklin, 38% of the taxa were crustaceans, 41% polychaetes, 13% molluscs, 5% anemones and 3% echinoderms. The most abundant taxon at Franklin over the six surveys was the dorvilleid polychaete *Schistomeringos loveni* making up 56% of the total abundance, followed by the terebellid polychaete *Pista australis* (14.7%), spionid polychaete *Pseudopolydora cf. paucibranchiata* (7.3%), nebaliid crustacean *Nebalia* sp. (6.4%), ampharetid polychaete *Samythella* sp. (4.8%), capitellid polychaete *Capitella* sp. (3.0%), echinoderm *Echinocardium cordatum* (2.6%) and other taxa (2.0% or less).

Total abundance

Table Head Central. Although the effect of distance on total abundance was only significant in survey 5, there was a clear pattern across all surveys with abundance peaking at 50-100 m from the cage (Figure 5.4-36, Table 5.4-1). Average total abundance at 50 and 100 m was also often higher than at the controls, but this difference was only significant between 50 m and the controls in surveys 3 and 4 (Table 5.4-1). There was no significant effect of survey at the 0 and 50 m from cage sites or more distant sites (500 m and CO; Figure 5.4-40, Table 5.4-2).

Gordon. The peak in abundance at Gordon appeared to occur at greater distances (100-500 m) from the cage compared to Table Head Central (Figure 5.4-37); the effect of distance was only significant in survey 4 (Table 5.4-1). The difference in abundance between the control and 50 and 100 m sites varied between surveys, and it was only in survey 6 that the difference between the controls and the 50 m site was significant. There was no significant effect of survey at the 0 and 50 m from cage sites or more distant sites (500 m and CO; Figure 5.4-40, Table 5.4-2).

Strahan. The peak in abundance at Strahan was much closer to the cage (0-50 m), and in surveys 3 and 4 the effect of distance was significant (Figure 5.4-39, Table 5.4-1). Although total abundance was greater at the 50 or 100 m sites compared to the controls, the differences were not significant in any of the surveys. There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-40, Table 5.4-2).

Franklin. The peak in abundance at Franklin, like at Gordon, was typically more distant from the cages (Figure 5.4-39) ranging from 50 m in survey 3, 100 m in surveys 1 and 4, and 250 m in survey 5. Although total abundance was greater at the 50 or 100 m sites compared to the

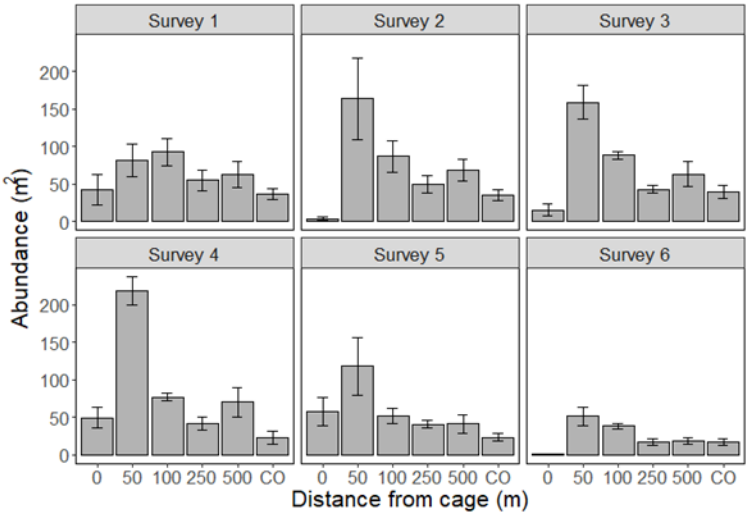


Figure 5.4-36 Total abundance (m^{-2}; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

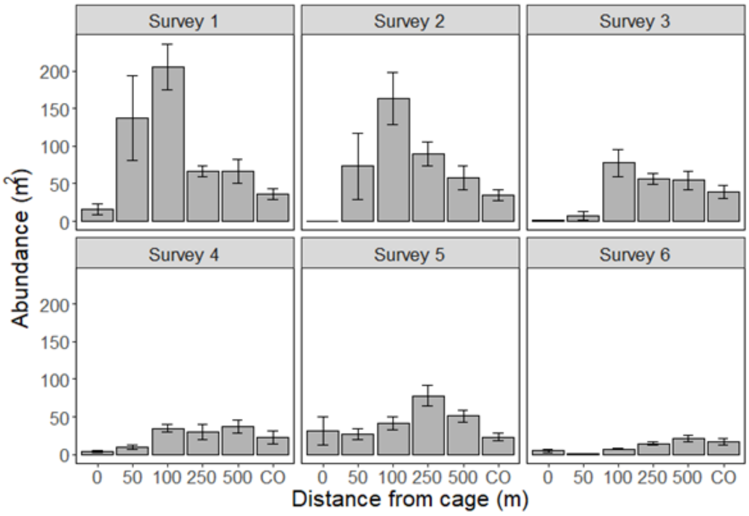


Figure 5.4-37 Total abundance (m^{-2}; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

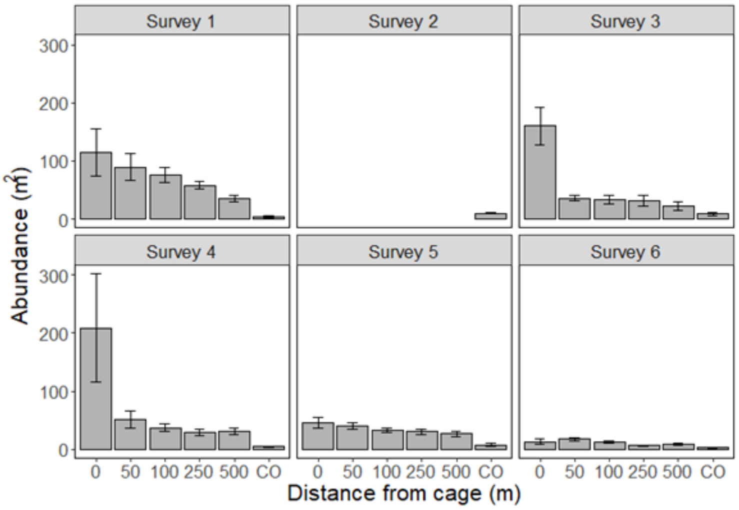


Figure 5.4-38 Total abundance (m^{-2}; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Data is not available for Survey 2.

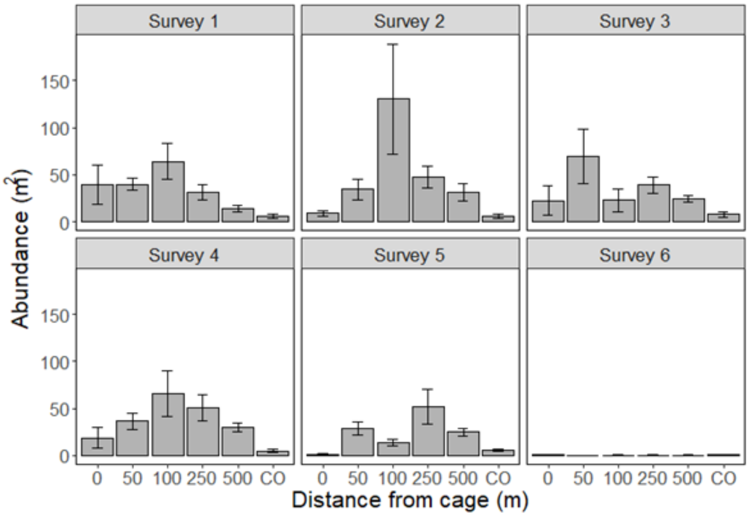


Figure 5.4-39 Total abundance (m^{-2}; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

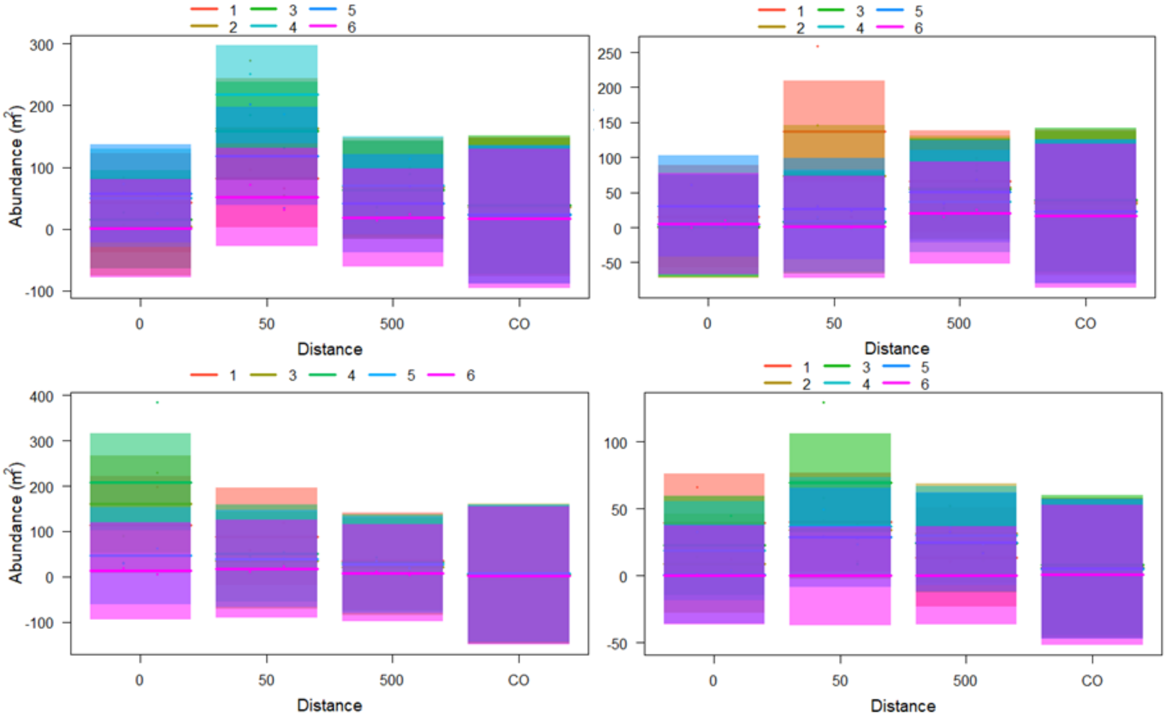


Figure 5.4-40 Total abundance (m^{-2}; predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

controls, the differences were not significant in any of the surveys (Table 5.4-1). There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-40, Table 5.4-2).

Species Richness

Table Head Central. Species richness decreased closer to the cage on the transects in all six surveys, and the effect of distance was significant in all surveys (Figure 5.4-41, Table 5.4-1). There was no significant difference in species richness between the 50 m or 100 m sites and controls in each of the surveys (Table 5.4-1). There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-45, Table 5.4-2).

Gordon. Species richness decreased closer to the cage on the transects in all six surveys. With the exception of survey 1, the effect of distance was significant in all surveys (Figure 5.4-42, Table 5.4-1). Species richness was typically lower at the 50 m and 100 m sites compared with the controls; the difference was significant between the 50 m and control sites in surveys 2, 3, 5 and 6 and between the 100 m and control sites in surveys 5 and 6 (Table 5.4-1). There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-45, Table 5.4-2).

Strahan. Species richness was lower closer to the cage in surveys 1, 4, 5 and 6 compared to the intermediate distances, but was also lower at the more distant sites (Figure 5.4-43). In survey 3 species richness increased in closer proximity to the cages, and the pattern with distance was significant (Figure 5.4-43, Table 5.4-1). At Strahan, species richness was typically higher at the 50 m and 100 m sites compared with the controls; the difference was significant between the 50 m and control sites in surveys 1, 4 and 6 and between the 100 m and control sites in surveys 1 and 4 (Table 5.4-1). Although species richness was lower at all the sites in survey 6 compared to the other surveys, the effect of survey was significant at the 0 m and 500 m sites but not at the 50 m and controls (Figure 5.4-44, Table 5.4-2).

Franklin. With the exception of survey 6 when fauna was largely absent, species richness was lower closer to the cage in all surveys, but like Strahan, was also lower at the more distant sites (Figure 5.4-44). The difference in species richness between the 50 m and 100 m sites and the controls was significant in survey 1 and between the 100 m and controls in survey 4 (Table 5.4-1). Although species richness was lower at all the sites in survey 6 compared to the other surveys, the effect of survey was significant at the 0 m, 50 m and 500 m sites but not at the controls (Figure 5.4-44, Table 5.4-2).

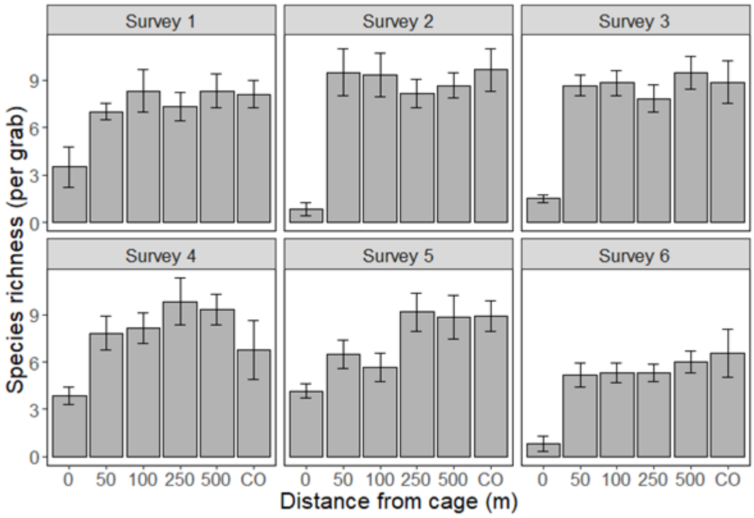


Figure 5.4-41 Species richness (per grab; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

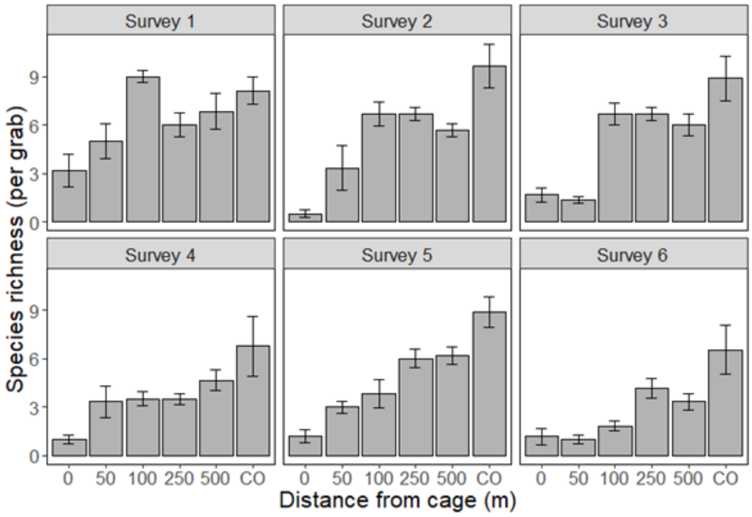


Figure 5.4-42 Species richness (per grab; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

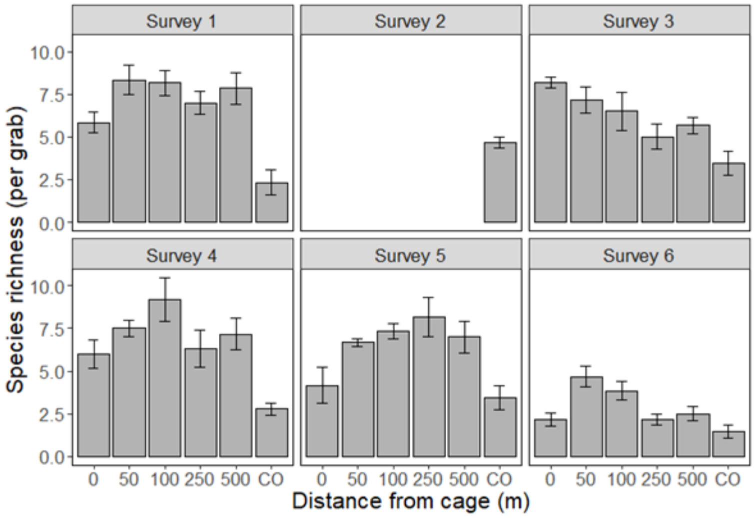


Figure 5.4-43 Species richness (per grab; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

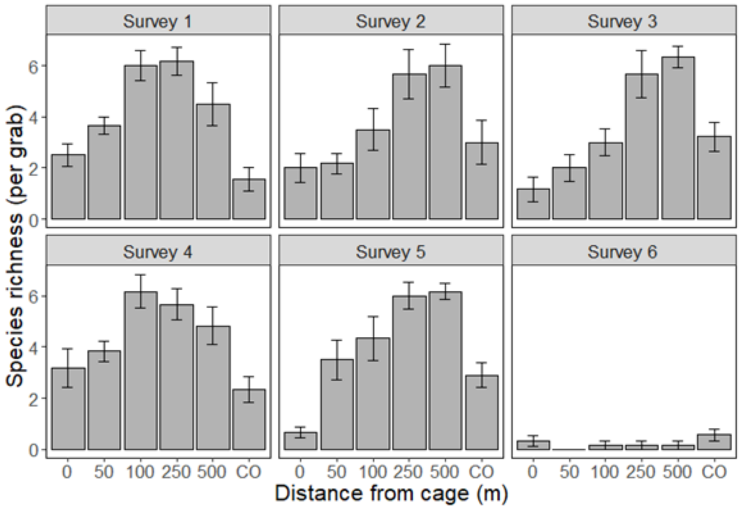


Figure 5.4-44 Species richness (per grab; average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

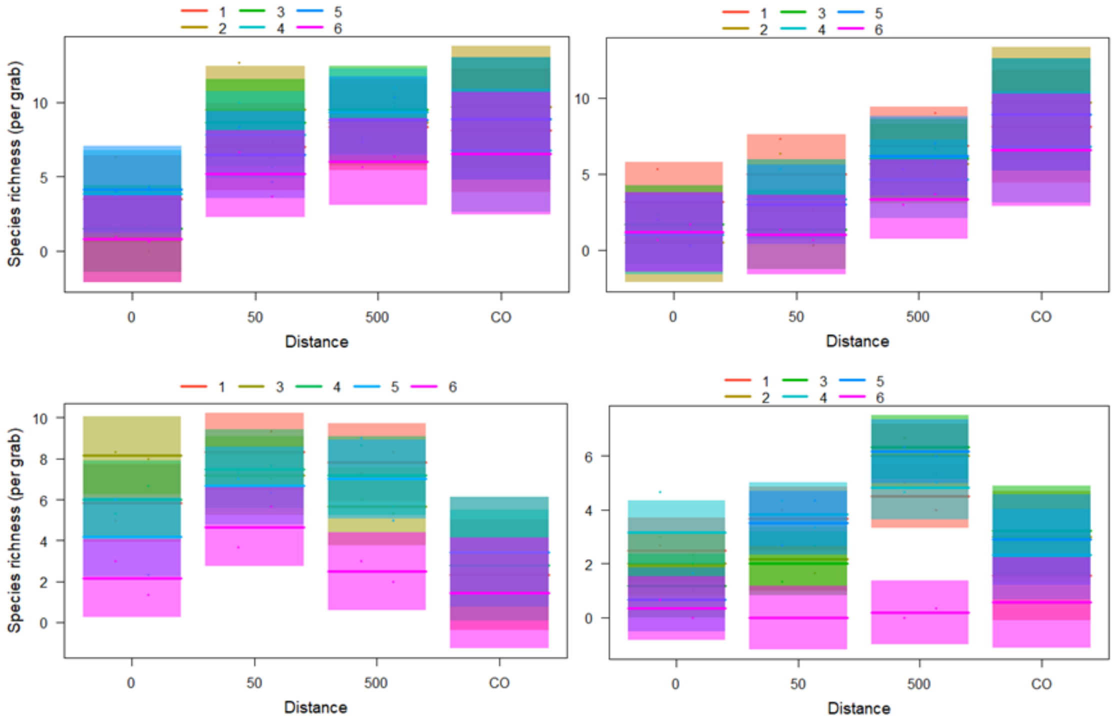


Figure 5.4-45 Species richness (per grab; predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Species diversity

Table Head Central. Species diversity was reduced closer to the cage on the transects in all six surveys and the effect of distance was significant in all surveys (Figure 5.4-46, Table 5.4-1). Species diversity was lower at the 50 m and/or 100 m sites compared with the controls in most of the surveys, but the difference was only significant in survey 5 between 100 m and control sites (Table 5.4-1). There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-50, Table 5.4-2).

Gordon. Species diversity was lower closer to the cage on the transects in all six surveys, but the effect of distance was only significant in surveys 2 and 4 (Figure 5.4-47, Table 5.4-1). Species diversity was lower at the 50 m and 100 m sites compared with the controls in all surveys; the difference was significant between the 50 m and control sites in surveys 2, 3, 5 and 6 and between the 100 m and control sites in survey 6 (Table 5.4-1). There was no significant effect of survey at either the sites closest to the cages (0 and 50 m) or the more distant sites (500 m and CO; Figure 5.4-50, Table 5.4-2).

Strahan. Species diversity was lower closer to the cage in surveys 1, 4, 5 and 6 compared to the intermediate distances, but was also lower at the more distant sites (Figure 5.4-48). In survey 3 species diversity appeared to increase in closer proximity to the cages. At Strahan, species diversity was higher at the 50 m and 100 m sites compared with the controls; the difference was significant between the 50 m and control sites in surveys 1, 3 and 6 and between the 100 m and control sites in surveys 1, 4 and 6 (Table 5.4-1). Although species diversity was lower at all sites in survey 6 compared to the other surveys, the effect of survey was significant at the 0 m and 500 m sites but not at the 50 m and controls (Figure 5.4-50, Table 5.4-2).

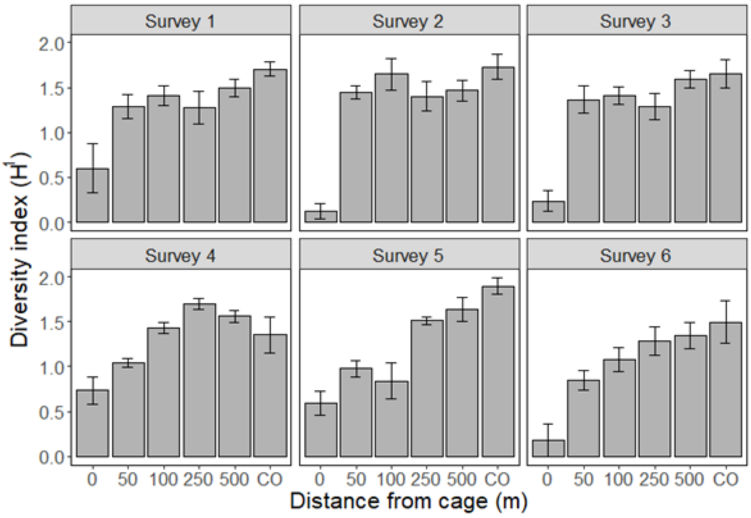


Figure 5.4-46 Species diversity index (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

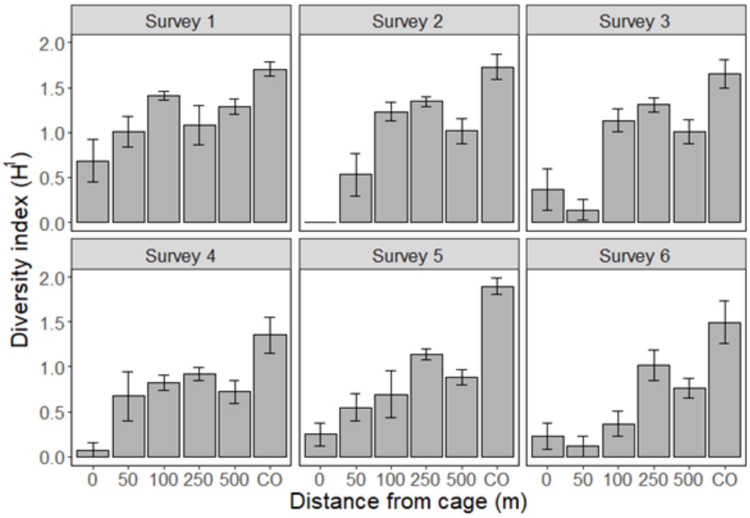


Figure 5.4-47 Species diversity index (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

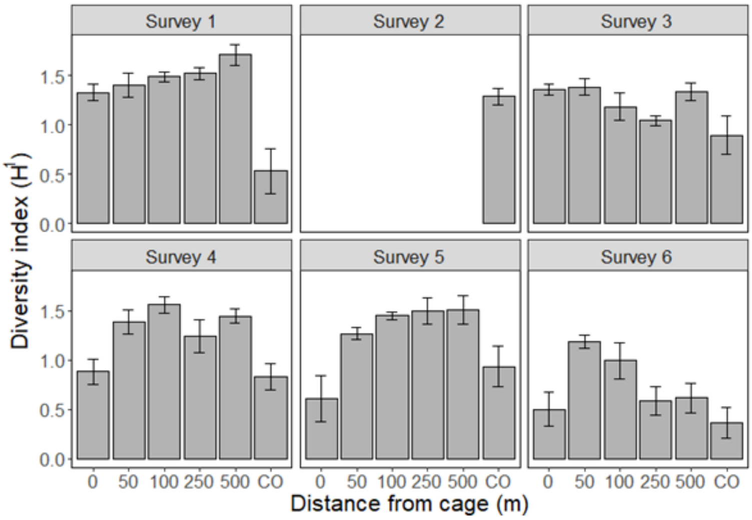


Figure 5.4-48 Species diversity index (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

Franklin. With the exception of survey 6 when fauna were largely absent, species diversity was often lower closer to the cage in all surveys, but like Strahan, was also lower at the more distant sites (Figure 5.4-49). The difference in species richness between the 50 m and 100 m sites and the controls was only significant in survey 1 between the 100 m and controls (Table 5.4-1). Although species diversity was lower at all sites in survey 6 compared to the other surveys, the effect of survey was significant at the 0 m, 50 m and 500 m sites but not at the controls (Figure 5.4-50, Table 5.4-2).

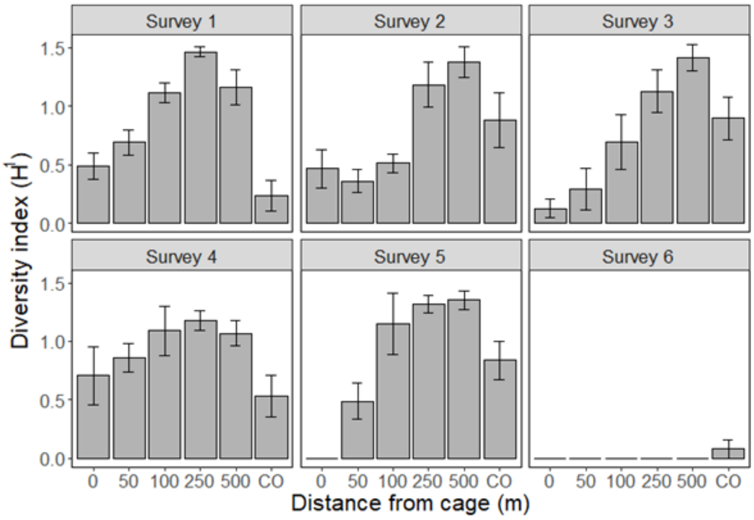


Figure 5.4-49 Species diversity index (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

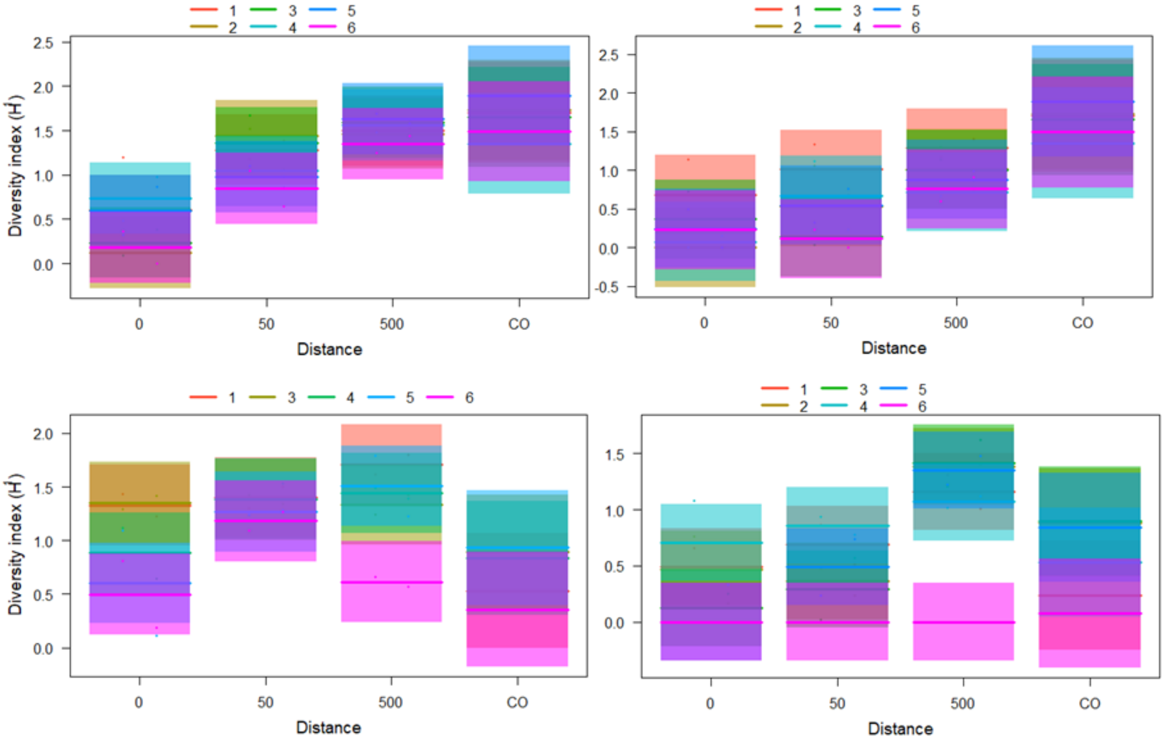


Figure 5.4-50 Species diversity index (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500 m and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

Multivariate analysis

The multivariate GLM showed that the composition of the faunal assemblages changed with distance in all surveys at each lease (Table 5.4-1). For the species that showed a significant effect of distance the fourth corner coefficients were plotted using a LASSO penalty for each survey (see Appendix A for plots of abundance by distance for individual species).

Table Head Central

In survey 1 the dorvilleid polychaete *Schistomeringos loveni* showed a strong positive relationship with proximity to the cage, and the sabellid polychaete *Branchiomma* sp. and bivalve *Parathyasira resupine* had a positive relationship with distance from the cage (Figure 5.4-51). In survey 2, *Schistomeringos loveni* and the nebaliid crustacean *Nebalia* sp. showed positive relationships with proximity to the cage and *Parathyasira resupine*, *Branchiomma* sp., *Pista australis,* a terebellid polychaete and the spionid polychaete, *Pseudopolydora cf. paucibranchiata,* all showed positive relationships with distance from the cage (Figure 5.4-52). In survey 3, *Schistomeringos loveni* and *Nebalia* sp again showed a positive relationship with proximity to the cage, and the same species were found in survey 2 together with the ampharetid polychaete *Samythella* sp. and ostracod *Philomedid* sp., which also showed a positive relationship with distance from the cage (Figure 5.4-53). A similar pattern was seen in surveys 4 (Figure 5.4-54) and 5 (Figure 5.4-55), but with some variation in the composition of species that showed a significant effect of distance. In survey 6, when faunal abundance declined throughout the harbour, *Schistomeringos loveni* remained the dominant taxon showing a positive relationship with proximity to the cage, while *Pseudopolydora cf. paucibranchiata* and *Parathyasira resupine* showed a positive relationship with distance from the cage (Figure 5.4-56). It is worth noting that though both *Pseudopolydora cf. paucibranchiata* and *Pista australis* were rare at the 0 m cage site, they were often positively correlated, and in greatest abundance at the intermediate distances; this is described further below.

The unconstrained ordination from the LVM shows the separation of samples based on their species composition and relationship with distance from the cage (Figure 5.4-57). In survey 1, species composition at 0 m overlapped with the assemblages found at 50 and 100 m. The 250 m and 500 m assemblages were distinct from each other with the 100 m assemblage overlapping the assemblage at 250 m. Of the most important species identified in the LVM, the polychaete *Schistomeringos loveni* was characteristic of the 0, 50 and 100 m sites (~331, 565 and 425 ind. m^{-2} respectively), but it was also found in lower densities at 250 and the 500 m sites (~67, 12 ind. m^{-2} respectively). The sabellid polychaetes *Parathyasira resupine* and *Branchiomma* sp. were not present at the 0 m sites but they both increased in abundance with distance from the cage. In survey 2 the assemblage at 0 m was distinct from the other distances, and there was less overlap between the other distances (Figure 5.4-57). The nebaliid crustacean *Nebalia* sp. and *Schistomeringos loveni* characterised the 0 m assemblage, but in survey 2 *Schistomeringos loveni* was far more abundant at 50 (420 ind. m^{-2}) and 100 m (240 ind. m^{-2}) relative to sites closer (25 ind. m^{-2} at 0 m) or more distant (12 & 3 ind. m^{-2} at 250 and 500 m respectively) from the cage. The spionid polychaete *Pseudopolydora cf. paucibranchiata*, the ampharetid polychaete *Samythella* sp., and terebellid polychaete *Pista australis* were characteristic of the intermediate distances, with all three species most abundant between 50 and 250 m from the cage and absent (or very rare) at 0 m. The sabellid polychaete *Branchiomma* sp. and the bivalve *Parathyasira resupine* were also found at the intermediate distances, but they were generally more abundant and characteristic at the more distant 500 m sites. In survey 3 the species composition and patterns with distance were consistent with that observed in survey 2, with the exception that the 0 m assemblage was more variable and the community tended to overlap with the other distances; *Nebalia* sp. and *Schistomeringos loveni* were still characteristic of the 0 m assemblage with few other species present, but these two species were also common at the intermediate distances. In survey 4 the assemblage at 0 m was again distinct, but otherwise the change in species composition with distance remained similar: *Nebalia* sp. and *Schistomeringos loveni* characteristic of the 0 m assemblage, *Pseudopolydora cf. paucibranchiata* and *Pista australis* characteristic of the intermediate distances and *Branchiomma* sp. and the bivalve *Parathyasira resupine* characteristic of the more distant sites. In survey 4, the amphipod *Charcotia australiensis* was also characteristic of the 0 m sites, and although not common in previous surveys, when present it was typically only observed at the 0 and 50 sites. In survey 5, the 0 m assemblage remained distinct from the other distances, again characterised by the presence of *Schistomeringos loveni* and *Nebalia* sp. In surveys 4 and 5, the opportunistic polychaete *Capitella* sp., a known indicator of organic enrichment around farms in SE Tasmania, was also characteristic of the 0 m assemblage, but was found in relatively low densities (42 and 39 ind. m^{-2} at 0 m in surveys 4 and 5 respectively) and not observed at other distances in either of these surveys. The species assemblage at 50 m was also distinct from the 250 and 500 m sites, with the 100 m often overlapping both. The ostracod *Philomedid* sp. and the bivalve *Parathyasira resupine* were characteristic of the more distant sites and *Pseudopolydora cf. paucibranchiata* and *Pista australis* were again characteristic of the intermediate distances. In survey 6, the decline in total abundance and species diversity observed at the other leases was also seen at Table Head Central, albeit to a lesser extent. Despite the decline in abundance, *Schistomeringos loveni* remained common at 50 and 100 m, and both *Pseudopolydora cf. paucibranchiata* and *Pista australis* were common from 50-500 m, although *Parathyasira resupine* was most common at 500 m.

Gordon

In survey 1 the nebaliid crustacean *Nebalia* sp. showed a strong positive relationship with proximity to the cage, and the polychaetes *Pseudopolydora* cf. *paucibranchiata*, *Pista australis*, *Phyllodoce* sp1, and *Samythella* sp. showed positive relationships with distance from the cage (Figure 5.4-58). In survey 2, the polychaetes *Branchiomma* sp. and *Pista australis* and echinoderm *Echinocardium cordatum* showed positive relationships with distance from the cage, and *Schistomeringos loveni*, *Samythella* sp., *Pseudopolydora* cf. *paucibranchiata* were positively related to intermediate distances (Figure 5.4-59). In survey 3

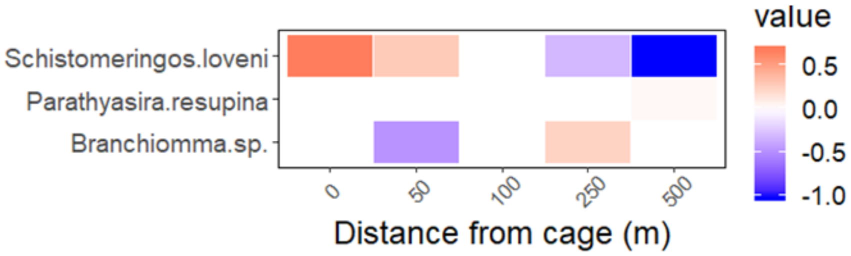


Figure 5.4-51 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 1.

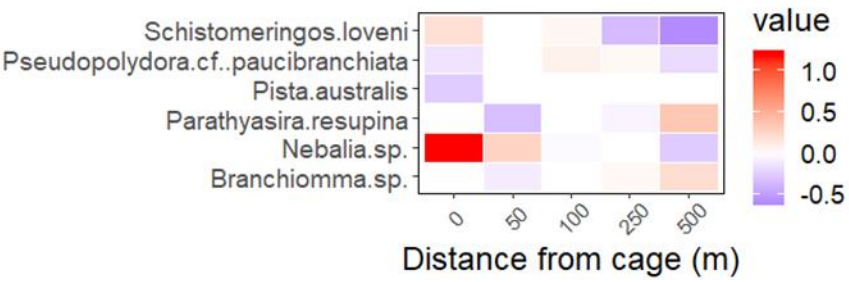


Figure 5.4-52 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 2.

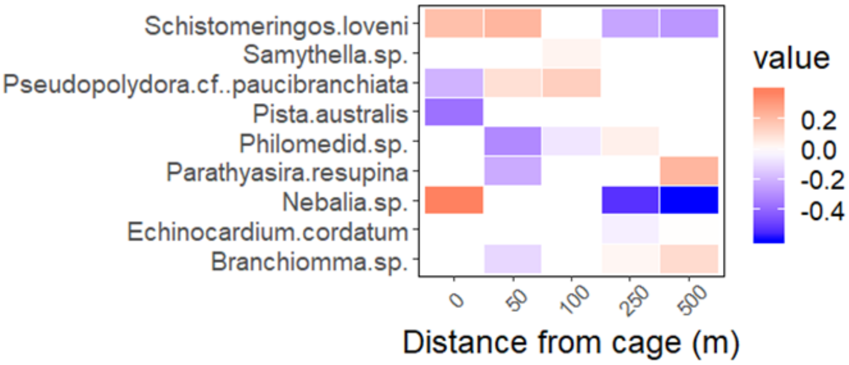


Figure 5.4-53 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 3.

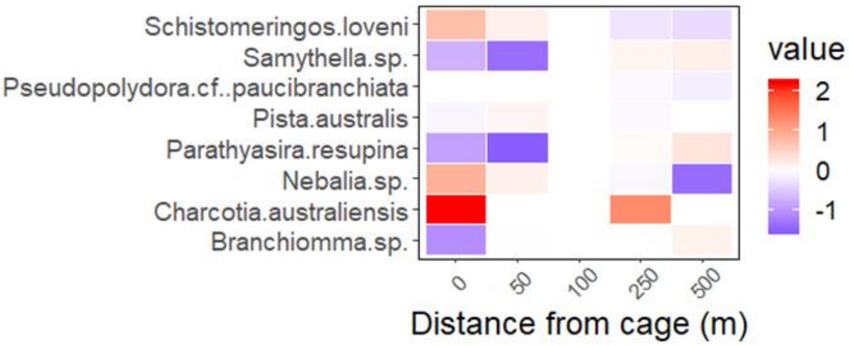


Figure 5.4-54 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 4.

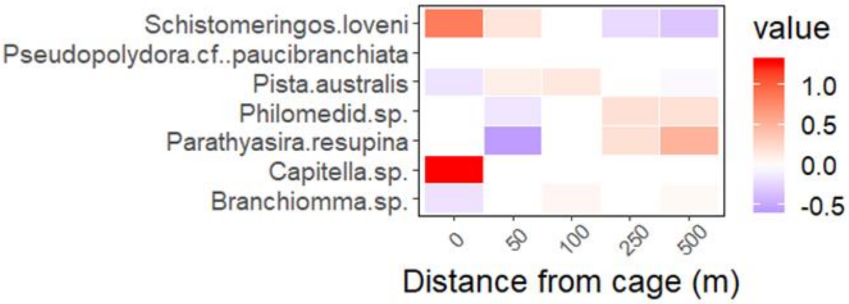


Figure 5.4-55 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 5.

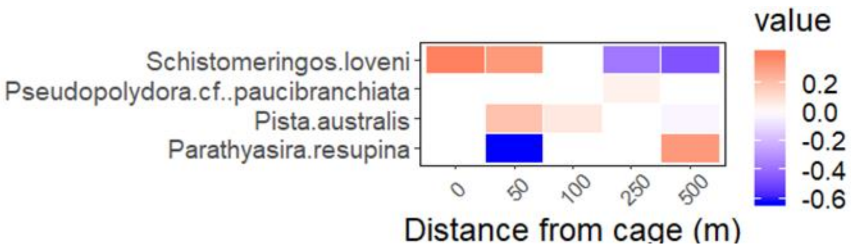


Figure 5.4-56 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Table Head Central lease in survey 6.

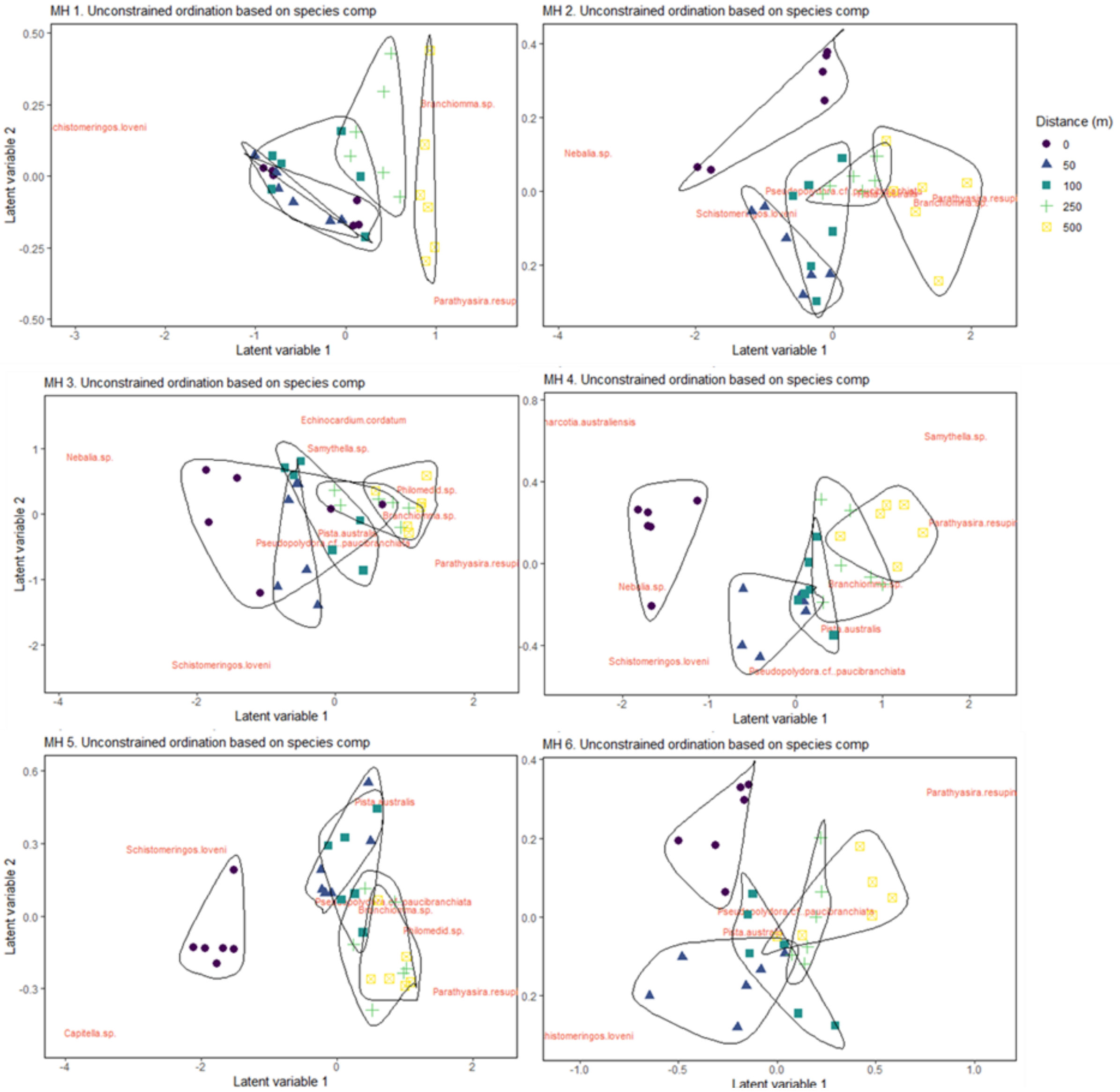


Figure 5.4-57 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1 - 6 at the Table Head Central lease. Samples are collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

a similar pattern was observed, with *Schistomeringos loveni, Samythella* sp., *Pseudopolydora* cf. *paucibranchiata* showing positive relationships with intermediate distances, and *Branchiomma* sp., *Pista australis* and *Echinocardium cordatum* showing positive relationships with the more distant sites (Figure 5.4-60). In survey 4, *Nebalia* sp. again showed a strong positive relationship with proximity to the cage and *Schistomeringos loveni* and *Capitella* sp. were positively associated with intermediate distances; *Pseudopolydora* cf. *paucibranchiata, Pista australis* and *Echinocardium cordatum* were positively associated with more distant sites (Figure 5.4-61). In this survey *Nebalia* sp. was absent at the cage site and in peak abundance at 50 m, and *Capitella* sp. peaked in abundance at 0 m but was also present at 50 m. A similar pattern was seen in survey 5 (Figure 5.4-62) with *Pseudopolydora* cf. *paucibranchiata, Pista australis* and *Echinocardium cordatum* positively associated with the more distant sites (Figure 5.4-55). In survey 6, the results from the multivariate GLM were not able to be interpreted due to the major decline in faunal abundance and diversity, consequently the plot with LASSO fits is not included.

The unconstrained ordination from the LVM shows the separation of samples based on their species composition and relationship with distance from the cage (Figure 5.4-57). In survey 1, species composition at 0 and 50 m from the cage was highly variable, most likely due to differences between the transects. The assemblage at 0 m was largely separated from the 100, 250 and 500 m assemblages which overlapped each other, with the 50 m overlapping both groups. Of the most key species identified in the LVM, *Nebalia* sp. was characteristic of the 0, 50 and 100 m sites and *Phyllodoce* sp1. was characteristic at 100, 250 and 500 m. Of the more abundant taxa, *Pseudopolydora* cf. *paucibranchiata* and *Samythella* sp. were both common across sites from 50 - 500 m, but were most abundant at 50 - 100 m, and *Pista australis* was also common across these distances but with no clear peak in abundance. In survey 2, there was virtually no fauna at the 0 m sites; only one *Schistomeringos loveni* and two *Nebalia* sp. were found across the six grabs. *Schistomeringos loveni* and *Pseudopolydora* cf. *paucibranchiata* were both common across sites from 50 - 500 m; *Schistomeringos loveni* was most common at 50 (689 ind. m^{-2}) and 100 m (1289 ind. m^{-2}) and *Pseudopolydora cf. paucibranchiata* at 100 (477 ind. m^{-2}) and 250 m (417 ind. m^{-2}) from the cage. *Samythella* sp. was also more common at the intermediate distances, with a peak abundance at 100 m from the cage. The species identified as being most characteristic of sites more distant from the cages were *Pista australis, Branchiomma* sp and *Echinocardium cordatum.* Significant variability in species composition at each distance was again evident in survey 3, but the pattern remained similar for the key species. *Schistomeringos loveni*, *Samythella* sp. and *Pseudopolydora* cf. *paucibranchiata* were characteristic of the intermediate distances and *Pista australis, Branchiomma* sp. and *Echinocardium cordatum* were characteristic of the more distant sites. Like the previous two surveys, the fauna at 0 m in survey 4 was quite depauperate, with *Nebalia* sp. as the main species found. The species characteristic of the assemblages at 50 and 100 m were *Schistomeringos loveni, Nebalia* sp. and *Capitella* sp., and at the more distant sites, *Pista australis* and *Echinocardium cordatum.* In survey 5 faunal abundance at 0 m was much higher, dominated by *Schistomeringos loveni* (365 ind. m^{-2}) and *Capitella* sp. (91 ind. m^{-2}); together with *Nebalia* sp., the same species were also characteristic of the assemblage at 50 m. *Pseudopolydora* cf. *paucibranchiata*, *Branchiomma* sp., *Pista australis* and *Echinocardium cordatum* were again characteristic of the more distant sites with peak abundances at 250 and 500 m from the cage. In survey 6 the tight clustering of the assemblages at 0 m, and some of the 50 and 100 m sites reflected the decline in faunal abundance and diversity. At the other sites, abundance and diversity had also declined, but species such as *Pseudopolydora* cf. *paucibranchiata* and *Pista australis* remained characteristic of the assemblages at the more distant sites. Although the abundance of

Schistomeringos loveni had also declined significantly from that recorded in the previous surveys it was still found at all distances (e.g. 10 - 60 ind. m^{-2}) in survey 6.

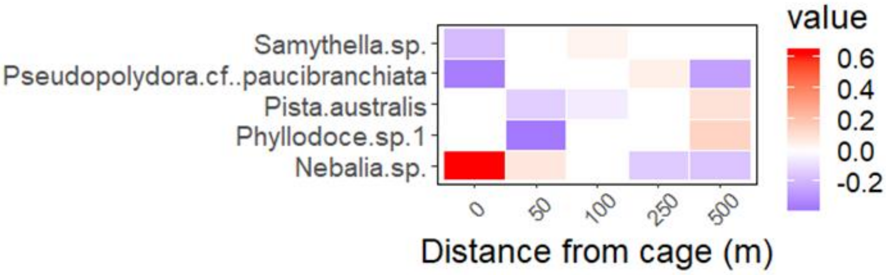


Figure 5.4-58 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Gordon lease in survey 1.

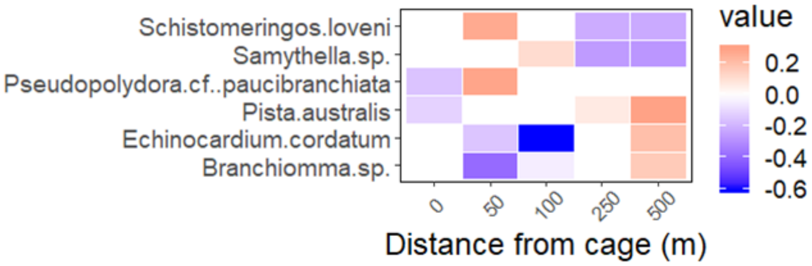


Figure 5.4-59 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Gordon lease in survey 2.

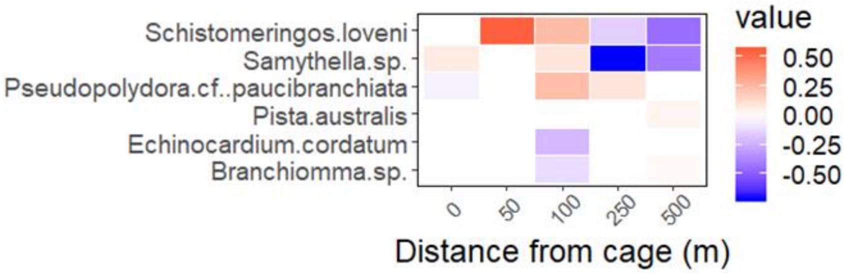


Figure 5.4-60 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Gordon lease in survey 3.

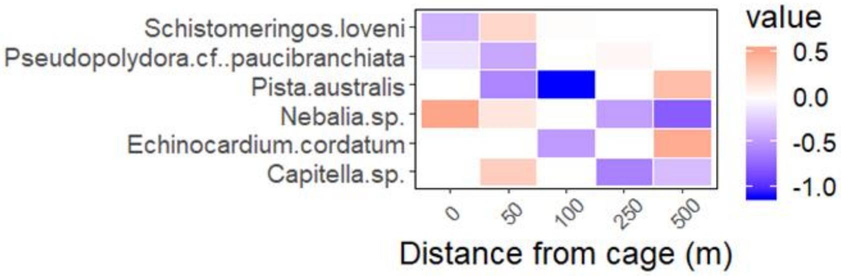


Figure 5.4-61 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Gordon lease in survey 4.

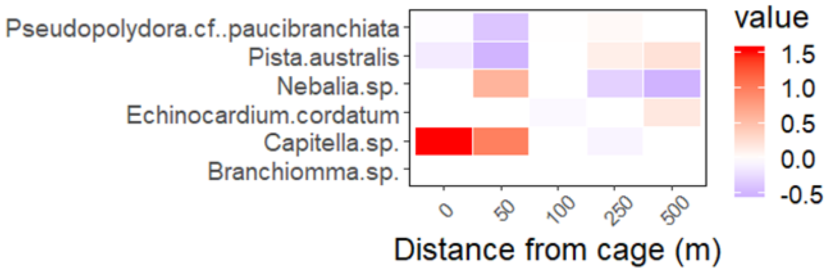


Figure 5.4-62 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that show a significant effect with distance from the Gordon lease in survey 5.

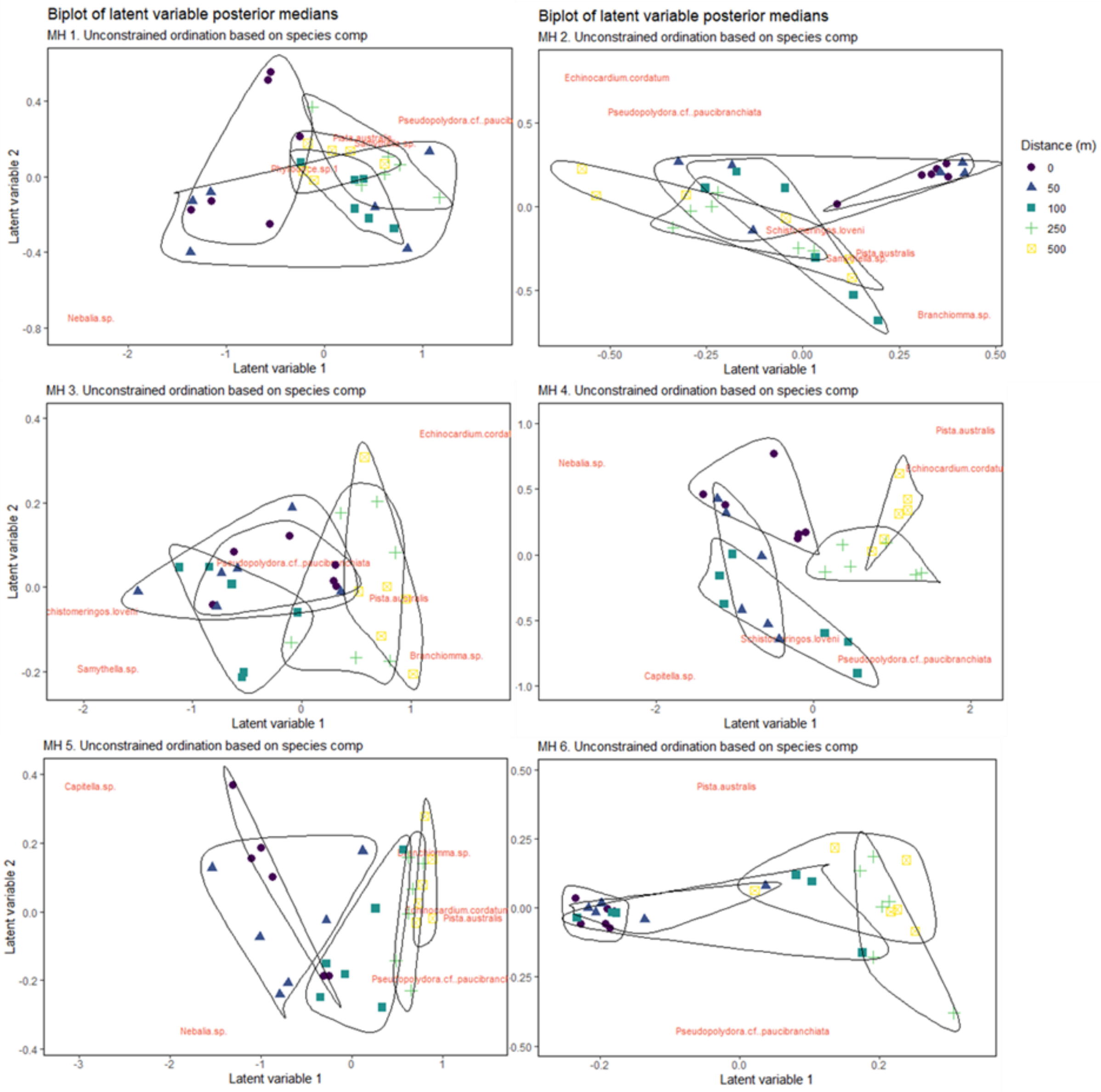


Figure 5.4-63 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1 - 6 (MH 1 – MH 6) at the Gordon lease. Samples were collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and far from the origin are more correlated.

Strahan

In survey 1 *Nebalia* sp., *Capitella* sp. and to a lesser extent *Schistomeringos loveni* all showed positive relationships with proximity to the cage (Figure 5.4-64); *Nebalia* sp. and *Capitella* sp. were far more abundant at 0 m, and *Schistomeringos loveni* at both 0 and 50 m, relative to other distances. In contrast, *Samythella* sp. and the sabellid polychaete *Euchone varibilis* showed positive relationships with distance from the cage. In survey 3, *Schistomeringos loveni*, *Nebalia* sp. and the amphipod *Charcotia australiensis* showed positive relationships with proximity to the cage and were most abundant at the 0 m cage sites (Figure 5.4-65). *Branchiomma* sp. and *Pista australis* showed positive relationships with distance from the cage. In survey 4 there were fewer species that showed a significant effect of distance (Figure 5.4-66). *Schistomeringos loveni* and *Nebalia* sp. again showed positive relationships with proximity to the cage; *Nebalia* sp. was far more abundant at the 0 m cage site (2306 ind. m^{-2}) compared to all other distances (<20 ind. m^{-2}) and *Schistomeringos loveni* was only found at 0 (62 ind. m^{-2}), 50 (178 ind. m^{-2}) and 100 m (25 ind. m^{-2}) sites. In contrast, the flabelliderid *Diplocirrus* sp1., showed a positive relationship with distance from the cage. In survey 5 only *Schistomeringos loveni* and *Pseudopolydora cf. paucibranchiata* showed a significant effect of distance; *Schistomeringos loveni* was more abundant closer to the cage and *Pseudopolydora cf. paucibranchiata* was more abundant further from the cage (Figure 5.4-67). In survey 6, when faunal abundance declined throughout the harbour, *Schistomeringos loveni* remained reasonably abundant at sites closer to the cage, whilst *Pseudopolydora* cf. *paucibranchiata* was more common at sites distant from the cage (Figure 5.4-68).

The unconstrained ordination from the LVM shows the separation of samples based on their species composition and relationship with distance from the cage (Figure 5.4-69). In survey 1, species composition at 0 m from the cage was highly variable and largely distinct from the other distances, likely due to variability between the transects. Of the most important species identified in the LVM, *Nebalia* sp., *Schistomeringos loveni* and *Capitella* sp. were characteristic of the assemblage at 0 m, with *Schistomeringos loveni* also common at 50 m., and *Samythella* sp. and *Euchone varibilis* were both common from 50-500 m. In survey 3 there was even greater overlap in species composition across the distances except for some samples at the cage (0 m) and most distant (500 m) sites. There were several taxa, notably *Nebalia* sp. *Schistomeringos loveni, Charcotia australiensis* and *Samythella* sp. far more abundant at 0 m, but in only one of the transects. Two of the most common species, *Pista australis* and *Pseudopolydora* cf. *paucibranchiata* were common across all distances, while *Branchiomma* sp was only abundant at the 500 m from cage sites. A similar pattern was again observed in survey 4 with very large overlap in species composition over distance. *Nebalia* sp. were highly abundant at one of the 0 m sites, *Diplocirrus* sp1. was only abundant at the 500 m sites and *Pista australis* and *Pseudopolydora cf. paucibranchiata* were common across all distances. In survey 4, there was separation in species composition between 0 m and 250 and 500 m, but with overlap for the intermediate distances. *Schistomeringos loveni* was more abundant at the 0-50 m sites and *Pseudopolydora* cf. *paucibranchiata* was more abundant at 100-500 m. Species such as *Pista australis* and *Echinocardium cordatum* were common across all distances driving the overlap seen in the ordination. In survey 6 *Schistomeringos loveni* and *Pseudopolydora* cf. *paucibranchiata* were again the key species driving the ordination. *Schistomeringos loveni* was only found at the 0-100 m sites and *Pseudopolydora cf. paucibranchiata* was present at all distances but was more common between 50 and 500 m.

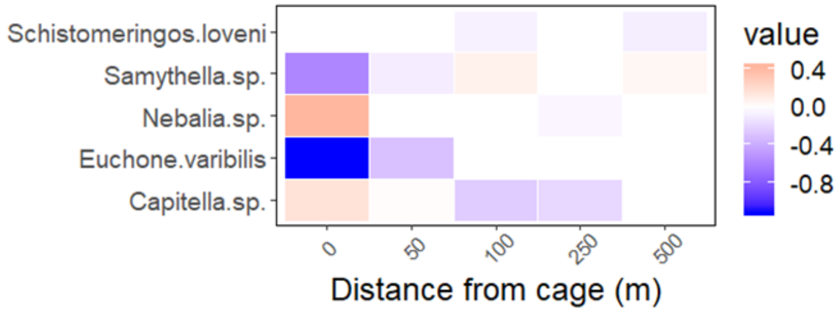


Figure 5.4-64 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Strahan lease in survey 1.

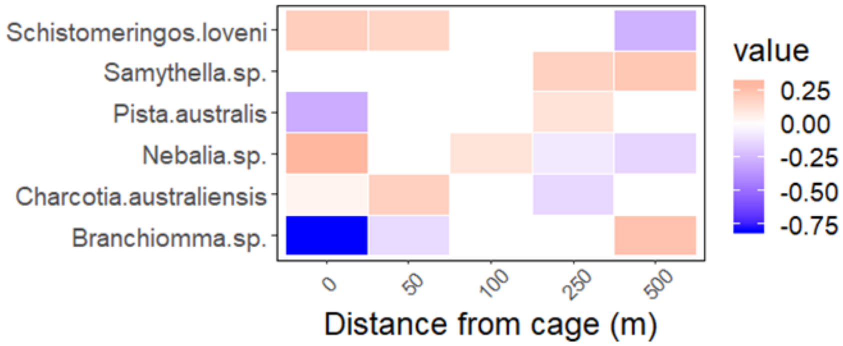


Figure 5.4-65 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Strahan lease in survey 3.

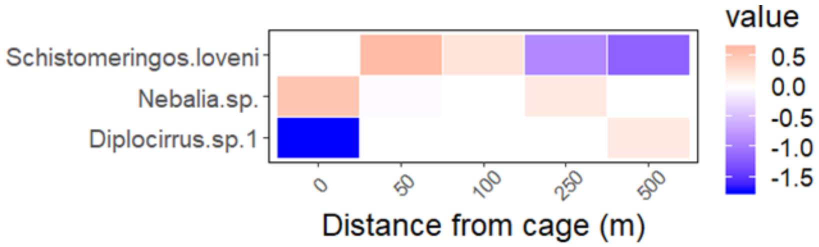


Figure 5.4-66 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Strahan lease in survey 4.

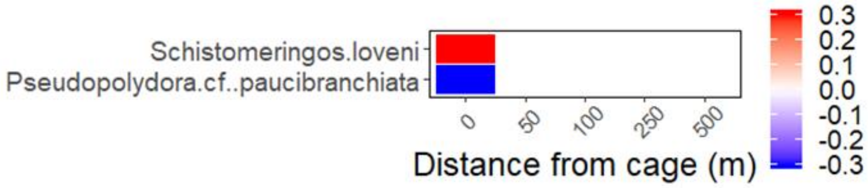


Figure 5.4-67 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Strahan lease in survey 5.

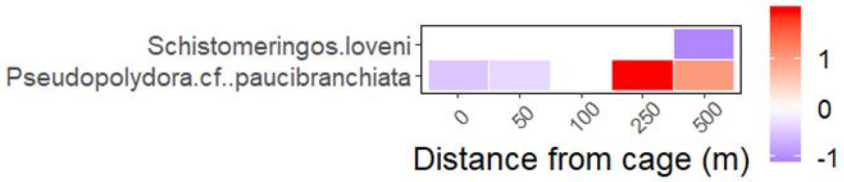


Figure 5.4-68 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Strahan lease in survey 6.

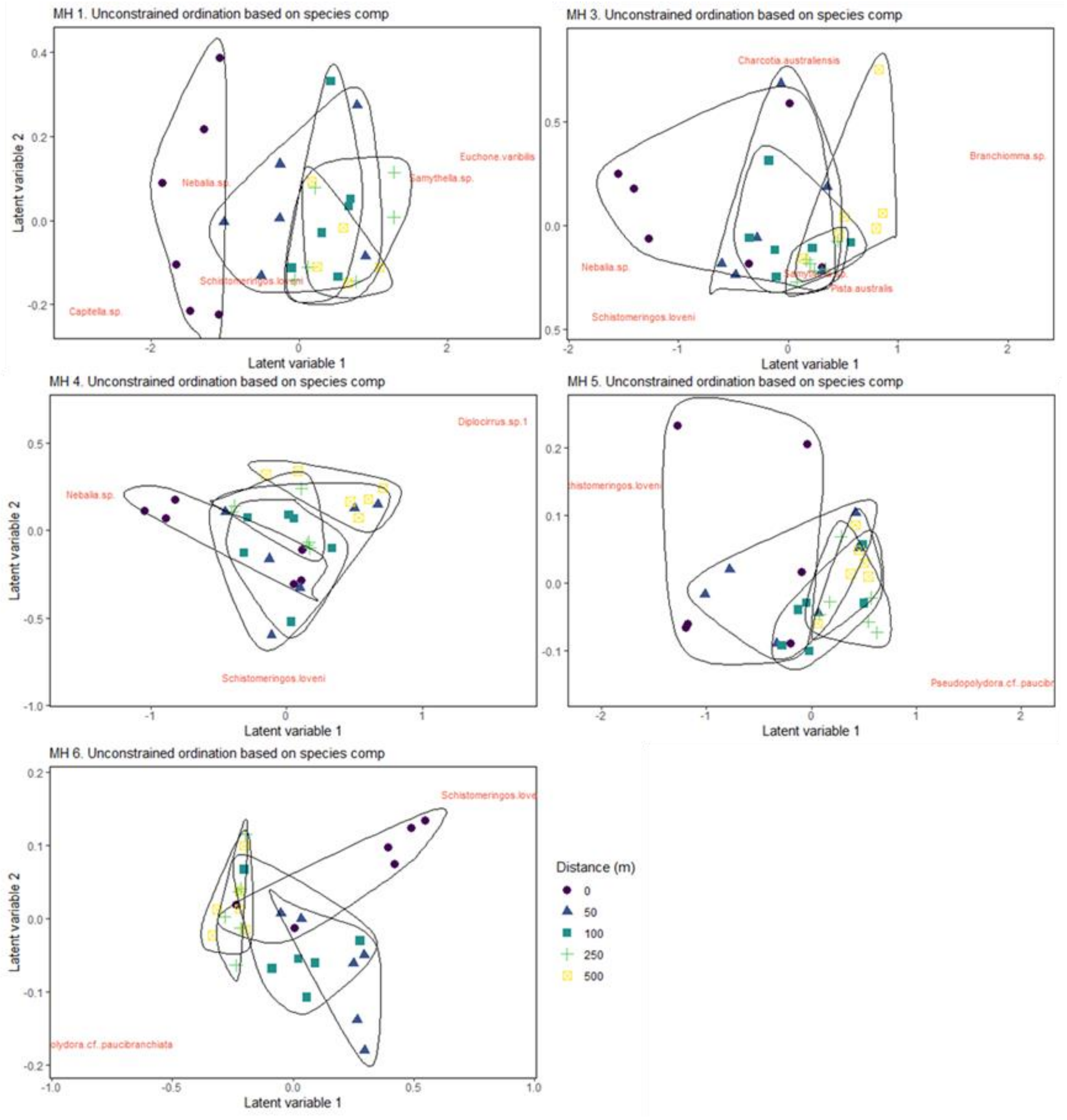


Figure 5.4-69 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1, 3-6 (MH 1 – MH 6) at the Strahan lease. The Strahan lease was not sampled in survey 2. Samples were collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and distance from the origin are more correlated.

Franklin

In survey 1 *Schistomeringos loveni* was by far the most abundant species, and together with *Nebalia* sp. showed a positive relationship with proximity to the cage (Figure 5.4-70). In contrast, *Samythella* sp., *Pseudopolydora* cf. *paucibranchiata* and *Pista australis* showed positive relationships with distance from the cage; all three species were absent from samples taken from 0 and 50 m from the cage. In survey 2, a similar pattern was observed, but *Schistomeringos loveni* and *Nebalia* sp. peaked in abundance further from the cage (50-100 m vs 0-50 m in survey 1). *Pseudopolydora cf. paucibranchiata*, *Pista australis* and *Echinocardium cordatum* showed positive relationships with distance from the cage; all three species were most abundant at 250-500 m from the cage (Figure 5.4-71). In survey 3 the patterns were less clear, with *Pseudopolydora* cf. *paucibranchiata*, *Pista australis*, *Echinocardium cordatum*, *Branchiomma* sp. all showing positive relationships with distance from the cage (Figure 5.4-72). In survey 4 *Nebalia* sp., *Charcotia australiensis* and *Capitella* sp. all showed positive relationships with proximity to the cage; *Nebalia* sp. was most abundant at 0 m, and *Charcotia australiensis* and *Capitella* sp. were most abundant at 50 and 100 m. *Schistomeringos loveni* showed a positive relationship with intermediate distances. It was common from 50-250 m but most abundant at 100 m from the cage (Figure 5.4-73). *Pseudopolydora* cf. *paucibranchiata*, *Pista australis, Branchiomma* sp. were absent at 0 and 50 m and most common between 100-500 m from the cage. In survey 5, *Schistomeringos loveni* was again most abundant at the intermediate distances and *Pseudopolydora* cf. *paucibranchiata*, *Pista australis* and *Branchiomma* sp. were again most common at sites farther from the cage (Figure 5.4-74). In survey 6, the results from the multivariate GLM were not able to be interpreted due to the major decline in faunal abundance and diversity and as such the plot with LASSO fits is not included; only five individuals were found across 30 grab samples.

The unconstrained ordination from the LVM shows the separation of samples based on their species composition and relationship with distance from the cage (Figure 5.4-69). In survey 1, species compositions at 0 and 50 m were distinct from the assemblage found at 100-500 m from the cages (Figure 5.4-75). Of the most important species identified in the LVM, *Schistomeringos loveni* and *Nebalia* sp., were characteristic of the assemblages from 0-50 m. Both species were also common at 100 m (and 250 m for *Schistomeringos loveni*), but *Pseudopolydora* cf. *paucibranchiata*, *Samythella* sp. and *Pista australis*, absent from 0-50 m, were identified as most characteristic of the assemblages between 100-500 m from the cages. In survey 2, the assemblages at 0-100 m overlapped but were distinct from the assemblage at 250-500 m. *Schistomeringos loveni* and *Nebalia* sp. were identified as characteristic of the 0-100 m group and *Pseudopolydora* cf. *paucibranchiata*, *Pista australis* and *Echinocardium cordatum* characteristic of the assemblage at 250-500 m. In survey 3 the assemblage at 500 m remained distinct from that at 0 m, but there was overlap in the assemblages at the intermediate distances. *Schistomeringos loveni* was the only abundant species at 0 m (301 ind. m^{-2}) and 50 m (1012 ind. m^{-2}). At 100 m (279 ind. m^{-2}) it was also the dominant taxa but with *Pista australis* (12 ind. m^{-2}) and *Pseudopolydora* cf. *paucibranchiata* (10 ind. m^{-2}) also present. *Schistomeringos loveni* was also present at 250 (207 ind. m^{-2}) and 500 m (27 ind. m^{-2}), but *Pista australis, Pseudopolydora* cf. *paucibranchiata*, *Branchiomma* sp. and *Echinocardium cordatum* were also common at these distances and identified as the characteristic species. In survey 4, *Schistomeringos loveni, Nebalia* sp., *Charcotia australiensis* and *Capitella* sp. were characteristic of the assemblages found in closer proximity to the cages (e.g. 0-50 m) and *Pista australis*, *Pseudopolydora* cf. *paucibranchiata* and *Branchiomma* sp. were characteristic of the assemblages more distant (e.g. 250-500 m) from the cages; at 100 m there was a mix of all these species. In survey 5, the assemblages at 0-100 m were again distinct to the assemblages found at 250 – 500 m. *Pista australis, Pseudopolydora cf. paucibranchiata* and *Branchiomma* sp. were characteristic of the assemblages at 250-500 m and *Schistomeringos loveni* was the dominant species characteristic of assemblages at 0-100 m. In survey 6, with only five individuals found across 30 grabs the ordination could not be run.

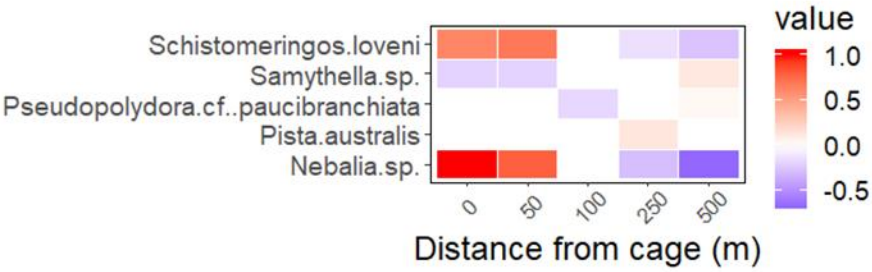


Figure 5.4-70 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Franklin lease in survey 1.

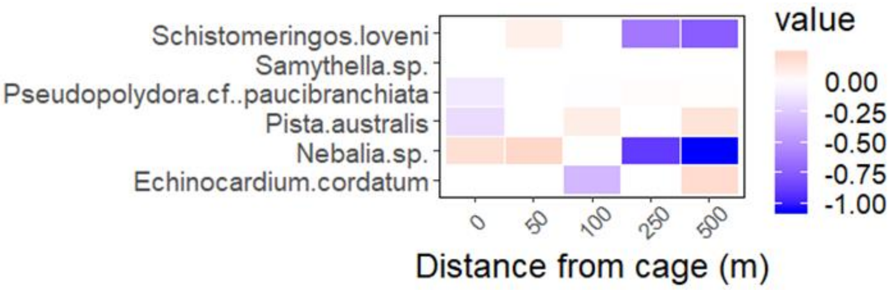


Figure 5.4-71 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Franklin lease in survey 2.

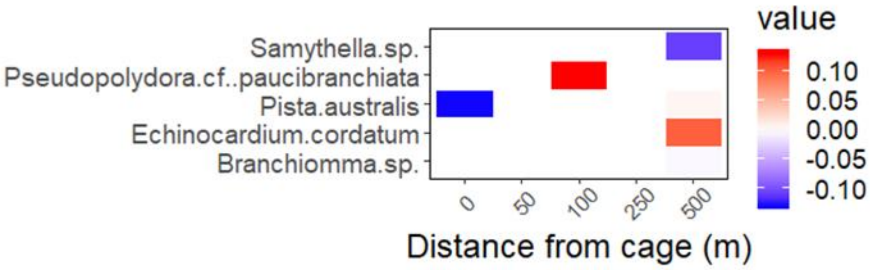


Figure 5.4-72 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Franklin lease in survey 3.

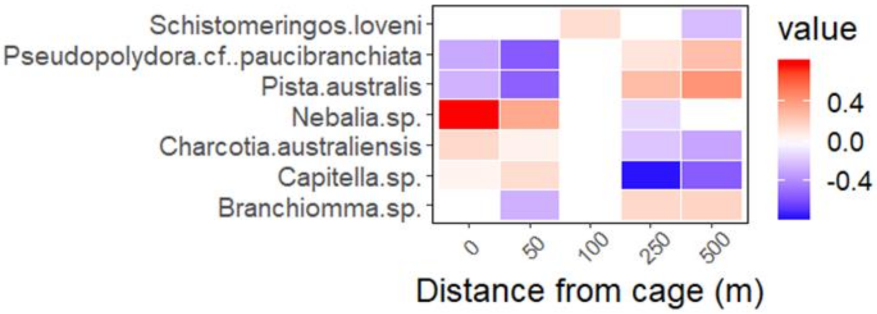


Figure 5.4-73 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Franklin lease in survey 4.

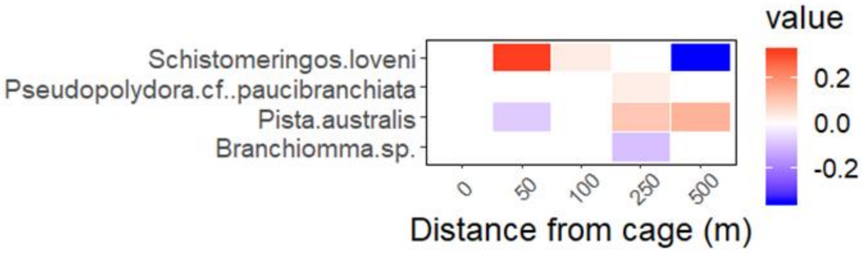


Figure 5.4-74 Species contribution to assemblage composition (Fourth corner coefficients) with LASSO penalised fits for species that showed a significant effect with distance from the Franklin lease in survey 5.

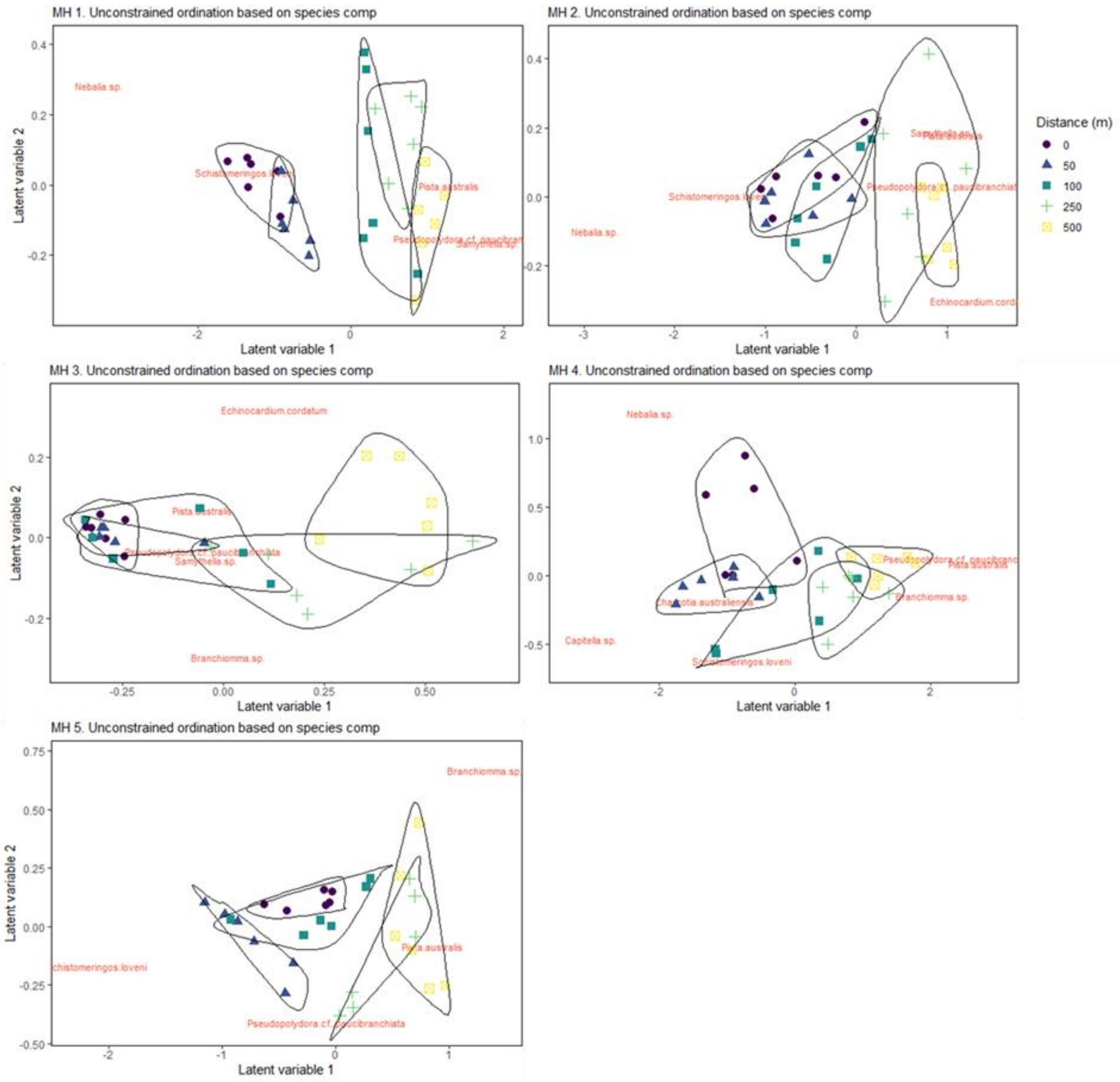


Figure 5.4-75 Biplots showing latent variable model (LVM) unconstrained ordination of samples and species coefficients for surveys 1-5 (MH 1 – MH 5) at the Franklin lease. Samples collected at different distances along the transects from the cage are identified by colour and symbol. Only the ten most important species are plotted (those with the strongest response to the latent variables). Species in the same direction and distance from the origin are more correlated. Due to the scarcity of fauna at this lease in survey 6 the model was not able to converge and create an ordination.

5.4.1.3 Visual

Table Head Central. There was a significant effect of distance on ROV health scores in all surveys, with 0 m sites clearly having the lowest scores (Figure 5.4-76; Table 5.4-2). There was no significant difference detected in ROV scores between the 50 m or 100 m sites and the controls in any of the surveys (Table 5.4-2). There was no clear difference in the ROV scores across surveys at the sites closer to, or more distant from the cages (Figure 5.4-80; Table 5.4-2).

Gordon. There was a very clear and significant effect of distance on ROV health scores in all surveys, with 0, 50 and often 100 m sites scoring the lowest (Figure 5.4-77; Table 5.4-2). With the exception of the difference between 100 m and the controls in survey 1, ROV scores were significantly lower at both 50 m and 100 m sites compared to the controls in each of the surveys (Table 5.4-1). There was no clear difference in the ROV scores across surveys at the sites closer to, or more distant from the cages (Figure 5.4-80; Table 5.4-2).

Strahan. The ROV health scores were lower at the 0 m cage site in each of the three surveys assessed, but the effect of distance was not significant in survey 4 (Figure 5.4-78; Table 5.4-2). There was no significant difference detected in ROV scores between the 50 m or 100 m sites and the controls in each of the surveys (Table 5.4-2) and there was no clear difference in the ROV scores across surveys at the sites closer to, or more distant from the cages (Figure 5.4-80; Table 5.4-2).

Franklin. There was a very clear and significant effect of distance on ROV health scores, with 0, 50 and often 100 m sites scoring the lowest in in each of the surveys (Figure 5.4-77; Table 5.4-2). Although the effect of distance was not significant (p = 0.052) in survey 6, the mean ROV health score at all distances from 0-500 m was negative. ROV scores were lower at both 50 and 100 m sites compared to the controls in each of the surveys, but the difference was not significant between 50 m and the controls in surveys 2 and 5, and between 100 m and the controls in surveys 2, 3 and 5 (Table 5.4-2). There was no clear difference in the ROV scores across surveys at the sites closer to, or more distant from the cages (Figure 5.4-80; Table 5.4-2).

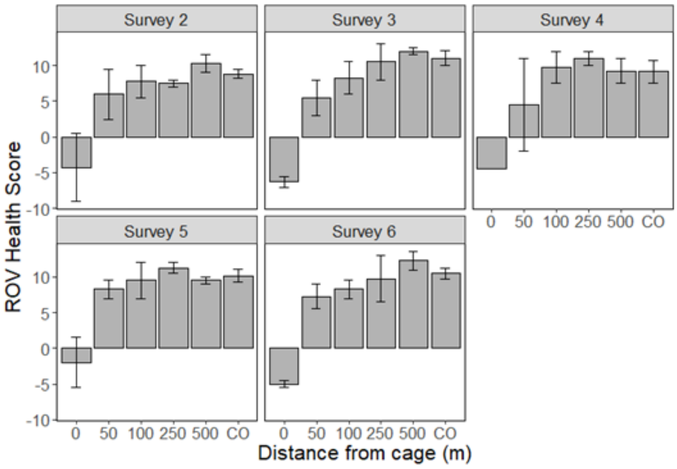


Figure 5.4-76 ROV health score (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease.

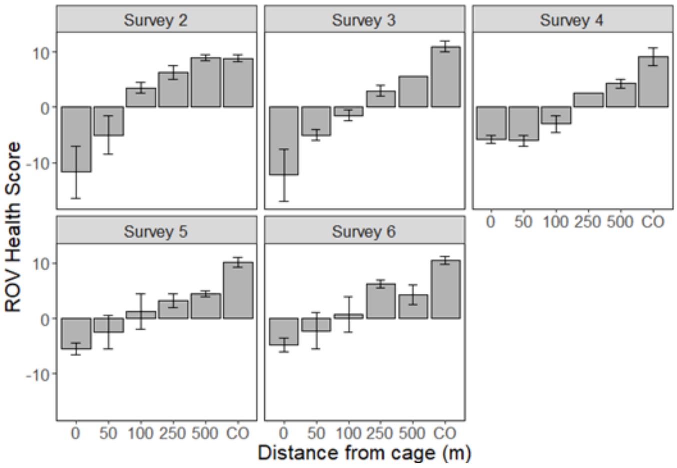


Figure 5.4-77 ROV health score (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease.

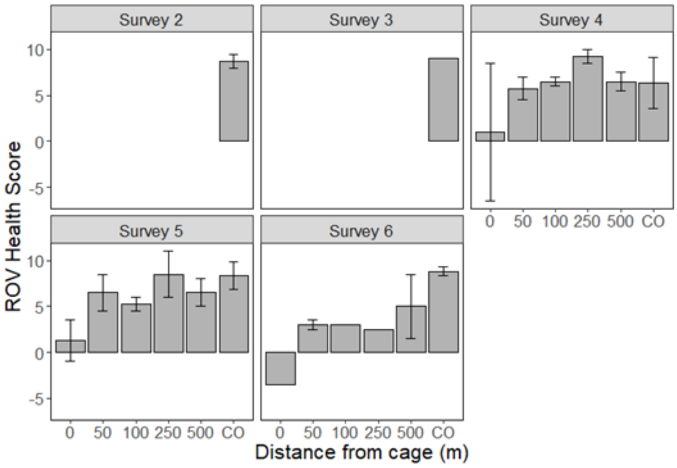


Figure 5.4-78 ROV health score (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease.

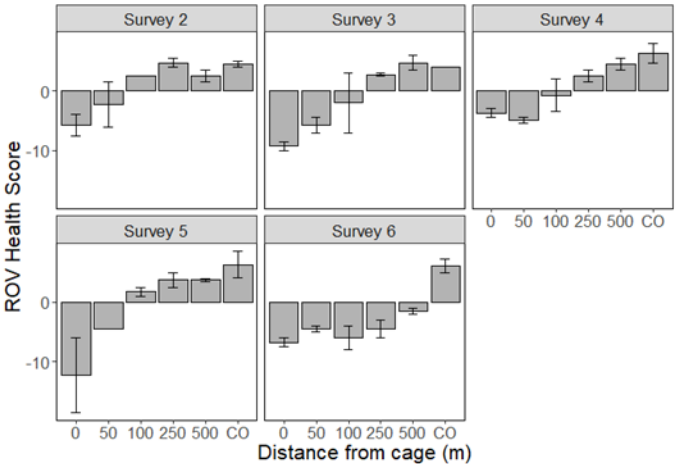


Figure 5.4-79 ROV health score (average ±SE) with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease.

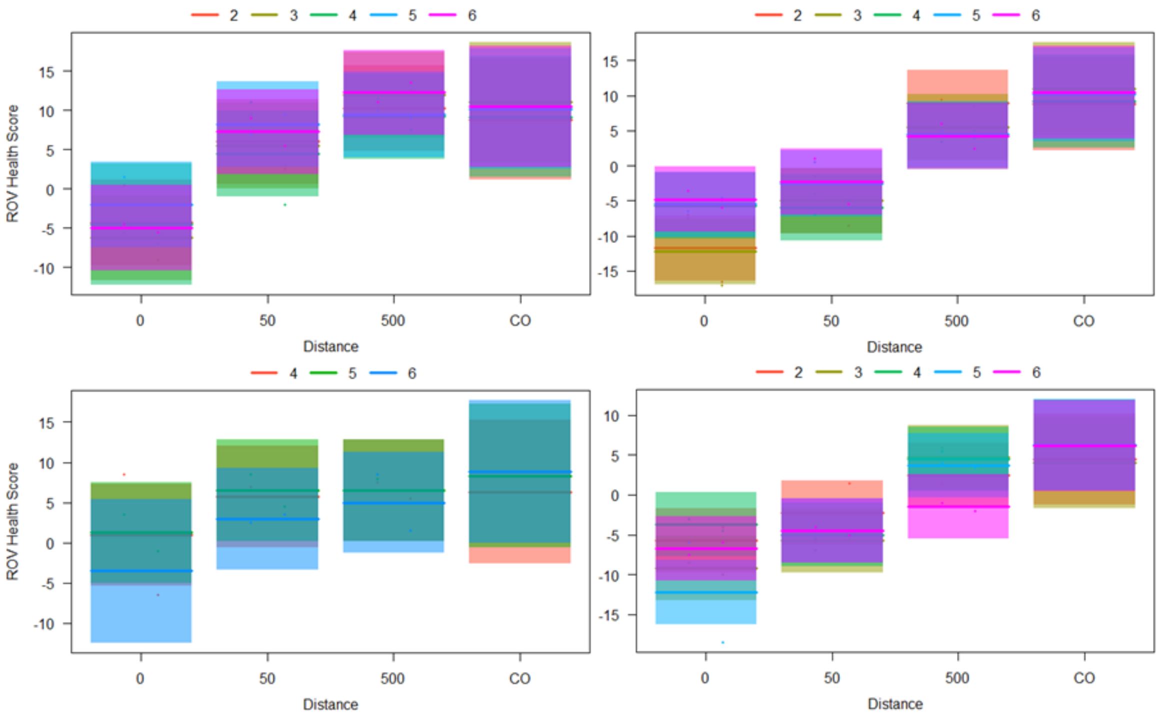


Figure 5.4-80 ROV health score (predicted mean ± 95% confidence intervals) from the GLM comparing surveys at the two distances closest to the cages (0 and 50 m) and at the distances furthest from the cages (500,and CO) at the Table Head Central (top left), Gordon (top right), Strahan (bottom left) and Franklin (bottom right) leases.

5.4.1.4 Sediment processes

Table Head Central. Sediment respiration measured via dissolved oxygen consumption decreased as a function of distance from the cage in both surveys (Figure 5.4-81,Table 5.4-3). Similarly, respiration measured as dissolved inorganic carbon production (i.e. principally carbon dioxide produced during respiration) decreased as a function of distance from cage in both surveys (Figure 5.4-81,Table 5.4-3).

Fluxes of ammonium also decreased as a function of distance from the cage in both surveys and were much higher at the 0 m cage site (Figure 5.4-81, Table 5.4-3). Nitrate + nitrite fluxes were also much greater and directed into the sediment (i.e. uptake) at all of the cage sites in both surveys (Figure 5.4-81, Table 5.4-3). Fluxes of phosphate were elevated at the cage site and negligible at all other distances in survey 4 (Figure 5.4-81, Table 5.4-3). In survey 5, phosphate fluxes were small and variable all distances.

Gordon. Sediment respiration measured via dissolved oxygen consumption decreased as a function of distance from cage in both surveys (Figure 5.4-82, Table 5.4-3), but the pattern was more evident and significant in survey 4. Respiration measured as dissolved inorganic carbon production was higher at the cage site in survey 4, but otherwise there was no distinct pattern with distance in any of the surveys (Figure 5.4-82, Table 5.4-3).

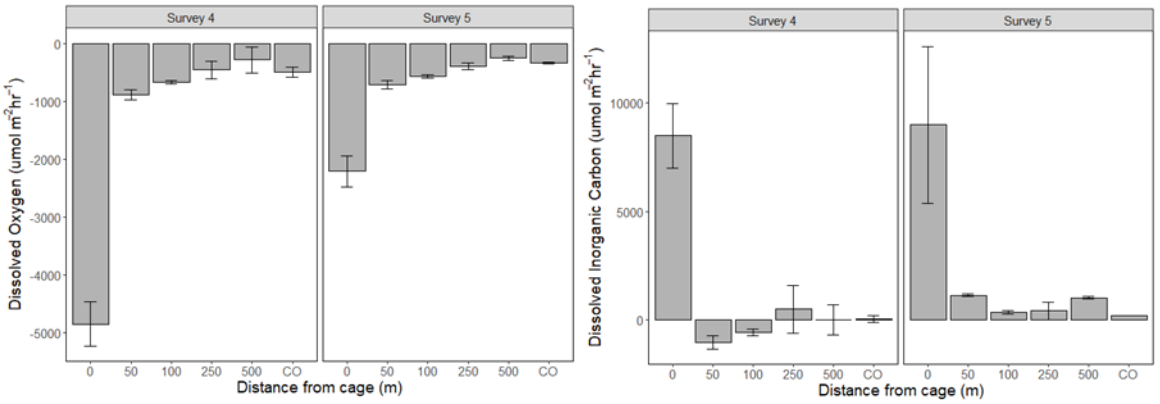
Fluxes of ammonium also decreased as a function of distance from cage in both surveys (Figure 5.4-82, Table 5.4-3), with much higher fluxes relative to other distances recorded at 0 and 50 m in survey 4 and 0 m in survey 5. There was a significant effect of distance for nitrate + nitrite in survey 4, with fluxes (out of the sediment) increasing with proximity to the cages. In survey 5, nitrate + nitrite fluxes were generally small and variable with no clear effect of distance. Fluxes of phosphate were elevated at the cage site and negligible, and directed into the sediments at all other distances in survey 4 (Figure 5.4-82, Table 5.4-3). In survey 5, phosphate fluxes were small and variable at all distances.

Strahan. Sediment respiration decreased as a function of distance from the cage in both surveys, but the pattern was more distinct in survey 4 (Figure 5.4-83,Table 5.4-3). Respiration measured as dissolved inorganic carbon production (i.e. principally carbon dioxide produced during respiration) decreased as a function of distance from the cage in survey 4 (Figure 5.4-83,Table 5.4-3). In survey 5 dissolved inorganic carbon production was variable and there was no clear pattern with distance.

Fluxes of ammonium were also higher in closer proximity to the cages, notably at 0 m in survey 4 and 0 and 50 m in survey 5 (Figure 5.4-83, Table 5.4-3). Nitrate + nitrite fluxes were directed into the sediment (i.e. uptake) at all of the sites in both surveys (Figure 5.4-83, Table 5.4-3). Although the effect of distance for nitrate + nitrite flux was not significant in either survey, the rate of uptake appeared higher closer to the cage in survey 4. Although fluxes of phosphate were highly variable, the effect of distance was significant in both surveys (Figure 5.4-83, Table 5.4-3). In survey 4, phosphate release from the sediment was elevated at the cage site, and in survey 5, phosphate flux was directed into the sediment closer to the cage and out of the sediment at the more distant sites.

Franklin. Sediment respiration measured via dissolved oxygen consumption decreased as a function of distance from the cage in both surveys (Figure 5.4-84, Table 5.4-3), but the pattern was more evident and significant in survey 4. Dissolved inorganic carbon data was only available in survey 4. There was a clear effect of distance, with dissolved inorganic carbon production elevated at the cage site relative to the other distances (Figure 5.4-84, Table 5.4-3).

The effect of distance was also significant for fluxes of ammonium, which decreased as a function of distance from the cage in both surveys (Figure 5.4-84, Table 5.4-3), and was the biggest change evident between 0 and 50 m in both surveys. Like Strahan, fluxes of nitrate and nitrite were directed into the sediments at Franklin. In survey 4, the effect of distance on nitrate and nitrite was not significant (Figure 5.4-84, Table 5.4-3). Uptake appeared higher but more variable at the sites closer to the cage (i.e. 0 - 100 m). In survey 5 the effect of distance was significant, due to the much higher flux into the sediment at the cage site. The effect of distance was significant for phosphate in both surveys, with elevated fluxes out of the sediment at 0 m relative to the other distances (Figure 5.4-82, Table 5.4-3).



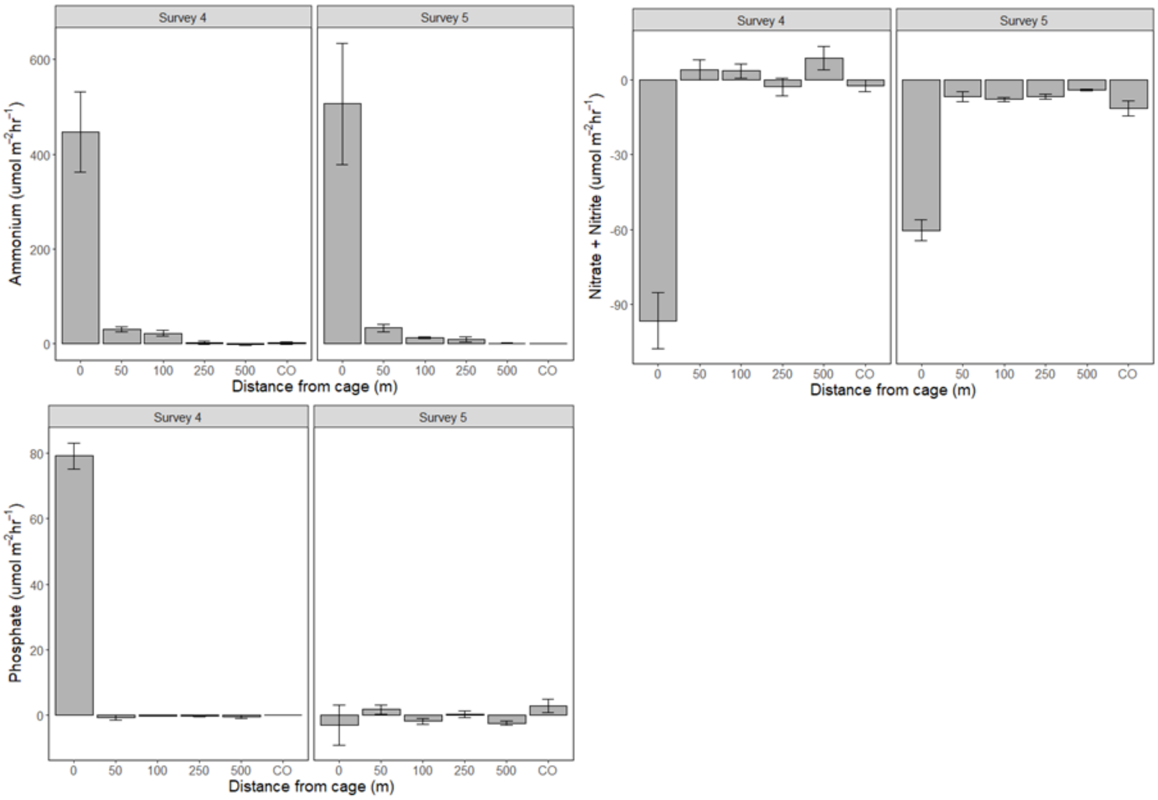
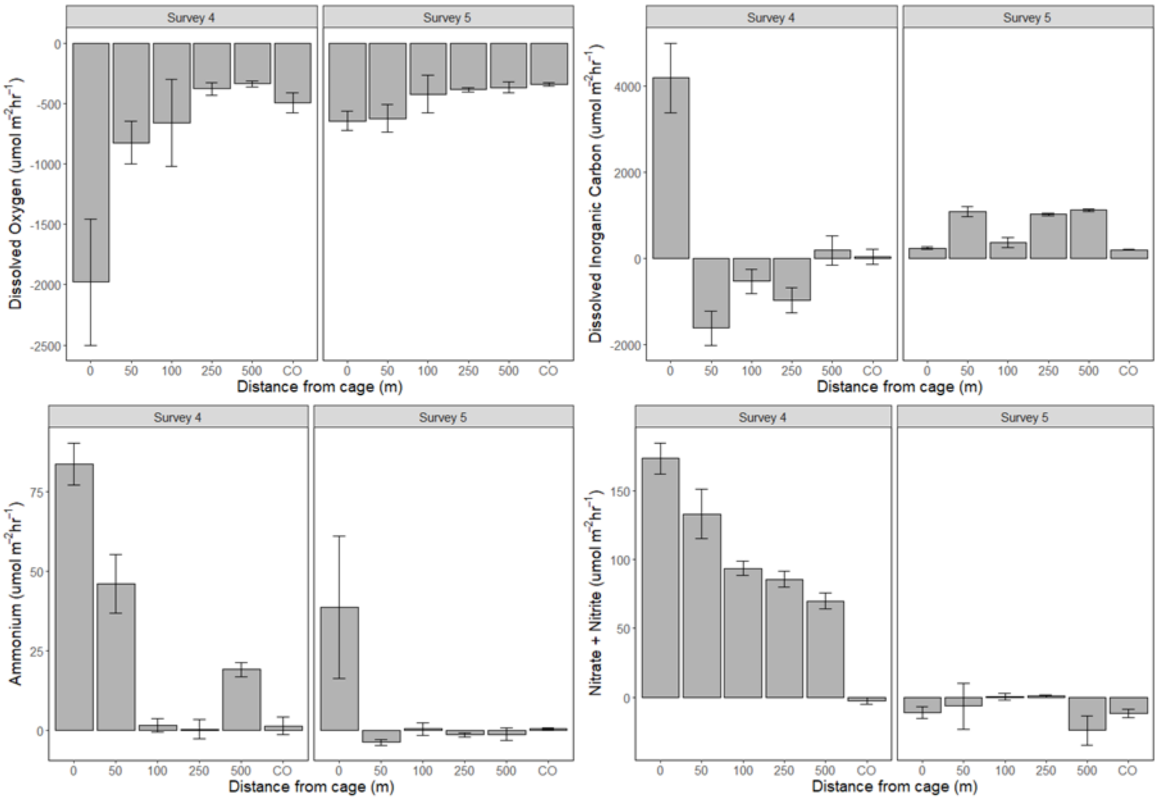


Figure 5.4-81 Sediment fluxes (umol m^{-2} h^{-1}; average ± SE) of dissolved oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages (on transects) and at outer controls (CO) for each survey at the Table Head Central lease. Sediment flux measurements were only conducted in surveys 4 and 5 and on one transect.



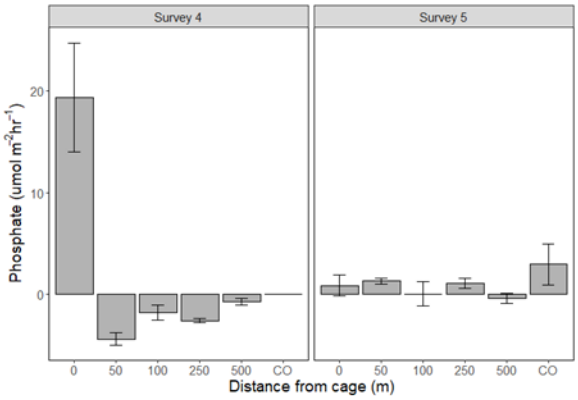


Figure 5.4-82 Sediment fluxes (umol m^{-2} h^{-1}; average ± SE) of dissolved oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages (on transects) and at outer controls (CO) for each survey at the Gordon lease. Sediment flux measurements were only conducted in surveys 4 and 5 and on one transect.

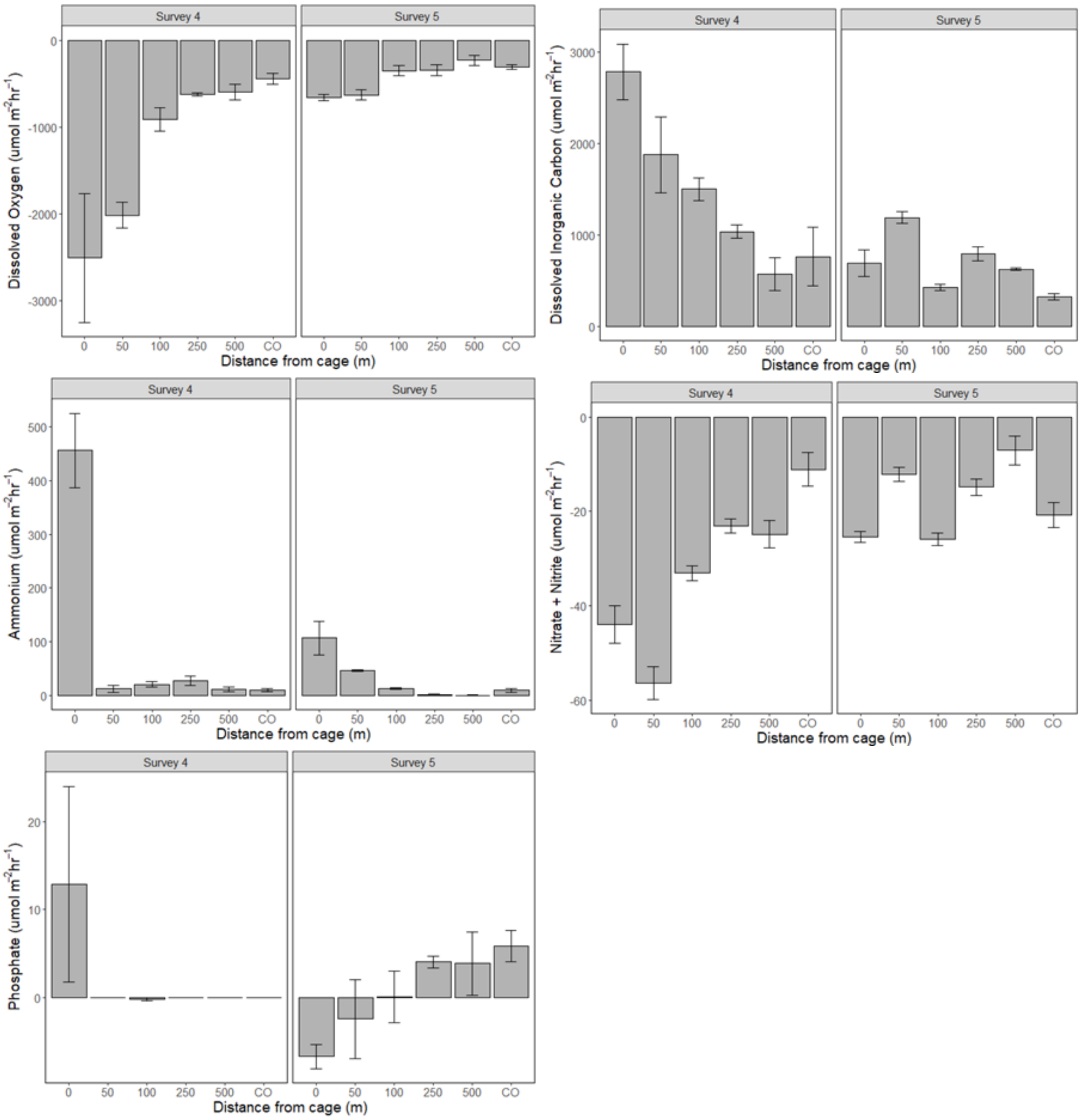


Figure 5.4-83 Sediment fluxes (umol m^{-2} h^{-1}; average ± SE) of dissolved oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages (on transects) and at outer controls (CO) for each survey at the Strahan lease. Sediment flux measurements were only conducted in surveys 4 and 5 and on one transect.

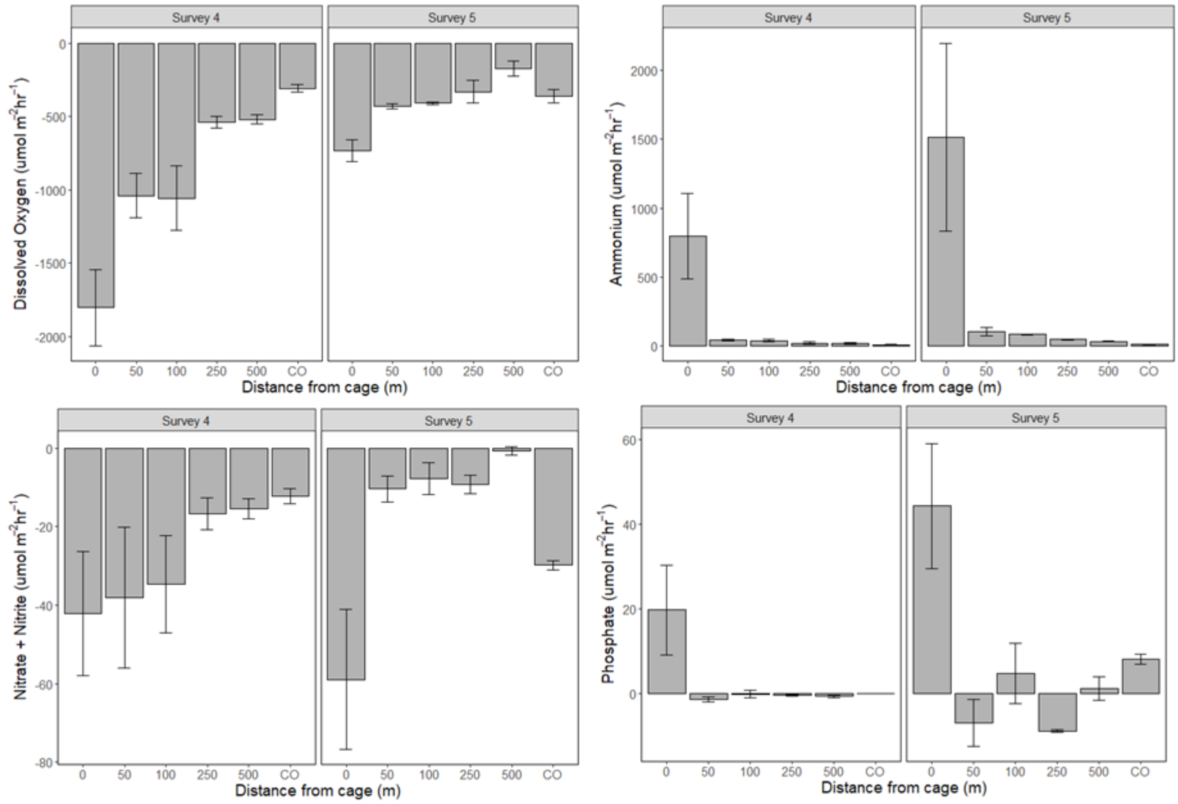


Figure 5.4-84 Sediment fluxes (umol m^{-2} h^{-1}; average ± SE) of dissolved oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages (on transects) and at outer controls (CO) for each survey at the Franklin lease. Sediment flux measurements were only conducted in surveys 4 and 5 and on one transect. Dissolved inorganic carbon data was not available in survey 5 except for the external control sites.

5.4.2 Discussion

Benthic Community

Macquarie Harbour is ecologically very different from farming areas in southern Tasmania and elsewhere. The sediments in this region are inherently depauperate (O'Connor et al. 1996; Talman et al. 1996; Edgar et al. 1999), which was once again evident in this study by low faunal abundance and species richness at the Macquarie Harbour control sites compared to those of Storm Bay and the Southern Channel. There was an average of 5-29 individuals per grab depending on the location in Macquarie Harbour, compared with an average of 129 individuals per grab at the Lippies lease control sites and 94 – 115 individuals per grab at the Trumpeter and Storm Bay 1 control sites. In Macquarie Harbour polychaetes dominated, making up between 70-80% of the community, followed by crustaceans (2-19%), molluscs (6-15%) and echinoderms (2-16%). In Macquarie Harbour, the most abundant taxa at the control sites were the terebellid polychaetes *Pista australis* and *Amaena cf. trilobata,* sabellid polychaetes *Branchiomma* sp. and *Euchone varibilis,* spionid polychaete *Pseudopolydora* cf. *paucibranchiata*, ampharetid polychaete *Samythella* sp., dorvilleid polychaete *Schistomeringos loveni*, trichobranchid polychaete *Terebellides narribri,* orbiniid polychaete *Leitoscoloplos bifurcatus*, amphipod *Byblis mildura*, ostracod *Philomedid* sp., bivalve *Parathyasira resupina* and echinoderm *Echinocardium cordatum.*

Response to enrichment

The overall benthic community changed discernibly with increasing distance from the cages; however, the pattern varied between leases and surveys. The variation appeared largely consistent with the influence of farm intensity and history, but local conditions were also clearly important in determining the magnitude of this response.

The biodiversity metrics, species richness (S), total abundance (N), and Shannon-Weaver Diversity Index (H') showed a response pattern generally consistent with both prior research and guidelines established for the south (Macleod and Forbes 2004), albeit that the expected changes occurred over a greater spatial scale. In Macquarie Harbour total abundance generally peaked at 50-100 m from the cage, except for the Strahan site where the peak in abundance was more consistent with previous observations from the Southern regions and typically at the 0 m cage site.

Though the pattern for species richness was more variable, it also increased with distance away from the cage; at Table Head Central and Strahan species richness was typically higher closer to the cage compared with the pattern seen at Gordon and Franklin. This response broadly aligns with the expectations outlined in the Species-Abundance-Biomass curves produced by Pearson and Rosenberg (1978, Figure 5.4-85) and reflects the zone of increased diversity which often occurs at distance from an enrichment source due to the crossover between tolerant and sensitive species. This result accords with previous research and expectation that impacted sites would be less diverse (Keeley et al. 2015; Macleod and Forbes 2004; Pearson and Rosenberg 1978; Figure 5.4-85).

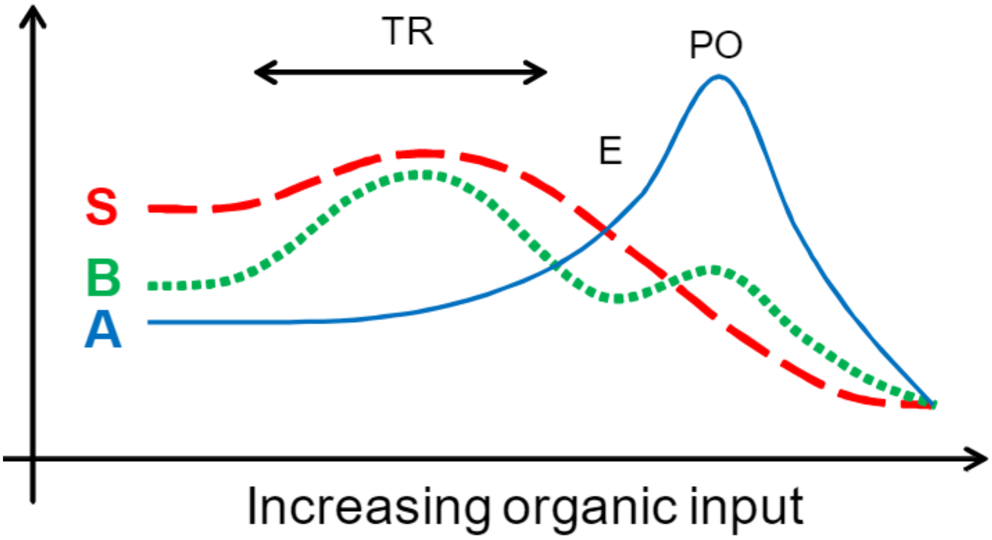


Figure 5.4-85 Pearson and Rosenberg's generalised SAB (model on effects of increased organic matter loadings on communities of benthic macrofauna.: S = Species, A = Abundance, B = Biomass, TR = Transition Zone, E = Ecotone Point, PO = Peak of opportunists (after Pearson & Rosenberg, 1978, Figure 2).

However, the naturally depauperate nature of Macquarie Harbour (O'Connor et al. 1996) places it at odds with some of the key premises set out in the monitoring and assessment guidelines established by Macleod and Forbes (2004). For example, under the guidelines a Shannon-Weaver Diversity index value of two was identified as indicative of a relatively unimpacted site, with major degrading effects seen at a diversity index under 1 (Macleod et al. 2004; Macleod and Forbes 2004; Hargrave et al. 2008). However, in Macquarie Harbour the mean diversity index (H') value was less than 2 across all distances, including controls, leases and surveys. The mean diversity index decreased with increasing distance into the harbour ranging from ~ 1.5-2 at the control sites in proximity to the Table Head Central and Gordon leases compared with 0-1.5 at the control sites in the proximity to Strahan and Franklin. Using the guidelines (Macleod & Forbes 2004), all the remaining sites would be considered moderately or majorly impacted. Similarly, impact criteria based on total abundance and species richness established through the guidelines are not likely to be suitable for Macquarie Harbour given the depauperate nature of the background ecology.

Whilst the guidelines recommend using the biotic categorisation in conjunction with a suite of other indicators and not as standalone thresholds, the results of this study highlight the biological impact categorisation and recommendations based on conditions in southern Tasmania do not directly apply to Macquarie Harbour, or at least need to be interpreted in a different manner. Furthermore, the results suggest any impact criteria established for Macquarie Harbour would need to take the specific location and background environmental conditions into consideration.

Whilst assessment of species diversity and abundance can be used to provide an understanding of the major changes in the overall community structure associated with organic enrichment and farming, understanding changes in taxonomic composition and functional ecology can provide greater insight into how the broader ecology and ecosystem function might have changed (Figure 5.4-85).

In Macquarie Harbour, burrowing and epibenthic fauna tend to dominate under/around the cages, with tube building and filter/suspension feeding fauna more common where conditions were slightly less depositional. The dorvilleid *Schistomeringos loveni* and leptostracean *Nebalia* sp. dominated at the 0 m cage sites, collectively making up 72% of individuals found at these sites across all leases and surveys. *Nebalia* sp. have been shown elsewhere to be prolific opportunists, frequently found underneath salmon cages (Sasaki & Oshino 2004; Brooks & Mahnken 2003; Keeley et al. 2012). Although not common in Macquarie Harbour, *Capitella* sp. were also associated with the enriched cage sites with peak abundances at 0-50 m. Similarly, the amphipod *Charcotia australiensis*, though not particularly common, was typically only found at distances closer to the source of enrichment. All these species are characterised by their abilities to either move through the water column and/or burrow into the sediment and alter their behaviour in response to changes in the physicochemical properties and organic enrichment of the sediment. They are also reproductively opportunistic, able to colonise rapidly and all year round.

The colony forming dorvilleid *Ophryotrocha shieldsi* was rarely detected in grabs, but from video footage was only recorded at 0 m and occasionally at 50 m from the cage. This species was typically observed in conjunction with the presence of *Beggiatoa*. The colonies sat above the sediment, and the “loose” nature of the colonies and small size of the worms likely explains why they were rarely collected in the grabs. The association of *O. shieldsi* with *Beggiatoa*-like bacterial mats is consistent with previous observations of a similar species, *O. cyclops*, which also forms colonies largely associated with *Beggiatoa* at aquaculture sites (Salvo et al. 2015). Previous studies have suggested that Ophryotrocha species may feed on these bacterial mats (Salvo et al. 2015; Taboada et al. 2013). Regardless, the presence of this species under and adjacent to the cages suggests it is highly adapted to the disturbed sediment and bottom water conditions associated with high levels of organic enrichment.

While the other dorvilleid, *Schistomeringos loveni* was a dominant taxon at the 0 m cage site it was typically far more abundant further out from the cages, with peak abundance at 50-100 m. The exception was at the Strahan lease where peak abundance of *Schistomeringos loveni* was at the 0 m cage site; this likely reflects the lower feed inputs at Strahan (see latter discussion). Like *O. shieldsi*, the presence of *S. loveni* was still clearly associated with the organic enrichment associated with salmonid farming. Many previous studies have suggested *S. loveni* and congeners, are opportunistic species. *S. loveni* was found to be up to 30 times more abundant at organically enriched sites under ropes used to propagate mussels than at reference sites in a New Zealand study (Hartstein & Rowden 2004). Whilst sediments under mussel farms may be less enriched than those under salmon farms, these results corroborate

Figure 5.4-86 Average abundance (ind. per grab) of key taxa with distance from the cages (on transects) and at the outer controls (CO) sites across surveys at a) Table Head Central, b) Gordon, c) Strahan and d) Franklin.

that *S. loveni* responds opportunistically to enrichment. *Schistomeringos annulata* was one of the top three most abundant species close to finfish farm operations (i.e., within 300 m) in a Canadian study (Bright 2001) and *Schistomeringos japonica* has been shown to aggregate underneath fish farms in Japan (Sasaki & Oshino 2004). In this study, *S. loveni* was most abundant at 50-100 m from stocked cages which suggests this species potentially has a tolerance window, enhanced under moderately enriched conditions but sensitive to the highly enriched conditions directly under salmonid cages.

Further out from the cages another group of taxa dominated at intermediate distances, typically peaking in abundance between 50 and 500 m from the cage depending on the lease, survey and feed inputs (Figure 5.4-86). The key species of this group were the terebellid *Pista australis,* spionid *Pseudopolydora* cf. *paucibranchiata,* sabellid *Branchiomma* sp., ampharetid *Samythella* sp. and echinoderm *Echinocardium cordatum*. These species include relatively sessile suspension feeders and surface deposit feeders likely to have a more limited capacity to move in and out of areas as the sediment quality changes; as such they will generally be more sensitive than their mobile counterparts to rapid changes in environmental conditions, including changes in organic enrichment and deterioration of environmental conditions. This spatial pattern of succession is aligned with the spatial transition from deposit to suspension feeders shown to be associated with increasing distance from cages identified in previous research (Macleod & Forbes 2004; Brooks & Mahnken 2003). At the more distant control sites, greater than 1000 m from the leases, the common species were characteristic of a different type of community. Here species such as the bivalve *Parathyasira resupina*, orbiniid *Leitoscoloplos bifurcatus*, sabellid *Euchone varibilis*, terebellid *Amaena cf. trilobata* and amphipod *Bybilus mildura* were characteristic of the community. These species ecologies are consistent with those shown in previous studies to be indicative of less enriched conditions, including studies from Southern Tasmania (Macleod & Forbes 2004; Edgar et al. 2005), which also found species of terrebellids and sabellids, as well as certain amphipods and bivalves to be indicative of lower levels of organic enrichment.

Influence of farm management and the environment

Although there was a clear change in species abundance and composition with distance from the cage and source of enrichment, there was marked variation in this relationship between leases and surveys (Figure 5.4-86). Some of this variation is likely a function of variability in background environmental conditions in Macquarie Harbour e.g., differences in depth, proximity to the ocean, changes in oxygen conditions and differences in farming intensity and history.

In this study, feed input within the leases ranged from <1,000t to >7,500t (per lease) for the 12 months preceding the surveys. There was also a marked difference in farming history, with lease 219 (Gordon) continuously farmed for approximately 15 years whilst the remaining three leases had only been operational (stocked) for less than two years. From an environmental perspective, lease 133 (Table Head) was closest to the entrance of the ocean; thus, recharge of oxygen in bottom waters was more frequent in comparison to lease 266 (Franklin), the most distant from the influence of the ocean. The depth at a lease and sample sites would also influence the exposure to reoxygenated bottom waters (Figure 5.4-87); lease 267 (Strahan) and the associated sample sites lie at 37 - 42m depth and sit adjacent to the deeper hole in the central harbour likely to first experience the denser more saline oxygenated waters following ocean recharge. In contrast, sites at the other leases spanned a greater depth range (Table Head Central: 21-41m; Gordon: 27-43m; Franklin: 26-37m) with the shallower sample sites requiring a larger or more prolonged oceanic recharge before experiencing an increase in oxygen concentrations.

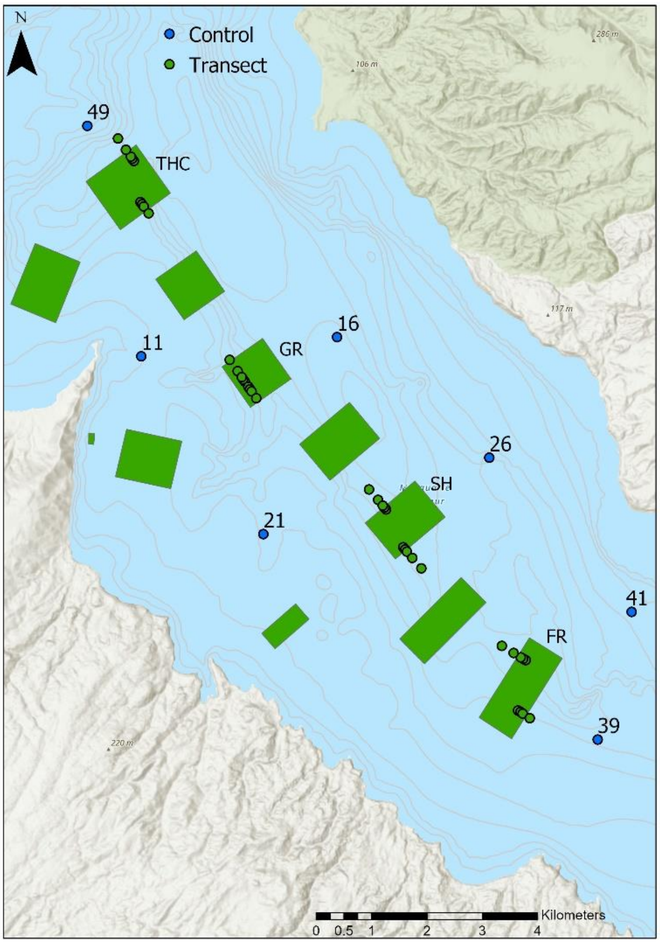


Figure 5.4-87 Map showing the Macquarie Harbour control (blue) and lease transect (blue) sites at Table Head Central (THC, MF 133), Gordon (GR, MF219), Strahan (SH, MF267) and Franklin (FR, MF266). There are two transects at each of the study leases with five sites at 0, 50, 100, 250 and 500 m from the cage on each transect.

When reporting on the first of these comprehensive benthic surveys conducted in January 2015 as part of FRDC 2014-038, Ross et al (2016) highlighted while the presence of both species of dorvilleids in Macquarie Harbour, *Ophryotrocha shieldsi* and *Schistomeringos loveni*, appeared to be a good indicator of organic enrichment, their presence reflected different levels of enrichment. *O. shieldsi* occurred predominately as colonies directly under stocked cages and was only occasionally observed out to 50 m, whereas the peak abundance of *S. loveni* was further away from the stocked cages at 50-100 m. These results suggest *S. loveni* is less tolerant of the conditions associated with highly enriched sediments (i.e., as would be found directly adjacent to stocked cages). However, the presence of each or both species could also be related to feed inputs and farm history. At Strahan (MF267) where feed inputs were comparatively low and, in a location, typically experiencing higher bottom water oxygen concentrations, the peak abundance of *S. loveni* was observed closer to the cage (0 m) compared with the other leases. At Strahan, the transition to suspension and deposit feeding species responding to the increased food supply, but less tolerant of the highly enriched conditions, also occurred closer to the cage relative to the other leases. Species such as *Pista australis* and *Echinocardium cordatum* were relatively common at the Strahan 0 m cage site across the surveys, and *Pseudopolydora* cf. *paucibranchiata*, *Branchiomma* sp., *Samythella* sp and *Euchone varibilis* were common at the intermediate distances.

In contrast, at Franklin (MF266) where feed inputs were the highest of all leases and bottom water oxygen concentrations often the lowest, the peak abundance of *S. loveni* and two of the other opportunistic species *Capitella* sp. and *Charcotia australiensis* occurred further from the cage. In this case, the transition to the suspension and deposit feeders that included *Pista australis*, *Echinocardium cordatum, Pseudopolydora* cf. *paucibranchiata, Branchiomma* sp., and *Samythella* sp. also occurred further from the cage compared to at Strahan. At Table Head Central the feed inputs were the second highest of all sites studied, but in this case the lease is closest to the influence of the ocean, and consequently here the opportunistic and enrichment tolerant taxa were only found at the cage site, and the transition to the less tolerant suspension and deposit feeders occurred at intermediate distances. At Gordon where feed inputs were the lowest over the duration of the study, peak abundance of the more tolerant species was interestingly actually further from the cage at 50 -100 m and the transition to species such as *P. australis, E. cordatum, P. paucibranchiata, Branchiomma* sp., and *Samythella* sp. also appeared to occur further from the cage. This result likely reflects the Gordon lease's more than a decade longer operating history than the other study leases.

Changes in sediment chemistry were also variable in space (between leases) and time (between surveys), again highlighting the complex interplay of variability in the background environmental conditions and influence of feed inputs and farming history when interpreting benthic responses. The influence of the estuarine gradient can be broadly seen from the stable isotope signatures of the sediments. The δ^{13}C signature of terrestrial organic matter (25 to -33‰) and freshwater phytoplankton (-25 to -30‰) is typically more depleted compared with marine particulate organic matter (-22 to -18‰) (see Middelburg and Nieuwenhuize, 1998). Similarly, marine organic matter usually has a δ^{15}N signature of 5 - 7‰ indicating it is derived from phytoplankton, whereas terrestrial organic matter generally has δ^{15}N values < 4‰ (see Middelburg and Nieuwenhuize, 1998). The δ^{15}N and δ^{13}C values from leases in the southern end of the harbour were generally more depleted consistent with a greater contribution of terrestrial and freshwater inputs, whereas leases closer to marine influences, Table Head Central (133) and Gordon (219) had more enriched δ^{15}N and δ^{13}C values. This was consistent across farm and control sites. The distribution of all sites based on δ^{15}N and δ^{13}C ratio shows sites to the bottom left of the plot (Figure 5.4-88) were those located in the southern end of the harbour, in the World Heritage Area and closest to the Gordon River, whilst sites at the top right were closer to the harbour entrance. Interestingly, a similar estuarine gradient was also observed in the C and N content in the sediments (Figure 5.4-89). In this case, the control and lease sites closest to the harbour entrance had the lowest sediment C and N content overall, with levels increasing on lease further south reflecting the increasing terrestrial inputs from the Gordon River.

Nonetheless, the influence of farming could often be seen quite clearly over this estuarine gradient. The lease samples clearly showed an enrichment signal in both the δ^{15}N and δ^{13}C signature and the C and N content, relative to the wider harbour signature, resembling fish feed (Crawford et al. 2003; Chen et al. 2003; Wang et al. 2013). Sites closer to the cage also often had a lower C:N ratio typical of fish feed. This pattern was particularly evident at the oldest lease, 219 (Gordon), where there was a clear gradient in isotope signatures, C and N content and C:N ratio with distance from the cage, and the footprint of farm derived organic matter was evident out to 250-500 m from the cage. These results are consistent with findings of an earlier study (Ross et al. 2015) also undertaken at the Gordon and adjacent Central lease which shares a similar farming history. The C and N content, C:N ratio and isotopic signature of sediment organic matter at the cage sites at both leases were distinct and indicative of farm enrichment across four surveys undertaken between November 2012 and September 2013. In the study by Ross et al. (2015), the 50 m sites were also consistently indicative of an intermediate level of enrichment across both leases and all four surveys when compared to the more distant control sites and cage sites. The 0 m cage sites at the other leases in this study also had a more depleted δ^{13}C and enriched δ^{14}N isotope signature, a higher C and N content and lower C:N ratio relative to the other distances, indicating definite nutrient enrichment. At the intermediate distances, however, the pattern was less clear. The C and N content was often higher, but not always. Given farm inputs were lowest at Gordon over the course of the study relative to the other leases, the results clearly suggest farming history can markedly affect the ability to detect impacts using this approach; the results from the earlier study at the Gordon and Central leases support this. In this study, the top 2 cm of sediment was sampled and homogenised for sediment analysis, providing an integrated measure of factors that influence this depth horizon. At the older lease (with a 15-year history of farming), it seems reasonable to assume a longer history of farming has produced a deeper layer of farm-affected sediment than at the newer leases, as such the full 2 cm at such sites may be influenced by farming. In contrast, at the other leases, operating for just over a year at the first survey (or ~ 3.5 years at survey 6), the depth of influence may not have extended to 2 cm, particularly at 50 m and further from the cage and source of enrichment.

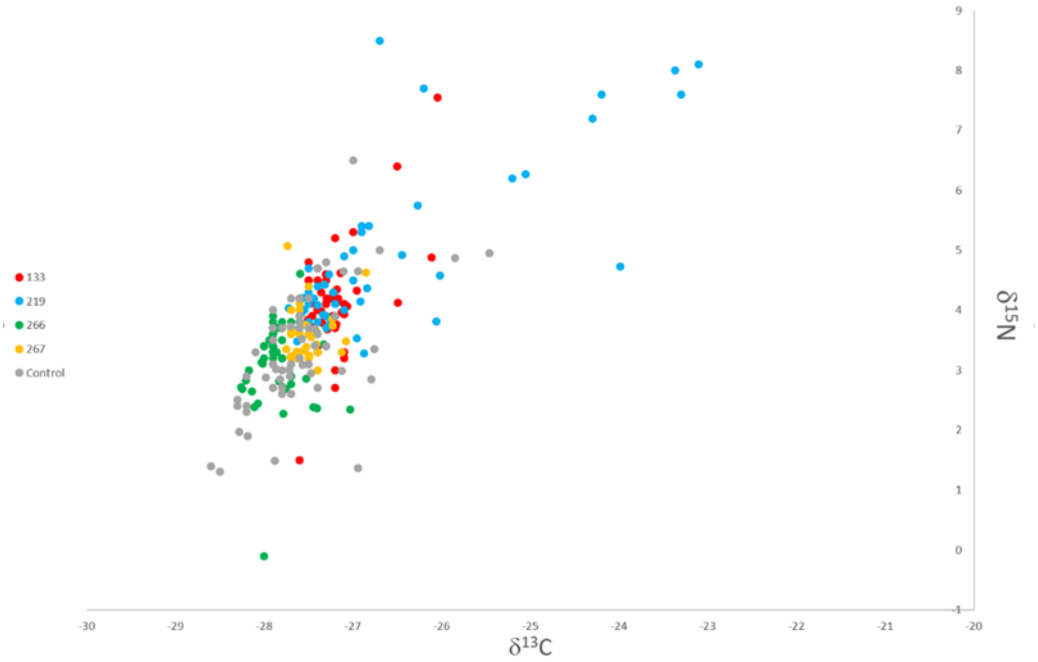


Figure 5.4-88 Distribution of sites based on carbon and nitrogen stable isotopes.

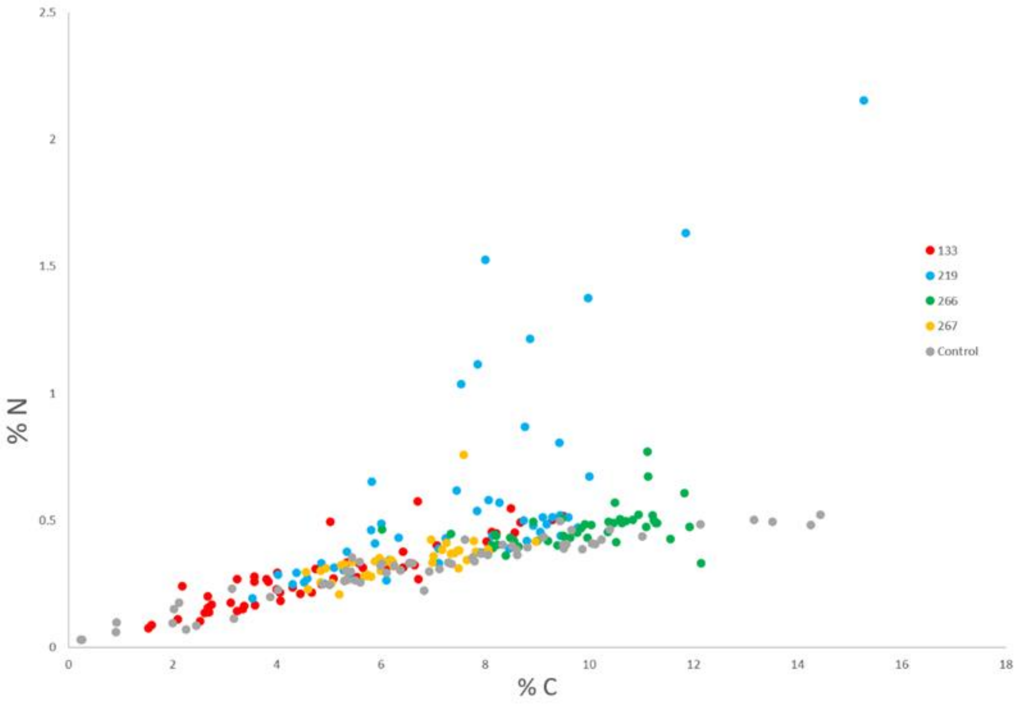


Figure 5.4-89 Distribution of sites based on carbon and nitrogen content.

For the other key physico-chemical parameters measured such as redox potential, the patterns were often difficult to interpret. This likely reflects, at least in part, the effect of low and variable bottom water oxygen conditions on the redox profile of the sediments. However, there still appeared to be a clear estuarine gradient within the harbour, with lower redox values with increasing distance from the harbour entrance, becoming negative at the control sites to the south. At Table Head Central the effect of farming on redox was evident with lower values in closer proximity to the cages. At Gordon this was notable in surveys 1 and 4, but in surveys 5 and 6 the opposite pattern was evident with more positive values in closer proximity to the cages. This is difficult to explain, given bottom water oxygen concentrations measured during these two surveys were lower, particularly in survey 6 when oxygen levels in bottom waters were less than 10% saturation. At the Strahan lease, there was no clear pattern with distance in surveys 1 and 4, but in surveys 5 and 6 redox values decreased with distance from the cage, as seen at Gordon. At Franklin, redox was notably lower at the cage site in 3 of the 4 surveys, but otherwise there was no clear pattern with distance.

The process measurements of organic matter breakdown conducted during surveys 4 and 5 provided an interesting contrast against the more readily measured indicators of impact described above (e.g., fauna and physico-chemical parameters). As expected, rates of organic matter mineralisation were significantly higher in proximity to the cages and source of enrichment compared to the more distant transect sites and control sites (Figure 5.4-90). Rates of respiration measured via oxygen consumption averaged ~1900 µmol O_2 m^{-2} h^{-1} at the cage sites and decreased with distance to an average of ~350 - 500 µmol O_2 m^{-2} h^{-1}; consumption rates at 50 and 100 m averaged ~900 and ~680 O_2 m^{-2} h^{-1} respectively. The rates at the cage sites were comparable not only to those recorded previously in Macquarie Harbour (Ross et al. 2015) but also to those reported from elsewhere under salmon cage aquaculture (e.g., Hargrave et al. 1993; Pereira et al. 2004). Respiration measured as production of dissolved inorganic carbon (principally carbon dioxide) was also much higher at the cage sites (~5500 µmol DIC m^{-2} h^{-1}), but it was noted that beyond the cage site there was no clear pattern with distance, with average production rates from 50-500m ranging between 450 and 750 DIC µmol m^{-2} h^{-1}.

In coastal sediments, particularly in low oxygen environments, anaerobic respiration of carbon is often dominant. Alternative oxidants (i.e., nitrate, manganese and iron hydroxides, sulphate) are used when the demand of oxidants exceeds the supply of oxygen (Middleburg et al. 2004). In Macquarie Harbour sediments, the rate of dissolved inorganic carbon production (DIC) often exceeded the rate of oxygen consumption, particularly in enriched farm sediments, indicating anaerobic respiration was common. Ammonium fluxes from the sediment largely reflected the patterns of respiration. Average rates of production of ammonium were significantly elevated at the cage sites (~500 µmol NH_4^+ m^{-2} h^{-1}) and, though production rates were markedly lower, ammonium production was elevated at 50 and 100 m compared to the more distant sites, including the controls. Nitrate + nitrite fluxes were quite variable and predominately directed into the sediment at all sites. This is consistent with a low oxygen environment where the process of nitrification (conversion of ammonia to nitrate in oxic conditions) in sediments is limited and the denitrification process (conversion of nitrate to nitrogen gas in anoxic conditions) must rely on sourcing nitrate from the water column rather than from nitrification in the sediments (see Cornwell et al. 1999). This pattern is what might be expected from enriched sediments, based on oxygen consumption across the sites, with the largest uptake at the cage sites where oxygen consumption is highest, and thus, availability is lowest. The higher fluxes of phosphate out of sediments at the most enriched sites is consistent with reduced sediment oxygen; phosphate is typically bound in oxidised sediments but may be released from reduced anoxic sediments.

Importantly, the sediment process measurements also appeared to reflect the stocking and fallowing regimes. At Table Head Central, stocking was similar and consistently high in both surveys for the three months prior to the survey. This was reflected in the sediment fluxes, with rates of DIC and ammonium production elevated, showing the expected response with distance. At Gordon, stocking of the cages in closest proximity to the sample sites in the three months preceding survey 4 was much higher than for survey 5. Rates of respiration measured both via oxygen consumption and DIC production were markedly elevated at the cage sites in survey 4, but not survey 5. Fluxes of ammonium and nitrate + nitrite were also higher in close proximity to the cages in survey 4 compared to survey 5. At Strahan, a similar pattern was evident, with rates of respiration, ammonium production and nitrate uptake markedly higher in survey 4 than survey 5 in proximity to the cages; this was consistent again with higher stocking levels preceding survey 4 than survey 5. At Franklin, stocking in the three months prior to each survey was higher than seen at any of the other leases. Whilst respiration, as measured via oxygen consumption, was relatively low at the cage site in both surveys, DIC production was very high at the cage site in survey 4 (no DIC data was available for survey 5), and ammonium production at the cage sites in both surveys was double that recorded at any of the other lease cage sites. This suggests anaerobic respiration was particularly dominant, consistent with higher feed inputs and lower bottom water oxygen concentrations in this part of the harbour.

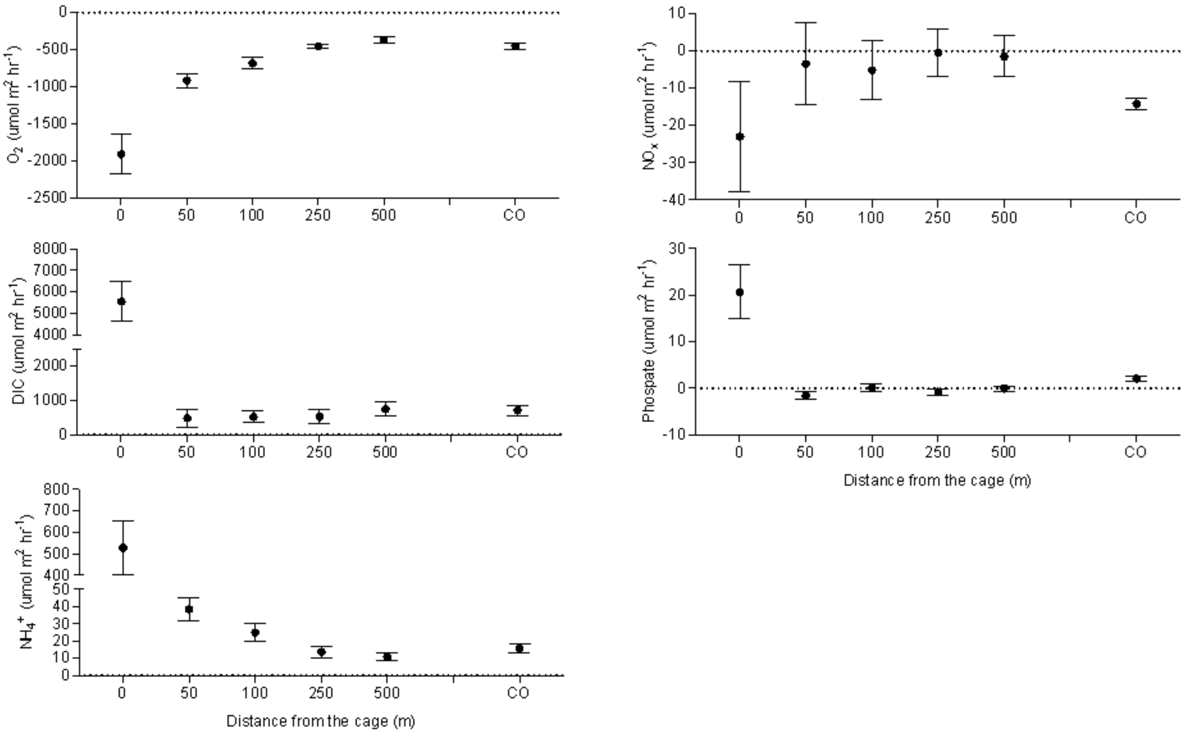


Figure 5.4-90 Summary of sediment fluxes in Macquarie Harbour. Fluxes of dissolved oxygen, dissolved inorganic carbon, ammonium, nitrate + nitrite and phosphate with distance from the cages and control sites averaged (± SE) across all leases, control sites and surveys.

One of the aims of the local scale work was to better understand the recovery responses of the sediments to farm management such as fallowing. This was often quite difficult given uncertainty over the timing of fallowing periods and the logistics of coordinating benthic surveys with changes in farm practices, particularly in remote locations such as Macquarie Harbour. Consequently, differences in farming intensity (feed inputs) and an objective understanding of changes in management between leases to identify the role of farm management was relied on. Fallowing in Macquarie Harbour is often, but not always, designed around cage-based movements rather than relocation of whole grids or leases, and there were few occasions when the benthic surveys corresponded directly with significant fallowing of the whole lease. The Gordon and Strahan leases were exceptions; Gordon was fallow or very lightly stocked between January 2016 to September 2016 and Strahan was fallow from January 2016 to May 2016. As a result, survey 4 (February 2016) and survey 5 (June 2016) provided better insight into the effect and benthic response of four-to-five months fallowing on these two leases. Although there was some evidence of a change in the C and N signatures of the sediment at the cage site because of lower feed inputs (e.g., ↓δN%N↑ C:N ratio) at Gordon, and to a lesser extent Strahan over this period, there was little evidence of a response in the benthic ecology. In contrast the sediment process measurements showed a clear decrease in sediment respiration and ammonium production at these two leases in survey 5 compared to survey 4. These results suggest sediment function (e.g., organic matter processing) in Macquarie Harbour may be quite responsive to changes in farm loads, but changes in sediment attributes (e.g., faunal communities) associated with this response may occur over a longer time scale. This is consistent with the findings from an assessment of recovery at the Gunpowder lease in SE Tasmania, which highlighted chemical measures improved relatively rapidly compared to the macrobenthos; with chemical measures having markedly improved after two months whilst a moderate impact in benthic communities could still be distinguished at the cage stations after 24 months of fallowing

(Macleod et al. 2004; 2008). However, Macleod et al. 2008 noted though community structure took longer to return to conditions equivalent to that of reference conditions, there had a been a marked change in the community composition by 12 months whereby ecological function had been restored.

Estimates of benthic faunal recovery rates vary from months to years, and Macleod et al. (2004) note the variability in these estimates is probably due to a combination of both ecosystem and farm management differences. Site characteristics such as water depth, particle size, current velocities and tidal effects are all known to be important in determining the rate and extent of both the degradation and recovery of sediments. In Macquarie Harbour, the inherent spatial and temporal variation in background environmental conditions, and most notably bottom water dissolved oxygen concentrations, play a major role in determining the ultimate response of a given area to enrichment and recovery. This was most evident in the final survey (survey 6) of this study conducted in Spring 2016 when dissolved oxygen levels within the Harbour declined to the lowest levels on record (see Ross & Macleod 2017). Whilst there was a significant decline in the total abundance and number of species collected from the benthic fauna at both the lease and external sites at this time, the magnitude of the change varied markedly across leases and external sites, with the effect again increasing with distance from the Harbour entrance. The major deterioration in water column and benthic conditions observed in the Spring 2016 survey saw an extension of this research through FRDC 2016-067. The results of these subsequent surveys have been discussed extensively in the eight update reports published during 2017-2020[2]. A key finding of that research (nine additional surveys) was fauna clearly having the capacity to recover, even after the declines observed during low dissolved oxygen events in Spring 2016 and 2017. In the winter surveys conducted the following year, signs of faunal recovery in both abundance and the number of species were clear, but the magnitude of recovery varied between leases and external sites. Recovery was slower at leases and external sites located further into the harbour, away from the entrance to the ocean, and at leases with longer or more intense farm history. Subsequent work has indicated faunal communities in the extensive shallow regions around the margins of the harbour were far less affected by the low DO events than the deeper waters. These shallower communities likely provide an important reservoir for species recruitment, and as such may play a critical role in recovery of benthic communities in the deeper regions. This again highlights the complex interplay, for both the ecology and recovery processes, between the direct effects of farm enrichment on food availability and sediment condition and the influence of bottom water dissolved oxygen concentrations.

Implications for monitoring and management

It is widely recognised benthic infauna evaluation is one of the most robust and sensitive ways to measure sediment condition and response to organic enrichment. This has been well documented for salmonid aquaculture in the growing regions in southern Tasmania. Although the benthic community in Macquarie Harbour is naturally depauperate and quite distinct, the results of this study provided further support that the basic pattern of response is broadly consistent with that expected in response to organic enrichment (see Macleod & Forbes 2004, Ross et al. 2015) but at an increased spatial scale (i.e. at greater distances from the source) than observed in southern regions, such that opportunistic and more enrichment tolerant species often dominate out to 50 -100 from the cage. Beyond this, suspension and

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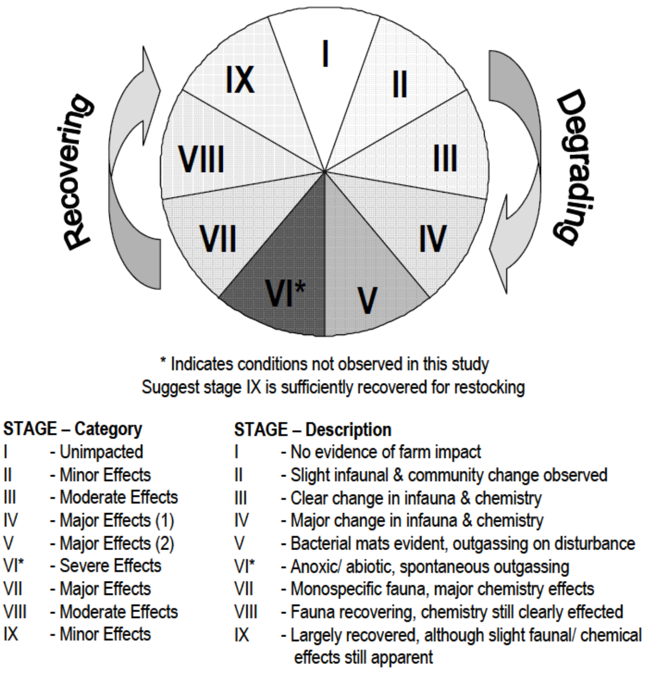
[2] All the reports can be found at https://www.imas.utas.edu.au/research/fisheries-and-aquaculture/publications-and-resources

surface deposit feeders proliferate in response to the increased food supply. Macleod and Forbes (2004) highlighted regional differences in the key biotic and abiotic indicators of impact/recovery stages based on substrate type, which reflected the differences between more exposed sandy and sheltered mud sites in southern Tasmania. This provides an important comparator and context to support the development of Macquarie Harbour specific guidelines that specifically characterise the different stages of impact and recovery in the harbour and provide for site specific environmental standards for the assessment of unacceptable biological impacts in licence conditions. However, a challenge and confounding factor in this case is the inherent environmental variability observed between leases (i.e. the role of distance from the harbour entrance and depth of the lease) and over time (i.e. the influence of system wide declines in bottom water dissolved oxygen concentrations).

In Macquarie Harbour, sediment conditions at the cage site often appeared to be approaching those equivalent to impact stage VI (Severe Effects), as described in previous characterisation of southern systems (Macleod & Forbes 2004; Figure 5.4-91) i.e. when sediments are devoid of fauna and bottom waters are anoxic. However, in Macquarie Harbour the naturally depauperate background conditions and low bottom water levels likely predispose the system and capacity for this to occur; the system wide decline in oxygen levels to extremely low in survey 6 saw several sites devoid of fauna both close to and distant from the cages. There were also several instances when faunal abundance and the presence of opportunists were extremely low at the cage site, suggesting enrichment levels may have exceeded the assimilative capacity of the sediments. Given these inherent conditions of Macquarie Harbour, the enrichment threshold for this stage may be lower than observed elsewhere. Keeley et al. (2019) highlighted variability in farm production levels and depositional fluxes will affect the threshold at which sediments become severely impoverished of benthic fauna and noted sites with low flow typically demonstrate a lower threshold. Other studies have also shown the importance of an essentially unlimited supply of oxygen in bottom waters in buffering against anoxia, and how this is often a particular feature at sites with high



* Indicates conditions not observed in this study
Suggest stage IX is sufficiently recovered for restocking

STAGE – Category		STAGE – Description	
I	- Unimpacted	I	- No evidence of farm impact
II	- Minor Effects	II	- Slight infaunal & community change observed
III	- Moderate Effects	III	- Clear change in infauna & chemistry
IV	- Major Effects (1)	IV	- Major change in infauna & chemistry
V	- Major Effects (2)	V	- Bacterial mats evident, outgassing on disturbance
VI*	- Severe Effects	VI*	- Anoxic/ abiotic, spontaneous outgassing
VII	- Major Effects	VII	- Monospecific fauna, major chemistry effects
VIII	- Moderate Effects	VIII	- Fauna recovering, chemistry still clearly effected
IX	- Minor Effects	IX	- Largely recovered, although slight faunal/ chemical effects still apparent

Figure 5.4-91 Impact and recovery stages (from Macleod et al. 2004)

assimilative capacity (e.g., ability to process waste and recover quickly; (Findlay & Watling 1997; Keeley et al 2013). Clearly, this is an important consideration for Macquarie Harbour.

The suitability of physico-chemical parameters as indicators of enrichment was more ambiguous in Macquarie Harbour. At the Gordon lease, the C and N signatures (content, ratio and isotopes) showed a clear enrichment gradient with distance, but this pattern was far less clear at the other three study leases. This may in part be due to the much longer farm history at Gordon compared to the other relatively new leases, and as such farm history should be considered when evaluating and comparing physico-chemical data. It is also suggested sampling of a shallower depth horizon be considered if sediment surveys are required in the future. Background variation in space (e.g., estuarine gradient) and time (e.g., fluctuations in oxygen levels) also make interpretation of physico-chemical measures challenging. This was particularly problematic for the biogeochemical measures[3] which can be influenced by variation in bottom water oxygen conditions. In a study of six finfish farming locations in Marlborough Sound, New Zealand, Keeley et al. (2013) reported the percentage of organic matter and several other biogeochemical variables, such as redox and sulphide, were poor predictors of biological conditions at the high flow sites. They reiterated the importance of validating the relationship between physico-chemical parameters and biological and environmental conditions. In Macquarie Harbour the flow rates would be considered intermediate, lying somewhere between Keeley et al. (2013) low flow and high flow sites; arguably the low and variable bottom water flows in Macquarie Harbour were more influential parameters in determining both oxygen levels and the resultant sediment condition Similarly, in their assessment of broadscale effects of salmonid aquaculture in south eastern Tasmania (based on environmental monitoring from 1997-1999 across 20 leases), Edgar et al. (2005) also reported biotic metrics were more sensitive to effects of farming than abiotic metrics.

An important aim of the Macquarie Harbour work was to establish the reliability of the visual indicators developed in the south and used in regulatory monitoring. The results shown here clearly suggest visual assessment of sediment condition can be very reliable, particularly when calibrated with site-specific criteria and weighting. The quantitative video scoring of sediment proposed by Crawford et al. (2001) and further developed by Macleod et al. (2004) in southern Tasmania was able to clearly discern the broad range of possible impact levels along a gradient of distance from the cage. It was even possible to discern, as described in the biological assessments, impacts that extended further from the cage at the Gordon and Franklin leases compared with Table Head and Strahan. In survey 6, sediment condition deteriorated at all distances at Strahan and Franklin, the two most southern leases where the decline in bottom water dissolved oxygen levels was more pronounced; this is illustrated well in the heat maps shown in .

Understanding the spatial scale of the benthic response to enrichment in Macquarie Harbour is not only important for the design of the benthic monitoring program, but also for cage positioning, particularly regarding overlapping footprints, distance from the lease boundary and the 35 m compliance points and ongoing planning. Cages close to the lease boundary are more likely to lead to benthic effects outside the lease. In this study the cages were between 100-250 m from the lease boundary, ostensibly to retain any impacts within the lease. In terms of establishing reference conditions, the results of this study indicate the 500 m sites

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[3] Although sulphide measurements were not included this study, subsequent measurements as part of project 2016-067 have shown that sulphide, like redox, is highly variable and difficult to interpret in response to enrichment.

are still likely to be influenced by low levels of enrichment based on the presence and abundance of suspension and deposit feeders. White et al. (2017) evaluated the fatty acid composition of sediments in key areas and reported traces of farm waste using fatty acids out to 500 m in Macquarie Harbour. This has significant implications for monitoring and would suggest reference locations should be at least 1 km away from farms but note this may be difficult considering the limited availability of locations with a similar depth, substrate and proximity to other farms.

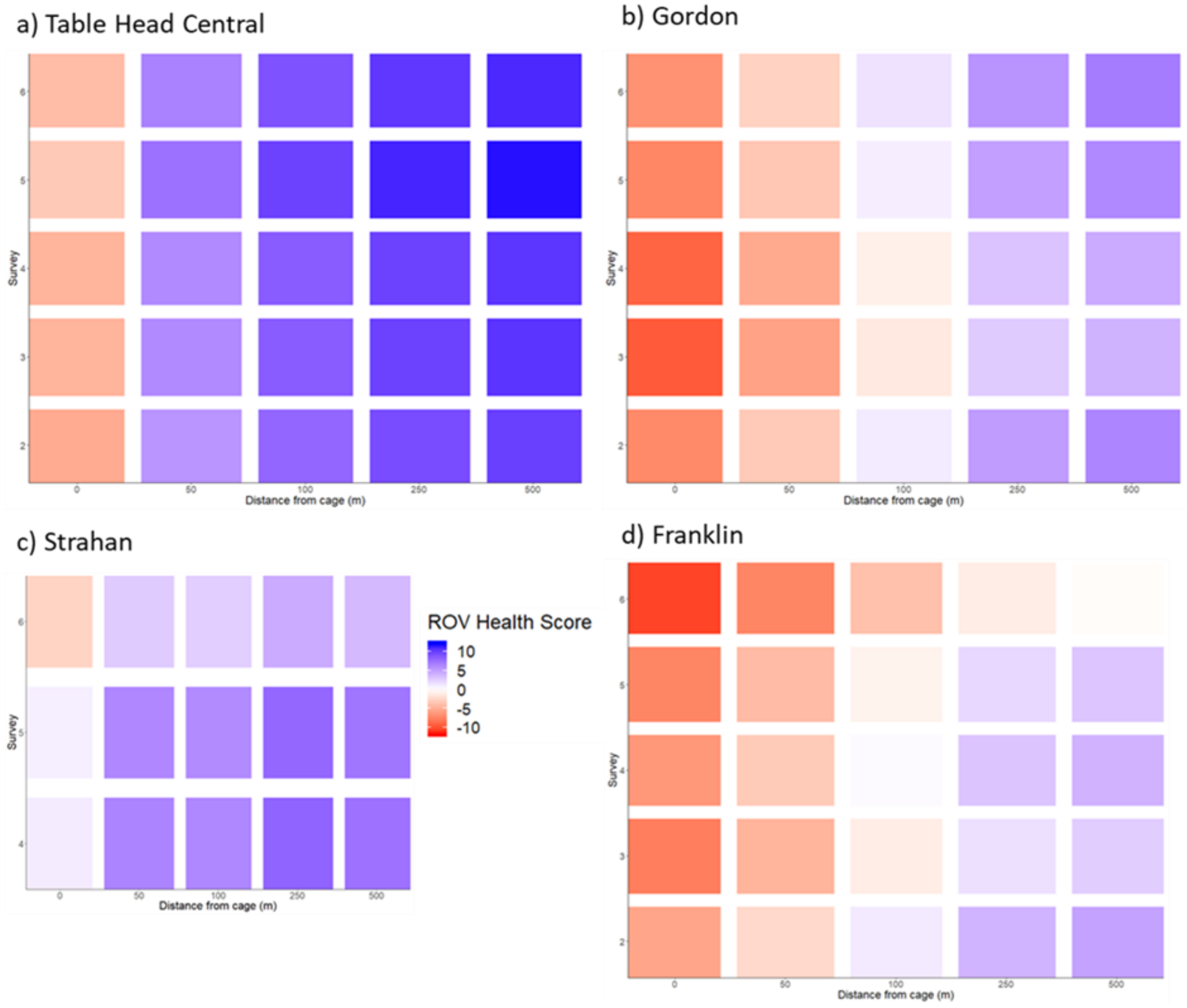

Figure 5.4-92 Heat maps showing ROV sediment health scores with distance from the cage for the surveys scored at each of the study leases.

Table 5.4-1 Results of the generalised linear model testing for the effect of distance for all physico-chemical parameters for each survey and the planned contrasts from the factorial ANOVA comparing the control sites and transect distances. Significant responses ($p < 0.05$) are shown in bold.

			Planned contrasts	
	F ratio	**P**	**CO v 50**	**CO v 100**
Redox				
Table Head Central				
Survey 1	0.155	0.704		
Survey 4	559	**<0.001**		
Survey 5	1.106	0.3702		
Survey 6	2.490	0.213		
Gordon				
Survey 1	18.707	**0.002**		
Survey 4	15.86	**0.028**		
Survey 5	5.812	0.095		
Survey 6	15.431	**0.029**		
Strahan				
Survey 1	0.012	0.891		
Survey 4	0.031	0.871		
Survey 5	2.707	0.198		
Survey 6	6.532	0.084		
Franklin				
Survey 1	0.121	0.737		
Survey 4	13.731	**0.034**		
Survey 5	9.143	0.057		
Survey 6	8.198	0.064		
% Carbon				
Table Head Central				
Survey 1	0.199	0.668	0.806	0.832
Survey 2	0.009	0.928	0.733	0.944
Survey 4	0.158	0.701	0.872	0.861
Survey 5	0.006	0.941	0.792	0.755
Survey 6	0.034	0.859	0.283	0.365
Gordon				
Survey 1	5.231	0.052	0.065	0.450
Survey 2	1.875	0.208	0.660	0.867
Survey 4	0.211	0.658	0.083	**0.019**
Survey 5	0.004	0.950	**<0.001**	**<0.001**
Survey 6	0.248	0.632	**<0.001**	**0.002**
Strahan				
Survey 1	1.305	0.286	0.921	0.962
Survey 2	NA	NA	NA	NA
Survey 4	1.832	0.213	0.482	0.496
Survey 5	0.624	0.452	0.197	0.225
Survey 6	0.660	0.440	1.000	0.941
Franklin				
Survey 1	0.3342	0.579	0.110	0.159
Survey 2	0.2929	0.603	0.287	**0.004**
Survey 4	2.514	0.151	0.418	0.999
Survey 5	0.510	0.495	**0.006**	**0.002**
Survey 6	1.410	0.270	**0.014**	**0.028**
% Nitrogen				
Table Head Central				
Survey 1	1.406	0.270	0.737	0.804
Survey 2	0.220	0.652	0.901	0.989
Survey 4	0.317	0.589	0.838	0.689
Survey 5	0.009	0.927	0.697	0.664
Survey 6	0.977	0.352	0.513	0.474
Gordon				
Survey 1	41.42	**<0.001**	0.185	0.733
Survey 2	13.239	**0.007**	0.987	0.992

Survey 4	20.003	**0.002**	0.350	0.486
Survey 5	6.814	**0.031**	0.372	0.538
Survey 6	9.792	**0.014**	0.105	0.409
Strahan				
Survey 1	2.648	0.142	0.957	0.875
Survey 2	NA	NA	NA	NA
Survey 4	18.501	**0.003**	0.647	0.410
Survey 5	1.036	0.339	0.986	0.995
Survey 6	0.111	0.747	0.831	0.728
Franklin				
Survey 1	2.343	0.164	**0.014**	**0.003**
Survey 2	0.004	0.954	0.354	0.078
Survey 4	11.292	**0.010**	0.673	0.996
Survey 5	6.222	**0.037**	0.400	0.287
Survey 6	5.134	0.053	0.071	0.198
C:N				
Table Head Central				
Survey 1	0.484	0.507	0.998	0.945
Survey 2	28.64	**<0.001**	0.058	0.602
Survey 4	3.19	0.112	0.492	0.052
Survey 5	3.94	0.082	**<0.001**	**<0.001**
Survey 6	14.36	**0.005**	0.995	0.964
Gordon				
Survey 1	129.47	**<0.001**	0.054	0.711
Survey 2	42.432	**<0.001**	0.178	0.433
Survey 4	61.16	**<0.001**	**<0.001**	0.069
Survey 5	10.144	**0.013**	**0.04**	0.119
Survey 6	18.976	**0.002**	0.470	0.879
Strahan				
Survey 1	<0.001	0.980	0.370	0.259
Survey 4	7.946	**0.022**	0.540	0.843
Survey 5	0.348	0.572	0.808	0.758
Survey 6	1.48	0.259	0.827	0.911
Franklin				
Survey 1	4.157	0.076	0.728	0.946
Survey 2	1.162	0.313	0.723	0.986
Survey 4	15.766	**0.004**	0.891	0.951
Survey 5	5.648	**0.045**	0.795	0.714
Survey 6	3.452	0.100	0.832	0.992
δ^{13}C				
Table Head Central				
Survey 1	14.109	**0.006**	0.919	0.919
Survey 2	2.846	0.130	0.979	0.998
Survey 4	12.66	**0.007**	**0.020**	0.066
Survey 5	12.0461	**0.008**	0.708	0.406
Survey 6	37.511	**<0.001**	0.640	0.950
Gordon				
Survey 1	115	**<0.001**	0.315	0.885
Survey 2	49.59	**<0.001**	0.813	0.986
Survey 4	72.093	**<0.001**	0.350	0.910
Survey 5	3.259	0.109	0.792	0.962
Survey 6	23.203	**0.001**	0.844	0.925
Strahan				
Survey 1	1.246	0.297	0.646	0.952
Survey 4	8.186	**0.021**	0.764	0.769
Survey 5	0.103	0.756	0.658	0.371
Survey 6	9.37	**0.016**	0.773	0.875
Franklin				
Survey 1	0.854	0.382	**0.003**	**0.012**
Survey 2	1.224	0.301	**<0.001**	**0.015**
Survey 4	3.36	0.104	0.985	0.879

Survey 5	0.004	0.954	0.730	0.871
Survey 6	7.561	**0.025**	0.392	0.336
δ^{15}N				
Table Head Central				
Survey 1	1.062	0.333	0.982	0.974
Survey 2	22.295	**0.001**	0.744	0.954
Survey 4	1.427	0.267	0.995	0.983
Survey 5	3.154	0.114	0.997	0.734
Survey 6	10.615	**0.012**	0.980	0.910
Gordon				
Survey 1	14.575	**0.005**	0.053	0.975
Survey 2	48.032	**<0.001**	0.506	0.976
Survey 4	65.768	**<0.001**	0.335	0.928
Survey 5	12.763	**0.007**	0.856	0.986
Survey 6	12.15	**0.008**	0.163	0.826
Strahan				
Survey 1	5.01	0.056	0.989	0.936
Survey 4	11.496	**0.009**	1.000	0.993
Survey 5	10.43	**0.012**	0.506	0.861
Survey 6	4.03	0.079	0.856	0.957
Franklin				
Survey 1	1.686	0.230	0.707	0.686
Survey 2	1.655	0.234	0.090	0.057
Survey 4	9.242	**0.016**	0.925	0.923
Survey 5	4.267	0.072	0.994	0.994
Survey 6	9.735	**0.014**	0.911	0.998
Dissolved Oxygen				
Table Head Central				
Survey 1	0.369	0.560	0.771	0.728
Survey 2	0.045	0.837	0.825	0.173
Survey 3	5.608	**0.045**	**0.011**	**<0.001**
Survey 4	0.756	0.410	0.697	0.649
Survey 5	1.553	0.281	0.641	0.587
Survey 6	0.096	0.766	**<0.001**	**<0.001**
Gordon				
Survey 1	4.597	0.064	**<0.001**	0.074
Survey 2	0.035	0.857	**<0.001**	**<0.001**
Survey 3	3.29	0.107	**0.007**	**0.028**
Survey 4	1.105	0.324	**<0.001**	**<0.001**
Survey 5	1.603	0.295	na	na
Survey 6	0.576	0.470	**<0.001**	**<0.001**
Strahan				
Survey 1	3.887	0.084	**<0.001**	**<0.001**
Survey 3	2.873	**0.129**	0.924	0.790
Survey 4	4.357	**0.070**	**<0.001**	**<0.001**
Survey 5	4.347	**0.128**	na	na
Survey 6	0	0.998	0.502	0.462
Franklin				
Survey 1	6.973	**0.030**	0.973	0.640
Survey 2	8.035	**0.022**	**0.001**	**0.004**
Survey 3	2.176	0.178	0.078	0.219
Survey 4	0.070	0.798	0.910	0.929
Survey 5	3.068	0.118	0.513	0.589
Survey 6	0.179	0.684	**0.021**	**0.023**
Total abundance				
Table Head Central				
Survey 1	0.456	0.532	0.658	0.521
Survey 2	0.724	0.42	0.331	0.838
Survey 3	0.761	0.408	**<0.001**	0.256
Survey 4	0.01	0.925	**<0.001**	0.475
Survey 5	9.365	**0.016**	0.358	0.910

Survey 6	1.057	0.334	0.135	0.652
Gordon				
Survey 1	0.707	0.425	0.573	0.213
Survey 2	1.900	0.205	0.886	0.258
Survey 3	4.96	0.057	0.588	0.459
Survey 4	8.465	**0.020**	0.637	0.752
Survey 5	1.534	0.251	0.994	0.866
Survey 6	4.179	0.075	**0.012**	0.193
Strahan				
Survey 1	2.776	0.134	0.463	0.573
Survey 3	14.951	**0.005**	0.885	0.910
Survey 4	5.299	**0.050**	0.939	0.971
Survey 5	3.366	0.104	0.151	0.310
Survey 6	0.878	0.376	0.159	0.435
Franklin				
Survey 1	0.289	0.606	0.602	0.233
Survey 2	0.402	0.545	0.957	0.428
Survey 3	0.027	0.873	0.472	0.958
Survey 4	0.779	0.403	0.731	0.317
Survey 5	2.652	0.142	0.700	0.958
Survey 6	0.628	0.451	0.233	0.488
Species richness				
Table Head Central				
Survey 1	6.721	**0.032**	0.928	0.997
Survey 2	10.742	**0.011**	0.999	0.995
Survey 3	34.411	**<0.001**	0.986	0.999
Survey 4	15.466	**0.004**	0.906	0.843
Survey 5	5.394	**0.049**	0.718	0.547
Survey 6	28.975	**<0.001**	0.601	0.674
Gordon				
Survey 1	3.215	0.111	0.550	0.952
Survey 2	11.781	**0.009**	**0.031**	0.453
Survey 3	7.966	**0.022**	**<0.001**	0.183
Survey 4	9.841	**0.014**	0.122	0.149
Survey 5	21.192	**0.002**	**<0.001**	**0.002**
Survey 6	6.187	**0.038**	**<0.001**	**<0.001**
Strahan				
Survey 1	2.265	0.171	**0.002**	**0.003**
Survey 3	6.146	**0.038**	0.092	0.199
Survey 4	0.604	0.459	**0.044**	**0.004**
Survey 5	6.108	**0.039**	0.330	0.199
Survey 6	0.086	0.777	**0.009**	0.074
Franklin				
Survey 1	9.70	**0.014**	**<0.001**	**<0.001**
Survey 2	5.77	**0.043**	0.922	0.971
Survey 3	12.158	**0.008**	0.703	0.988
Survey 4	4.521	0.066	0.501	**0.012**
Survey 5	33.37	**<0.001**	0.916	0.614
Survey 6	0.628	0.451	0.233	0.488
Species diversity				
Table Head Central				
Survey 1	6.672	**0.035**	0.723	0.851
Survey 2	21.965	**0.002**	0.672	0.970
Survey 3	37.177	**<0.001**	0.521	0.636
Survey 4	25.248	**0.001**	0.302	0.926
Survey 5	9.365	**0.016**	0.055	**0.022**
Survey 6	41.425	**<0.001**	0.093	0.64
Gordon				
Survey 1	3.835	0.086	0.273	0.788
Survey 2	12.138	**0008**	**0.029**	0.537
Survey 3	4.468	0.067	**<0.001**	0.267

Survey 4	11.198	**0.010**	0.126	0.289
Survey 5	5.043	0.055	**0.028**	0.057
Survey 6	4.049	0.079	**<0.001**	**<0.001**
Strahan				
Survey 1	9.056	**0.017**	**<0.001**	**<0.001**
Survey 3	1.157	0.314	**0.003**	0.128
Survey 4	7.505	**0.025**	0.054	**0.007**
Survey 5	4.347	0.128	0.728	0.470
Survey 6	0.244	0.635	**0.004**	**0.037**
Franklin				
Survey 1	13.517	**0.006**	0.099	**<0.001**
Survey 2	4.526	0.066	0.341	0.580
Survey 3	8.549	**0.019**	0.520	0.927
Survey 4	3.171	0.113	0.691	0.343
Survey 5	22.626	**0.001**	0.665	0.723
Survey 6	-	-		
ROV Health Score				
Table Head Central				
Survey 2	21.749	**0.002**	0.837	0.974
Survey 3	89.026	**<0.001**	0.220	0.684
Survey 4	9.460	**0.018**	0.731	0.995
Survey 5	27.362	**<0.001**	0.850	0.980
Survey 6	74.663	**<0.001**	0.564	0.922
Gordon				
Survey 2	30.656	**<0.001**	**0.010**	0.498
Survey 3	38.764	**0.003**	**<0.001**	**0.004**
Survey 4	10.206	**0.013**	**<0.001**	**<0.001**
Survey 5	15.899	**0.004**	**0.002**	**0.039**
Survey 6	10.868	**0.011**	**0.004**	**0.037**
Strahan				
Survey 4	3.596	0.945	0.995	1.000
Survey 5	7.918	**0.023**	0.842	0.616
Survey 6	9.409	**0.018**	0.136	0.136
Franklin				
Survey 2	14.758	**0.005**	0.107	0.821
Survey 3	17.448	**0.003**	**0.049**	0.316
Survey 4	7.019	**0.030**	**<0.001**	**0.012**
Survey 5	26.712	**<0.001**	0.074	0.625
Survey 6	5.181	0.052	**<0.001**	**<0.001**

Table 5.4-2 Results of the generalised linear model testing for the effect of survey at the two distances closest to the cages (0 and 35/50 m) and at the three distances furthest from the cages (500, 1000 and CO). Significant responses ($p < 0.05$) are shown in bold.

	0 m	**50 m**	**500 m**	**CO**
Table Head Central				
Redox	NA	NA	NA	NA
Organic Carbon (%)	1.000	1.000	1.000	1.000
Organic Nitrogen (%)	1.000	1.000	1.000	1.000
$\delta^{13}C$	**<0.001**	**<0.001**	**<0.001**	**<0.001**
$\delta^{15}N$	0.154	1.000	1.000	1.000
C:N	**0.034**	0.407	**0.035**	**0.011**
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	0.172
Total Abundance	1.000	0.192	1.000	1.000
Species Richness	1.000	1.000	1.000	1.000
Species Diversity	0.60	0.721	1.000	1.000
ROV Score	1.000	1.000	1.000	1.000
Gordon				
Redox	NA	NA	NA	NA

Organic Carbon (%)	**0.035**	**0.008**	0.229	0.374
Organic Nitrogen (%)	0.862	1.000	1.000	1.000
δ^{13}C	**<0.001**	**<0.001**	**<0.001**	**<0.001**
δ^{15}N	**0.002**	**.0152**	1.000	1.000
C:N	1.000	1.000	1.000	1.000
Dissolved Oxygen (mg/L)	**0.046**	**0.046**	**0.012**	0.073
Total Abundance	1.000	0.264	1.000	1.000
Species Richness	1.000	1.000	1.000	1.000
Species Diversity	1.000	0.537	1.000	1.000
ROV Score	0.208	1.000	1.000	1.000
Strahan				
Redox	NA	NA	NA	NA
Organic Carbon (%)	0.687	0.487	**0.026**	0.687
Organic Nitrogen (%)	1.000	1.000	1.000	1.000
δ^{13}C	**<0.001**	**<0.001**	**<0.001**	**<0.001**
δ^{15}N	0.114	1.000	1.000	1.000
C:N	1.000	1.000	1.000	1.000
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Total Abundance	0.287	1.000	1.000	1.000
Species Richness	**0.001**	0.189	**0.002**	0.829
Species Diversity	**0.006**	0.979	**0.003**	0.979
ROV Score	1.000	1.000	1.000	1.000
Franklin				
Redox	NA	NA	NA	NA
Organic Carbon (%)	**0.025**	**0.002**	0.136	0.147
Organic Nitrogen (%)	**0.005**	1.000	1.000	1.000
δ^{13}C	**<0.001**	**<0.001**	**<0.001**	**<0.001**
δ^{15}N	0.327	1.000	1.000	1.000
C:N	**0.004**	0.513	1.000	1.000
Dissolved Oxygen (mg/L)	**<0.001**	**<0.001**	**<0.001**	0.475
Total Abundance	1.000	0.897	1.000	1.000
Species Richness	**0.011**	**<0.001**	**<0.001**	0.207
Species Diversity	**0.030**	**0.030**	**<0.001**	0.066
ROV Score	0.150	1.000	0.564	1.000

Table 5.4-3 Results of the generalised linear model testing for the effect of distance for sediment nutrient fluxes for surveys 4 and 5 at the Table Head Central, Gordon, Strahan and Franklin leases.

	Table Head Central		**Gordon**		**Strahan**		**Franklin**	
	F ratio	**p**	**F ratio**	**p**	**F ratio**	**p**	**F ratio**	**p**
Dissolved Oxygen								
Survey 4	46.694	**0.006**	341.58	**<0.001**	13.879	**0.034**	21.401	**0.002**
Survey 5	139.21	**0.001**	7.38	0.073	7.006	0.077	63.094	**0.004**
Dissolved Inorganic								
Survey 4	10.381	**0.049**	5.777	0.096	44.074	**0.007**	25.753	**<0.001**
Survey 5	19.834	**0.021**	3.279	0.168	0.017	0.905		
Ammonium								
Survey 4	31.653	**0.011**	10.977	**0.045**	19.796	**0.021**	6.370	**0.036**
Survey 5	27.963	**0.013**	16.229	**0.027**	83.533	**0.003**	26.736	**0.014**
Nitrate + Nitrite								
Survey 4	21.067	**0.019**	37.558	**0.009**	1.588	0.297	1.028	0.340
Survey 5	26.467	**0.014**	0.005	0.948	2.268	0.229	43.827	**0.007**
Phosphate								
Survey 4	20.314	**0.020**	10.819	**0.046**	19.903	**0.021**	27.302	**<0.001**
Survey 5	0.318	0.613	0.589	0.499	28.695	**0.013**	11.099	**0.045**

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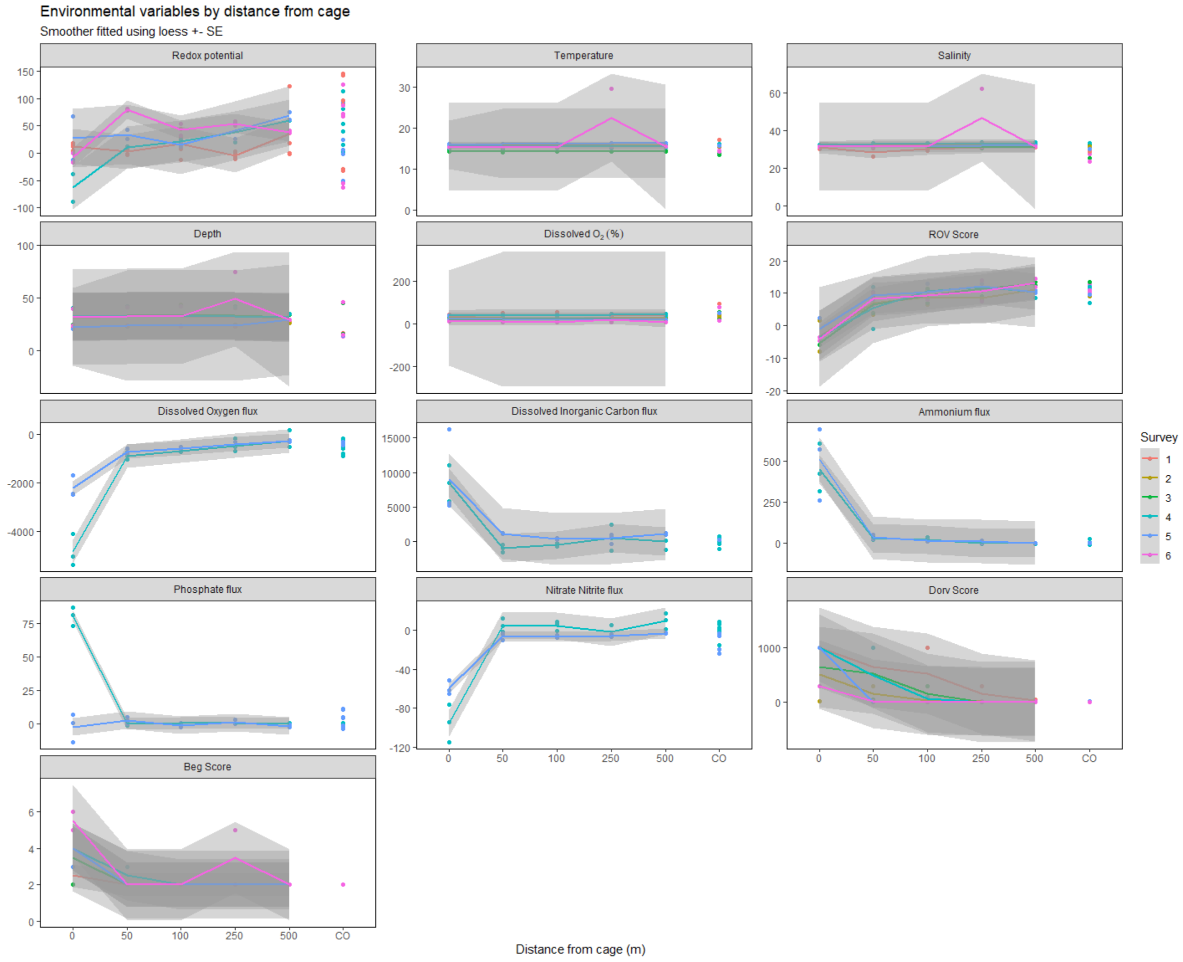
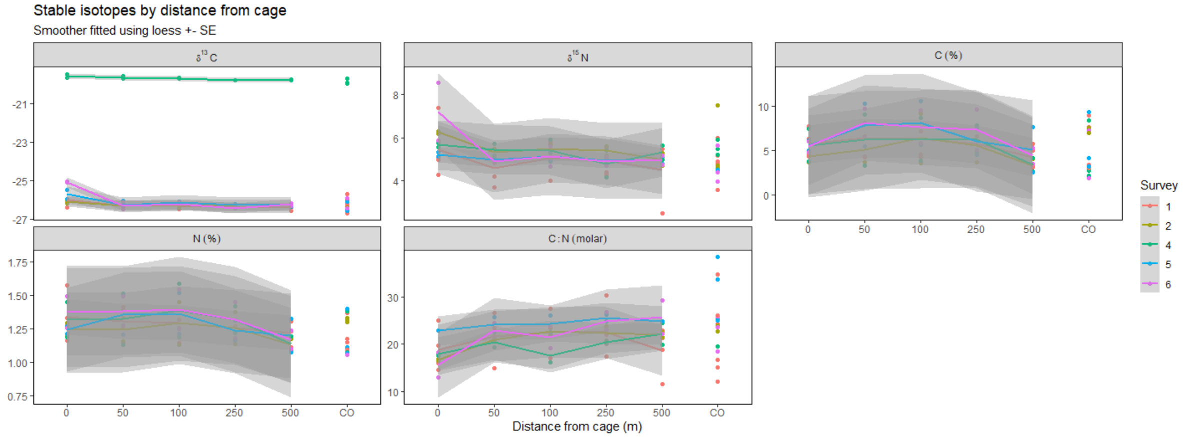
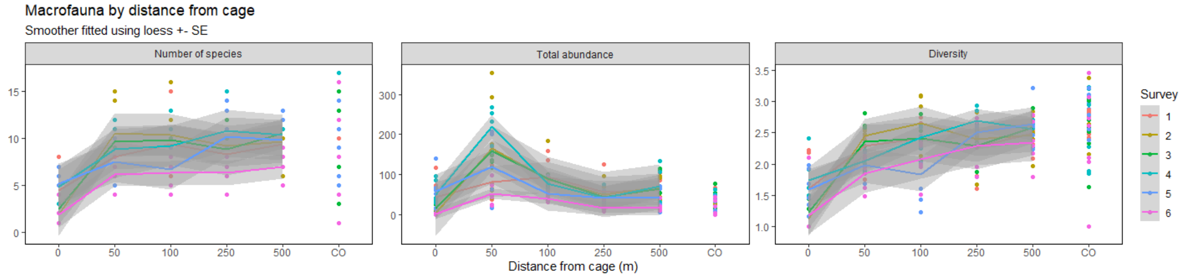
White CA, Bannister RJ, Dworjanyn SA, Husa V, Nichols PD, Kutti T, Dempster T (2017) Consumption of aquaculture waste affects the fatty acid metabolism of a benthic invertebrate. Science of the Total Environment 586:1170-1181

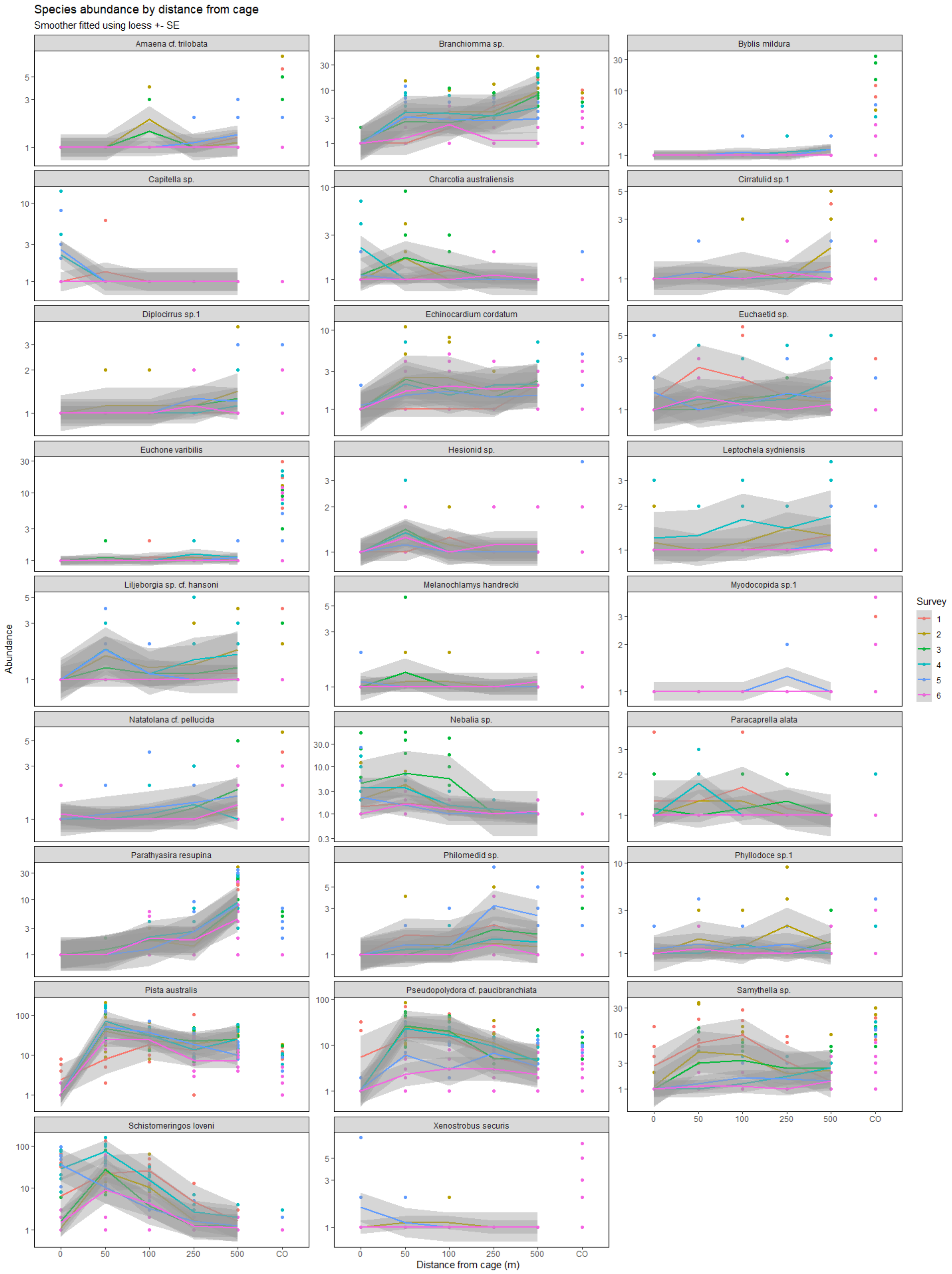
5.4.4 Appendix

5.4.4.1 Table Head Central

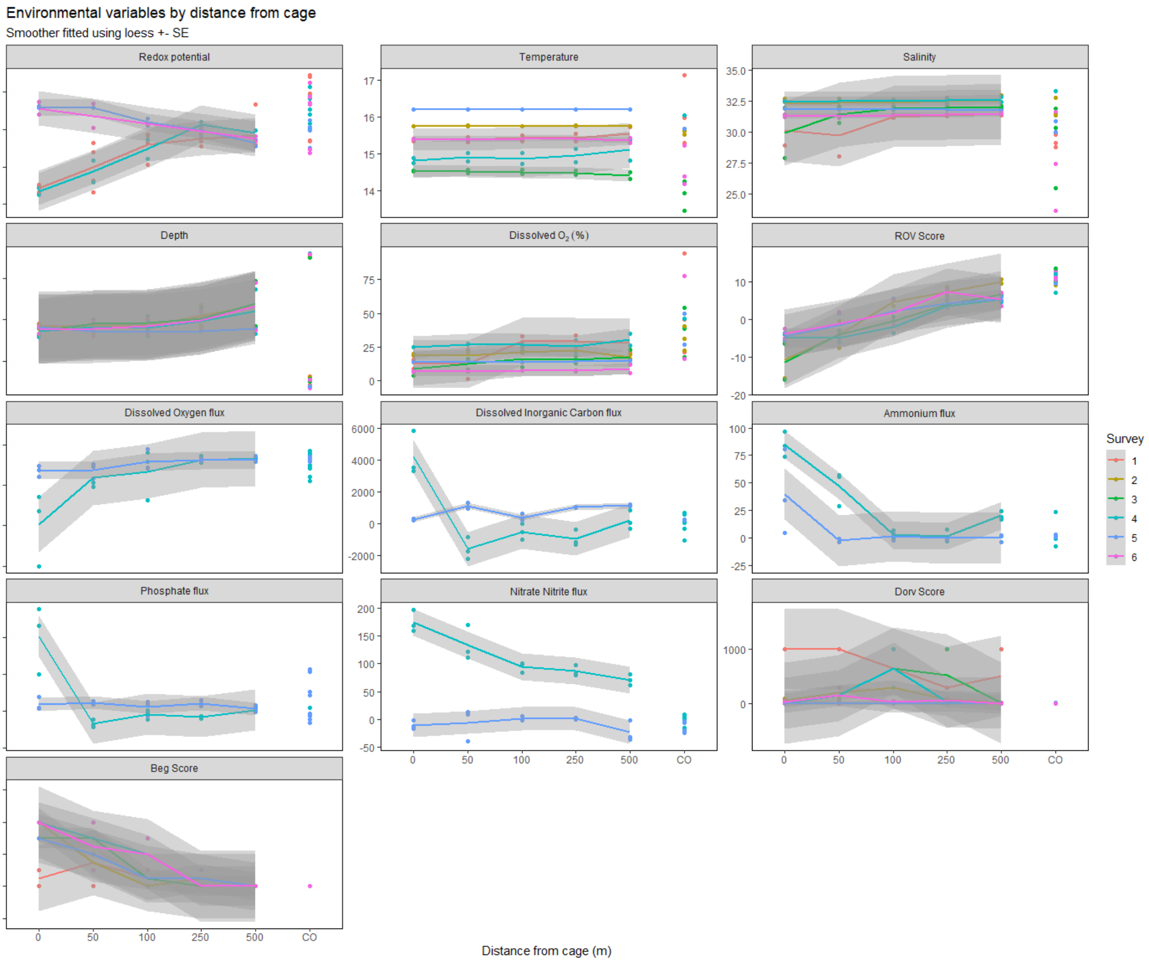


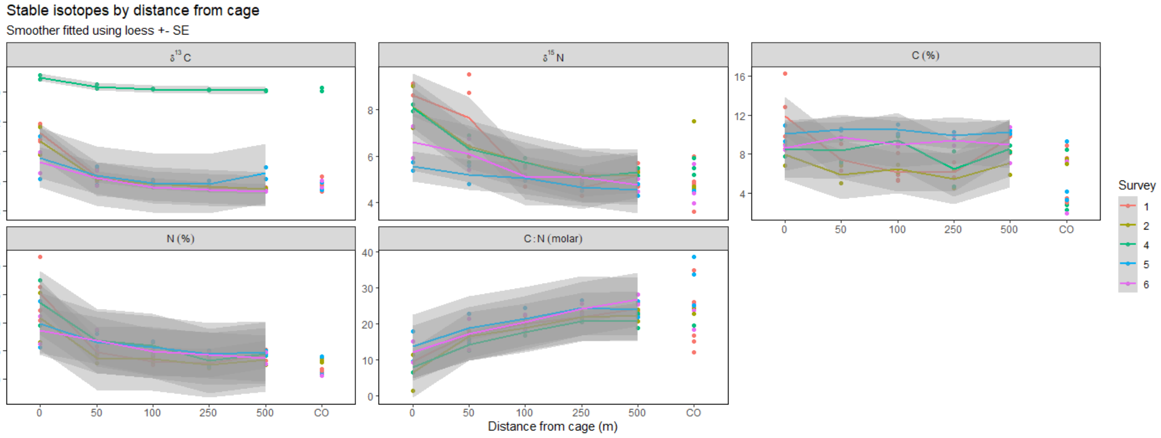


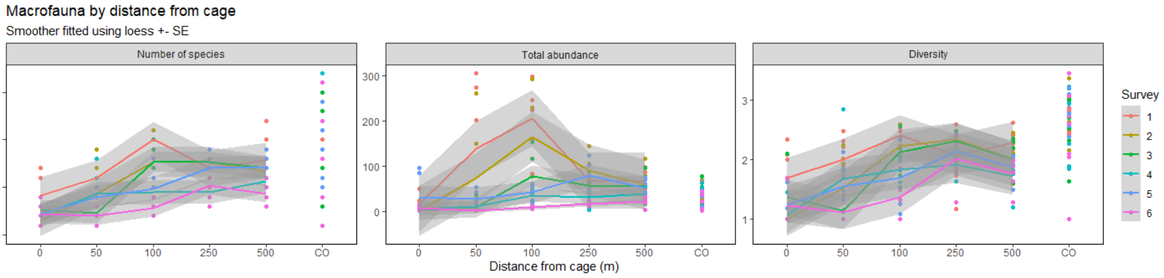


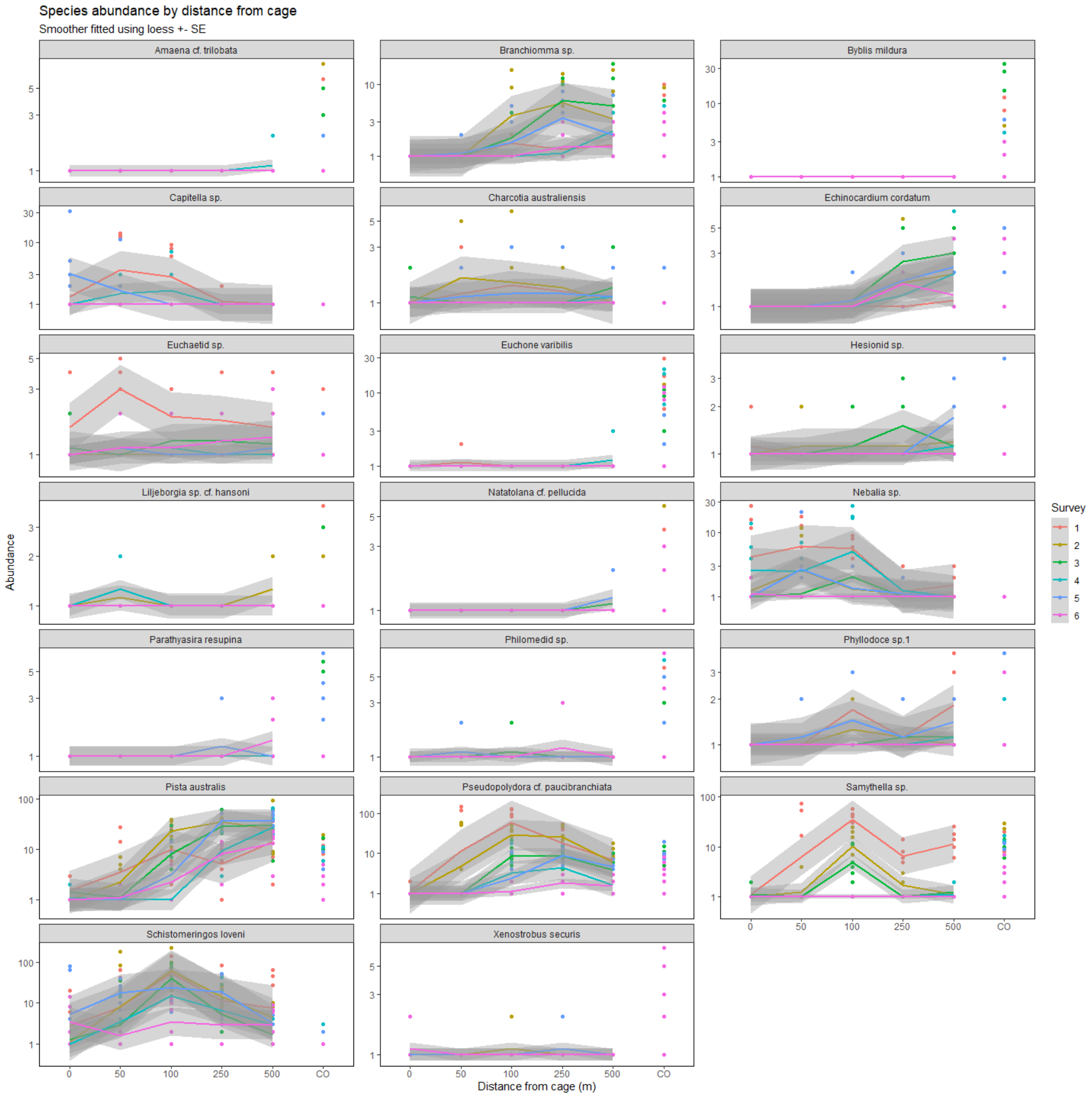

Species abundance by distance from cage
Smoother fitted using loess +- SE
Amaena cf. trilobata
Branchiomma sp.
Byblis mildura
Capitella sp.
Charcotia australiensis
Cirratulid sp.1
Diplocirrus sp.1
Echinocardium cordatum
Euchaetid sp.
Euchone varibilis
Hesionid sp.
Leptochela sydniensis
Liljeborgia sp. cf. hansoni
Melanochlamys handrecki
Myodocopida sp.1
Natatolana cf. pellucida
Nebalia sp.
Paracaprella alata
Parathyasira resupina
Philomedid sp.
Phyllodoce sp.1
Pista australis
Pseudopolydora cf. paucibranchiata
Samythella sp.
Schistomeringos loveni
Xenostrobus securis
Abundance
Distance from cage (m)
0
50
100
250
500
CO
Survey
1
2
3
4
5
6

5.4.4.2 Gordon

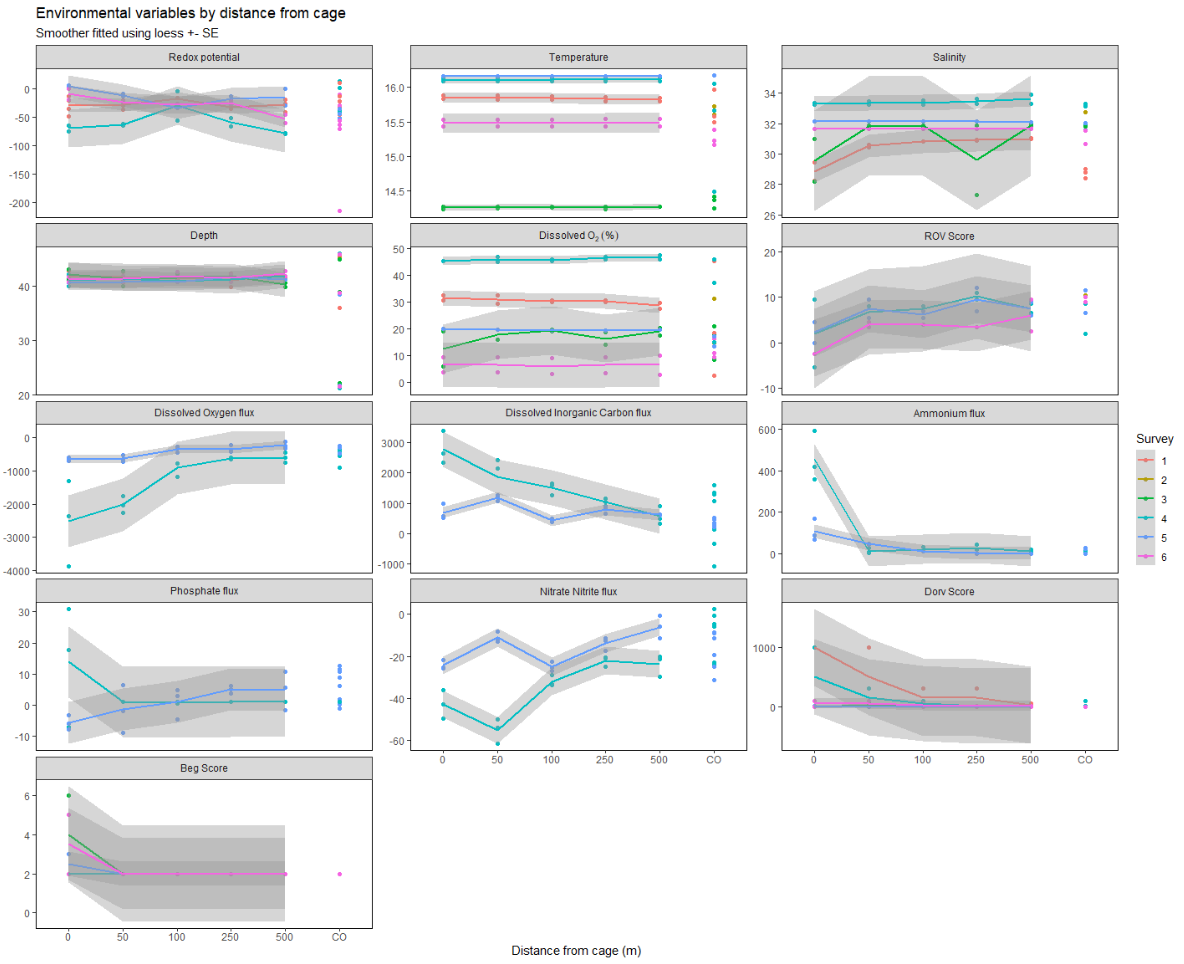

Environmental variables by distance from cage
Smoother fitted using loess +- SE
Redox potential
Temperature
Salinity
Depth
Dissolved O2 (%)
ROV Score
Dissolved Oxygen flux
Dissolved Inorganic Carbon flux
Ammonium flux
Phosphate flux
Nitrate Nitrite flux
Dorv Score
Beg Score
0
50
100
250
500
CO
Distance from cage (m)
Survey
1
2
3
4
5
6

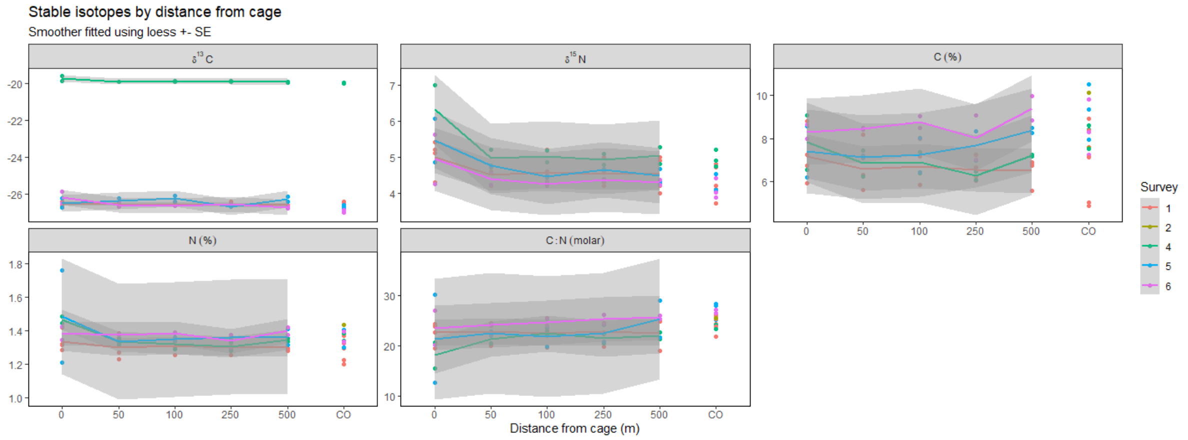

Stable isotopes by distance from cage
Smoother fitted using loess +- SE
δ13C
δ15N
C (%)
N (%)
C : N (molar)
0
50
100
250
500
CO
Distance from cage (m)
Survey
1
2
4
5
6

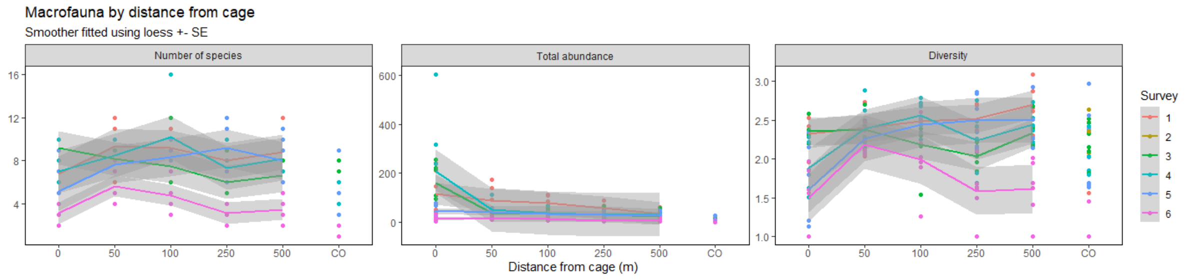

Macrofauna by distance from cage
Smoother fitted using loess +- SE
Number of species
Total abundance
Diversity
0
50
100
250
500
CO
Distance from cage (m)
Survey
1
2
3
4
5
6

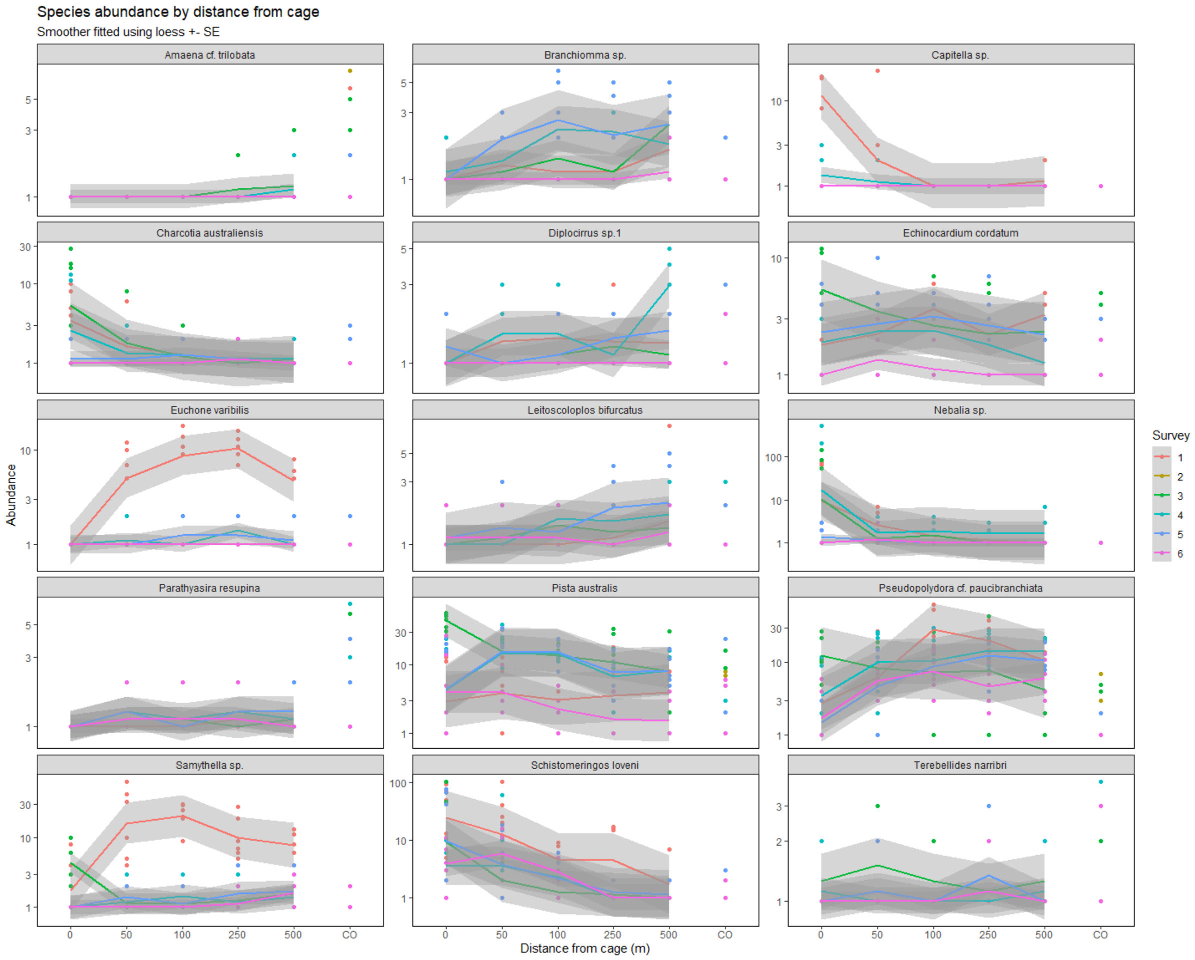

Species abundance by distance from cage
Smoother fitted using loess +- SE
Amaena cf. trilobata
Branchiomma sp.
Byblis mildura
Capitella sp.
Charcotia australiensis
Echinocardium cordatum
Euchaetid sp.
Euchone varibilis
Hesionid sp.
Liljeborgia sp. cf. hansoni
Natatolana cf. pellucida
Nebalia sp.
Parathyasira resupina
Philomedid sp.
Phyllodoce sp.1
Pista australis
Pseudopolydora cf. paucibranchiata
Samythella sp.
Schistomeringos loveni
Xenostrobus securis
Abundance
Distance from cage (m)
0
50
100
250
500
CO
Survey
1
2
3
4
5
6

5.4.4.3 Strahan








Species abundance by distance from cage
Smoother fitted using loess +- SE
Amaena cf. trilobata
Branchiomma sp.
Capitella sp.
Charcotia australiensis
Diplocirrus sp.1
Echinocardium cordatum
Euchone varibilis
Leitoscoloplos bifurcatus
Nebalia sp.
Parathyasira resupina
Pista australis
Pseudopolydora cf. paucibranchiata
Samythella sp.
Schistomeringos loveni
Terebellides narribri
Abundance
Distance from cage (m)
0
50
100
250
500
CO
1
2
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10
30
100
Survey
1
2
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5
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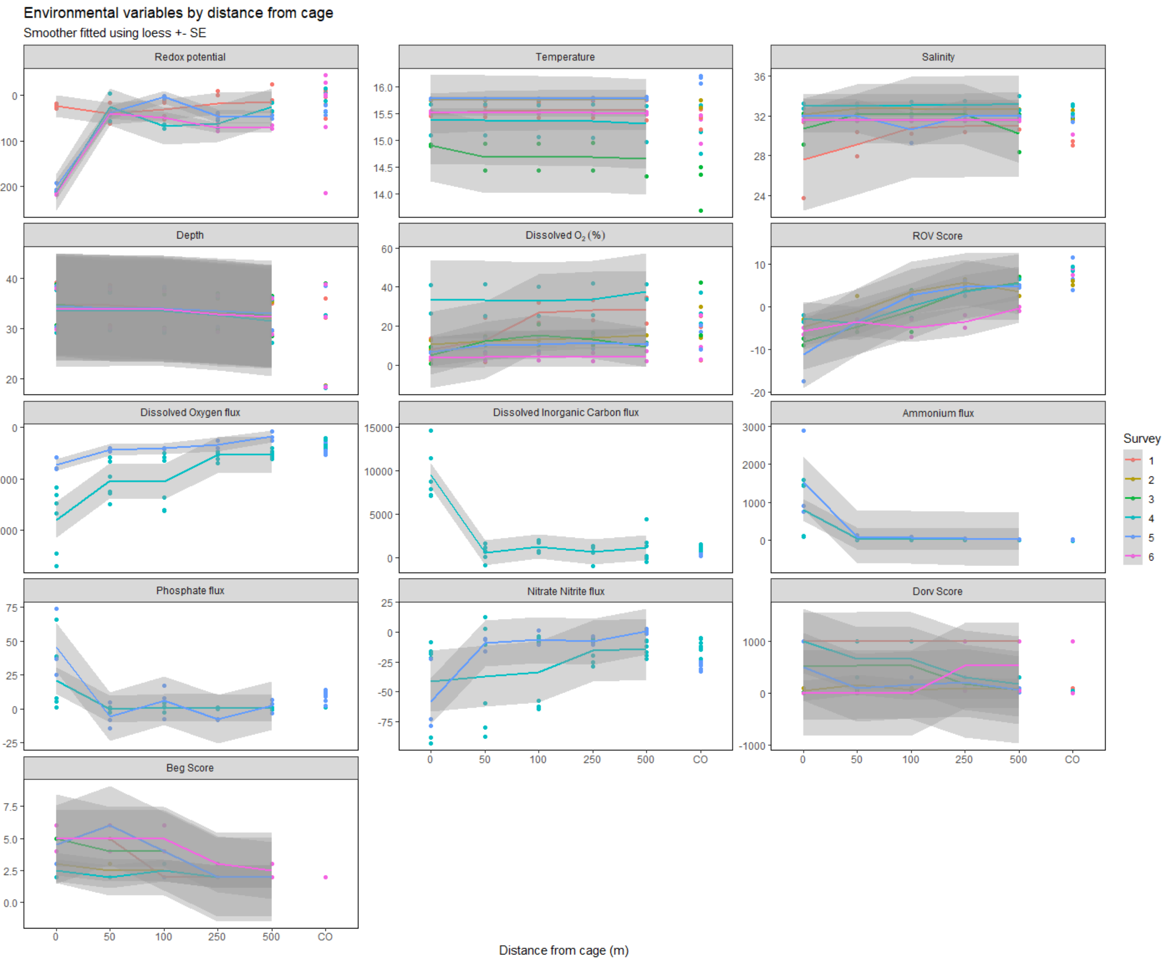
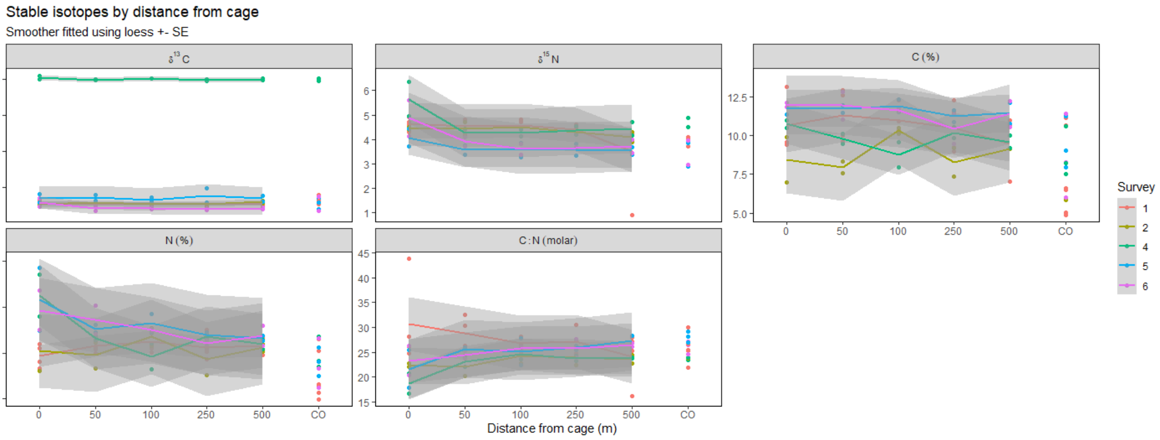
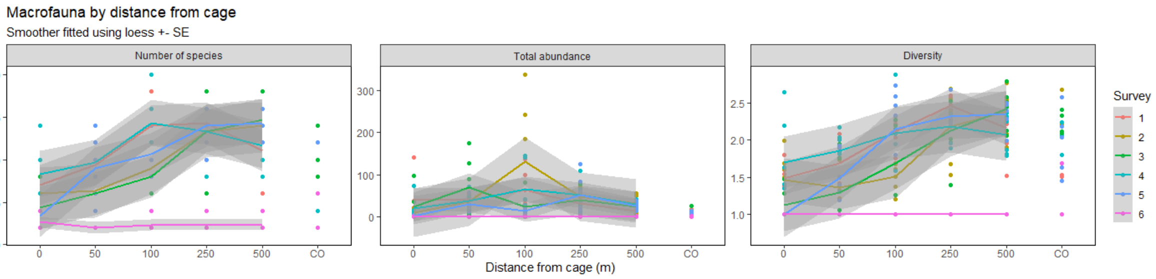
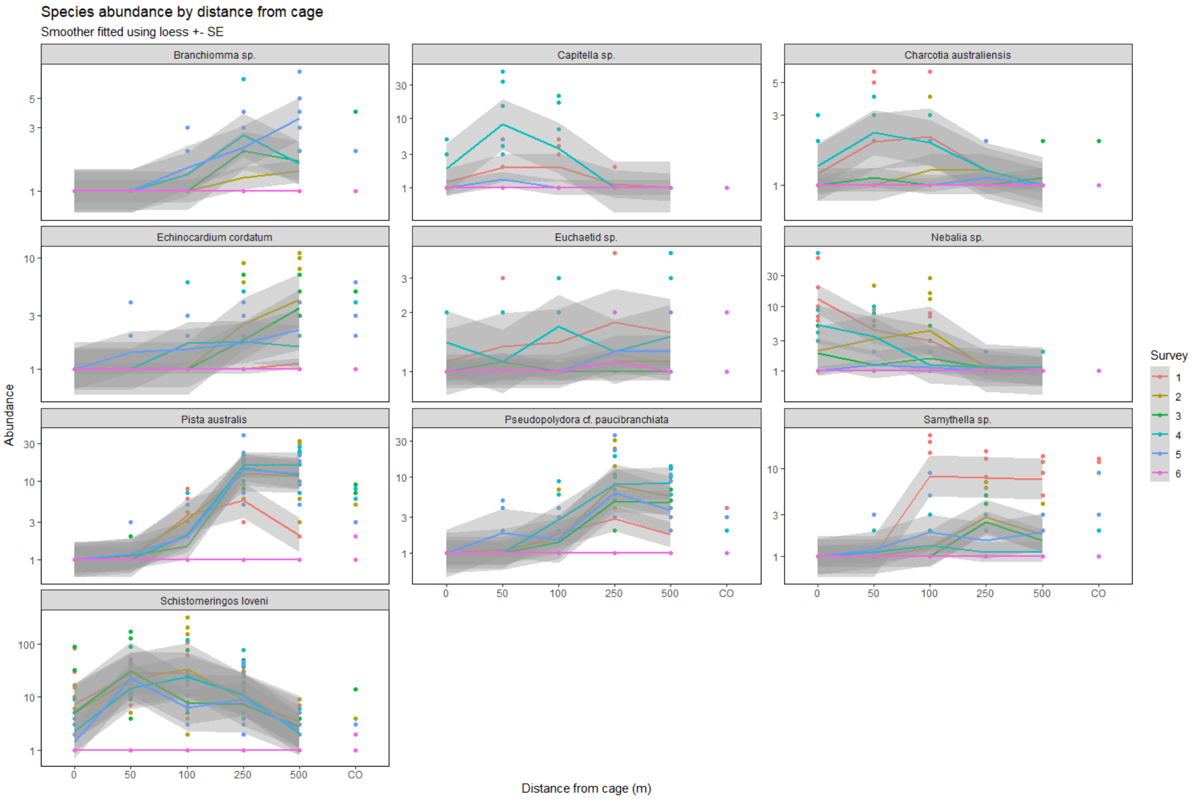
5.4.4.4 Franklin








Species abundance by distance from cage
Smoother fitted using loess +- SE
Branchiomma sp.
Capitella sp.
Charcotia australiensis
Echinocardium cordatum
Euchaetid sp.
Nebalia sp.
Pista australis
Pseudopolydora cf. paucibranchiata
Samythella sp.
Schistomeringos loveni
Abundance
Distance from cage (m)
0
50
100
250
500
CO
1
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Survey
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5.5 Synthesis

Tasmanian marine farming practices, monitoring requirements and regulations have been underpinned by extensive research that has documented the local scale effects of organic enrichment on sediment health. This foundational research established the 35 m compliance point for ongoing monitoring, developed a suite of biotic and abiotic indicators that identify different stages of impact and recovery, and validated visual techniques for assessing sediment health.

In the time since this research on the local scale impacts of particulate waste was used to guide sustainable farming practices, the industry has continued to grow and evolve. Production approaches have advanced (e.g., larger cages, more efficient and automated feeding systems, increased smolt sizes) and farms have expanded into new areas. Maintaining the community and industry expectations for high environmental performance requires an understanding of how these changes in technology and production and farming in new areas/environments might change environmental interactions. The recent expansion of farming in Macquarie Harbour and the benthic response observed provided a very timely reminder for this need. The monitoring approach initially implemented in Macquarie Harbour was based on extensive understanding of sediment interactions with farming practices in the established farming regions of southern Tasmania. However, the results of a tactical FRDC project (2014- 038) demonstrated that the response of benthic communities in Macquarie Harbour differed from that observed in southern Tasmania, highlighting the need to validate local scale monitoring approaches. This study was designed to assess the local scale interactions in the two new growing areas and environments in southern Tasmania (Storm Bay and the Southern Channel) as well as Macquarie Harbour, with the intention to ensure that monitoring and the indicators of impact remain fit for purpose.

5.5.1 Comparison of benthic communities and environmental conditions

A key finding of this study is that the benthic ecology and environmental conditions in Macquarie Harbour were distinct from the other two new growing areas in southern Tasmania. The sediments in Macquarie Harbour were inherently depauperate (O’Connor et al. 1996; Talman et al. 1996; Edgar et al. 1999), with all sites showing low faunal abundance, species richness and diversity (Table 5.5-1) relative to other sites in southern Tasmania. Only 5-29 individuals per grab were found in Macquarie Harbour control samples, as compared with an average of 130 individuals per grab at the Lippies lease control sites and 94-115 individuals per grab at the Trumpeter and Storm Bay 1 control sites. The average number of taxa found in Macquarie Harbour grabs was 5, compared with around 34 taxa at the Lippies and Storm Bay control sites; this equated to a total 52, 165 and 198 individual taxa being recorded over the course of the study at Macquarie Harbour, Lippies and Storm Bay control sites. In Macquarie Harbour polychaetes dominated, making up between 70-80% of the community, with a variable but much smaller presence of crustaceans, molluscs, and echinoderms. In contrast, control communities from the southern leases were more taxonomically diverse: Lippies - polychaetes 34%, crustaceans 47%, molluscs 13% and echinoderms 3%; Trumpeter and Storm Bay 1 - polychaetes 18-25%, crustaceans 41-59%, cnidarians 4-22%, molluscs 10-15% and echinoderms 1-2% (see Table 5.5-2 for a comparison of the most abundant taxa and phyla across the study regions). Whilst there are a range of other factors driving the differences in benthic community composition between study regions (e.g. lower productivity of west coast estuaries) differences in sediment grain size composition (see Table 5.5-3) will have a major influence on differentiating the macrofaunal communities found in each region (Anderson 2008). Macquarie Harbour sediments were demonstrably finer than those at the other two study regions thus explaining the dominance of surface deposit feeding polychaetes at the control sites in Macquarie Harbour. In contrast, the coarser sediments in Storm Bay and in the Southern Channel better suit the mobile crustaceans, whilst the higher proportion of silts at the Southern Channel control sites might also explain the greater representation of polychaetes relative to Storm Bay.

Differences in the other abiotic measures also highlighted the different environmental conditions in each region. Sediment redox levels revealed the highly reducing environment in Macquarie Harbour sediments (redox potential typically <50 mV) relative to the oxidising environment at the two southern study regions (redox potential typically >300 mV). These conditions in Macquarie Harbour are a function of the highly stratified water column, deep central basin and shallow entrance to the ocean, which results in reduced mixing of bottom waters and naturally low dissolved oxygen conditions. In contrast the other two study regions are in relatively open, well mixed areas where oxygen concentrations of bottom water are typically high. Over the course of the study the average bottom water concentration in Macquarie Harbour sites was typically less than 2.5 O_2 mg l^{-1} (~30% saturation) whilst at the Storm Bay and Southern Channel sites levels were consistently above 7 O_2 mg l^{-1} (~85% saturation). It is also important to note that within Macquarie Harbour there was a gradient in the bottom water oxygen concentrations and subsequently redox levels, with levels declining with distance from the harbour entrance. Indicators and management expectations need to take account of this gradient which will influence the prevailing ecology and potential response to organic enrichment.

Redox potential can be a reliable indicator of the redox reactions that underpin the cycling of carbon and nitrogen in coastal and inland waters. In coastal sediments, particularly in low oxygen environments, anaerobic respiration of carbon can naturally dominate, but a change in the respiration/ oxidation potential of the sediments can also be a good indicator of a decline in environmental condition, so it is important to understand the level and context for any such change. Across the three study regions rates of respiration (oxygen consumption) were similar, averaging ~400-500 µmol O_2 m^{-2} h^{-1}, as were the rates of dissolved inorganic carbon production (DIC). However, in Macquarie Harbour the rate of DIC production varied markedly along the harbour, typically being much higher than the rate of oxygen consumption at sites further from the harbour entrance, indicating that anaerobic respiration of organic matter was more common in this region. Anaerobic respiration will be exacerbated where bottom water oxygen concentrations are low and where organic enrichment of the sediments is high.

Lower oxygen concentrations will also affect the pathways that the system breaks down organic matter and processes nitrogen, as when oxygen concentrations in sediments are lower, the process of nitrification (conversion of ammonia to nitrate) becomes limited, and the denitrification process (conversion of nitrate to nitrogen gas) must rely on sourcing nitrate from the water column rather than from nitrification in the sediments (see Cornwell et al. 1999). This study found that the fluxes of nitrate and nitrite at control sites were predominately directed into the sediment at Macquarie Harbour, and more so when moving further into the harbour, but out of the sediment at the Storm Bay and Southern Channel control sites. Ross et al. (2016a) and Maxey et al. (2016) found that populations of nitrifiers (notably ammonia oxidising archaea) and rates of nitrification, respectively, increased markedly in the water column with depth in Macquarie Harbour. Put simply, the results indicate that the microbial transformation of ammonia to nitrate (nitrification) is occurring in the water column and the transformation of nitrate to nitrogen gas (denitrification) in the sediments in Macquarie Harbour, whereas in the south, both processes appear to occur primarily in the sediments.

Interestingly, sulphide concentrations were generally lower in Macquarie Harbour at both control and enriched cage sites relative to levels observed in the southern sites. Under organically enriched conditions sulphate reduction to sulphide will increase due to the reduced supply of oxygen, and this is typically reflected in elevated sulphide concentrations in the sediments under active salmonid aquaculture cages. However, the concentration of sulphides can also be influenced by interaction with iron (Łukawska-Matuszewska et al. 2019). Macquarie Harbour is known to be rich in metals (including iron) derived from historic mining activities. Thus, the lower sulphide concentrations measured in the harbour at both control and enriched sites could be a function of the sulphide being rapidly scavenged by iron minerals, if so then sulphide would be a less reliable indicator of enrichment and farm impacts in Macquarie Harbour.

The carbon and nitrogen signatures of the sediments provided further insight into the differences in the sediment environments and the potential sources of background organic material in each of the regions. Marine organic matter usually has a $\delta^{15}N$ signature of 5-7‰ and a $\delta^{13}C$ signature of -22 to -18‰, whereas terrestrial and freshwater derived organic matter generally has $\delta^{15}N$ values < 4‰ and a $\delta^{13}C$ of -25 to -33‰ (see Middelburg & Nieuwenhuize 1998). The sediments in Macquarie Harbour had more depleted $\delta^{15}N$ (3-4‰) and $\delta^{13}C$ (-26 to -27‰) signatures compared with Storm Bay ($\delta^{15}N$ 8-9‰ and $\delta^{13}C$ -20 to -21‰) and the Southern Channel ($\delta^{15}N$ 8.5‰ and $\delta^{13}C$ -23‰) highlighting a far greater contribution of terrestrial and freshwater inputs to the organic matter pool in Macquarie Harbour. Furthermore, the $\delta^{15}N$ and $\delta^{13}C$ signatures of sediments at sites closer to the harbour entrance were more enriched, consistent with an increased contribution from marine sources. In the south, the more depleted $\delta^{13}C$ of sediments at the Southern Channel relative to Storm Bay control sites may reflect a contribution of terrestrial inputs from the Huon estuary.

5.5.2 Response to farming

Consistent with the findings from numerous other studies, both local (e.g. Macleod et al. 2004; Edgar et al. 2005) and overseas (e.g. Karakassis et al. 2000; Tomassetti et al. 2005; 2009; Keeley et al. 2015), community structure of the macrobenthic fauna was found to be a sensitive and reliable measure of sediment condition. However, there were some notable differences in the response to enrichment that would need to be considered in developing and applying infaunal indicators of performance, particularly in Macquarie Harbour.

The community response to enrichment in all the study locations was predictable and could be aligned with levels of organic enrichment. The response at the study leases in Southern Tasmania was like that previously outlined by Macleod and Forbes (2004), with clearly identifiable stages and indicators of deterioration and recovery. The most enriched and impacted cage sites were still typically dominated by the opportunistic polychaete *Capitella* sp. but occasionally other taxa, such as the *Mediomastus* sp. were also locally abundant (e.g. at the Lippies lease in the Southern Channel). Having said this, *Capitella sp.* was far more abundant (~x4) at the cage sites at Lippies than at the Storm Bay leases, indicating the need to consider both the species mix and absolute abundances in a regional context (Table 5.5-2). As noted earlier, sediment type has a significant influence on both the form and function of the macrofaunal communities (Snelgrove and Butman 1994), consequently some regional modification of potential indicator species may be required, particularly in areas of minor or moderate impact. Capitellidae prefer more depositional, siltier sediments such as those found at Lippies (Table 5.5-3). The bivalve *Mytilus galloprovincialis* (the common blue mussel) and amphipod *Jassa* sp. were also characteristic of the cage sites at the southern leases. However, these species are common net fouling species and so as Edgar et al. (2005) previously reported are likely a result of dislodgement from the overhead nets during cleaning.

There was a notable increase in certain taxa at the cage sites following a period of fallowing at the Southern sites, and these were typically the same species that were most abundant at the sites 35 m from the cage. These included the polychaetes *Pectinaria antipoda, Pectinaria cf. dodeka, Mediomastus* sp. and *Perinereis* sp., heart urchin *Echinocardium cordatum*, amphipods from the family Phoxocephalidae, the dogwhelk *Nassarius nigellus* and the two introduced bivalves *Corbula gibba* and *Theora lubrica*. These species were likely responding opportunistically to the low to moderate levels of organic enrichment in the sediments either, after fallowing or with distance from stocked cages. Macleod et al. (2008) found that the gastropod *Nassarius nigellus*, introduced bivalves *Theora lubrica* and *Corbula gibba,* and amphipods from the family Phoxocephalidae were all indicative of recovering sediments and minor/moderate impacts and that *Echinocardium cordatum* appeared to thrive at low levels of enrichment. Consequently, it would be fair to consider these species as potential indicators of low/ moderate enrichment in these new farming regions. Interestingly, studies from the Northern Hemisphere (Pearson & Rosenberg 1978; Borja et al. 2000) have considered the genera *Echinocardium* and *Pectinaria* to be indicative of undisturbed conditions. Macleod et al. (2004) and Edgar et al. (2005) both concluded that whilst biotic patterns observed along gradients of organic enrichment were broadly consistent with the generalised patterns noted in the Northern Hemisphere, there were some notable exceptions. They suggested that perhaps the reference conditions in the Northern Hemisphere were naturally affected by higher organic loadings than Tasmanian reference conditions, regardless of finfish influences, and Macleod et al. (2004) observed that comparable levels of impact inferred by changes in community structure in Tasmania were associated with considerably lower levels of chemical degradation, as measured by sulphide levels, than elsewhere.

Whilst under or close to the cages the community was dominated by relatively mobile and enrichment tolerant deposit feeding species, as you move away from the cage, the community composition changed in a manner consistent with decreasing levels of enrichment. This transitory community was found both as you move away from the cages and where there was evidence of low-level organic enrichment (e.g. after fallowing). It generally included a mix of surface deposit feeders, suspension feeders and scavengers able to rapidly take advantage of the increased food supply, but which were more sensitive (less tolerant) to the reduced sediments often found under cages. Whilst the general pattern of community response was evident in all regions, once again the specific species varied between leases/ regions. At the

Lippies lease in the Southern Channel the species that increased in abundance at the intermediate distances (100-200 m) included the polychaetes *Prionospio kulin*, *Scalibregma cf. inflatum*, *Terebellides kowinka* and *Ampharetid* sp.1, the tanaid crustacean *Apseudid* sp.2 and bivalve *Channelaxinus adelaideanus.* At the Storm Bay leases the species that were more common at the intermediate distances were the spionid polychaetes *Spionid* sp.4 and *Dipolydora giardi,* onuphid polychaete *Hirsutonuphis intermedia,* amphipods *Tipimegus cf. thalerus* and *Hippomedon cf. hippolyte,* ostracod *Euphilomedes* sp., brittle star *Ophiura cf. kinbergi* and anemone *Edwardsii* sp. A number of these species (or genera) have previously been identified as characteristic of the transitory community between heavily impacted and unimpacted communities (e.g., Pearson & Rosenberg 1978; Macleod et al. 2008; Edgar et al. 2005; 2010). Further from the cages (500 m +) where there was little or no evidence of the impacts of organic enrichment, the bivalve *Pratulum thetidis* and several amphipod species, notably *Ampelisca cf. australis*, *Byblis mildura, Isaeid* sp.3 and *Aorid* sp.1 were characteristic of these assemblages in Storm Bay, whilst the brittle star *Amphiura elandiformis* was also characteristic of the unimpacted assemblages at the Lippies lease. Several of these species are consistent with the species matrix previously described as “low-impact” indicators by Macleod and Forbes (2004).

In the current study, changes in both the community composition and organic matter mineralisation rates (rates of respiration and nitrogen cycling) were evident up to 200 m from the cages at all three of the southern leases. Whilst this is a somewhat larger spatial zone of effect than indicated in the studies by Macleod et al (2004, 2006) it is consistent with findings of a suite of other and more recent studies in Tasmania and abroad. In Canada, the footprint of fish farms in low flow, depositional sites of similar depths to that of the current study was found to extend between 40-150 m from the cages (e.g. Brooks et al. 2002; Hargrave 2003). A state-wide meta-analysis of benthic monitoring data associated with salmon farms in Tasmania also found that subtle effects on macrofaunal communities were evident out to at least 50-150 m from the cage (Edgar et al. 2010). Assessments from more dispersive sites in comparable depths (30-50 m) in New Zealand found effects to 300-400 m from the cages (Keeley et al. 2013) and even out to 600 m in a later study (Keeley et al. 2019).

The farm footprint in the current study was generally larger than previously documented by Macleod et al. (2004, 2006) and this is likely a result of changes in the prevailing environmental conditions at the newer leases as well as changes in farming practices (i.e. cage design and stocking levels). The depth of the sites and the mean bottom current speeds measured at the newer leases in southern Tasmania were increased relative to the original studies (Macleod et al. 2004). The current threshold required for particulate resuspension/ dispersion has long been considered to be 9.5 cm s^{-1} in modelling studies (Cromey et al. 2002); however, resuspension of faecal particles is reported at current speeds of between 7 to 15 cm s^{-1} (Cromey et al. 2002b). This is pertinent given that feed wastage on farms is now generally very low (typically <1%) and faeces make up proportionally most of the waste. It is also likely that intermittent periods of elevated flow will also have a major effect on overall dispersion; at Lippies and Storm Bay mean bottom water current speeds were 9-10 cm s^{-1} and 6-7 cm s^{-1} respectively, and maximum flows regularly exceeded these mean values.

Consistent with the higher bottom water current speeds, the footprint appeared to be a more diffuse (less clear) at the Lippies site compared with that seen at Storm Bay. However, at the Lippies site we also need to consider that farm history and previous production levels may also have had an influence, given the lower production at Lippies relative to the Storm Bay leases at the time of the surveys.

The community transition with distance from the source of organic matter in Macquarie Harbour was quite different due to the unique environmental conditions and benthic ecology in this region. The highly enriched cage sites were still dominated by opportunistic species, but rather than capitellids, in Macquarie Harbour it was the dorvilleid polychaetes *Schistomeringos loveni* and *Ophryotrocha shieldsi* and the leptostracaen crustacean *Nebalia* sp., which characterised these communities. All these opportunistic species are characterised by their abilities to move through the water column and/or burrow into the sediment and alter their behaviour in response to changes in the physicochemical properties of the sediment, meaning that they can quickly adapt to and exploit changes in organic loading. Despite both dorvilleid species being found at the cage sites, their distribution appeared to reflect different sensitivities to highly reduced sediments conditions. The colony forming dorvilleid *O. shieldsi* was rarely detected in grabs but often recorded in video footage at 0 m (typically in conjunction with the presence of *Beggiatoa*) and occasionally at 50 m from the cage, whereas *S. loveni* abundances tended to peak at 50-100 m from cages. The terebellid *Pista australis*, spionid *Pseudopolydora cf. paucibranchiata*, sabellid *Branchiomma* sp., ampharetid *Samythella* sp., echinoderm *Echinocardium cordatum* and amphipod *Charcotia australiensis* were all characteristic of the transitory community found between 50 and 500 m from the cage depending on production levels and location in the harbour. These species include a mix of sessile suspension feeders and surface deposit feeders that are likely to have a more limited capacity to move in and out of areas as the sediment quality changes. Although most of the species were different to those comprising the intermediate/ transitory communities in the south, the functionalities were similar and the list does include *Echinocardium cordatum* and spionid, terebellid and ampharetid polychaetes which are common across these regions.

Interestingly, Borja et al. (2000) considered the spionid polychaete *Pseudopolydora cf. paucibranchiata* a second order opportunist, congeners of the genera *Branchiomma,* and *Pista* and *Echinocardium cordatum* as sensitive to organic enrichment whilst in Macquarie Harbour these taxa were in greater abundance around the leases, presumably in response to increased food availability. This further highlights not just the regional differences, particularly with Macquarie Harbour, but also the notable exceptions in Tasmanian results and those reported for the Northern Hemisphere.

Another important consideration for Macquarie Harbour was the fact that there were significant differences in response between leases and surveys. This appeared to be largely driven by variation in local environmental conditions and differences in farm management. At the Strahan lease, where feed inputs were generally low and the lease was in a location that typically experienced higher bottom water oxygen concentrations, the transitory community appeared to be less tolerant of highly enriched conditions closer to the cage than the other leases. In contrast, at the Franklin lease where feed inputs were the greatest and bottom water oxygen concentrations often the lowest, the peak in opportunists and transitory community occurred further from the cage than was observed at the Strahan lease. At Table Head Central, the feed inputs were relatively high, but this lease was closest to the influence of the ocean, and it would seem that this proximity to the ocean may lessen the impact as the transition to the less tolerant species (more sensitive community) occurred at intermediate distances. Whilst the Gordon lease had the lowest feed inputs, the peak abundances of opportunists and transitory species extended further from the cage. We believe this result may reflect the longer (>20 years) history of farm production at this lease and as such the legacy of farming on the sediments.

5.5.3 Implications for management and monitoring

Marine farming development plan management controls stipulate 'there must be no unacceptable impacts 35 m outside the boundary of the marine farming lease area.' The 35 m point was chosen based on findings from a combination of overseas research and some preliminary research conducted in Tasmania (Ye et al. 1990) which suggested that if a cage was to be located at the lease boundary the environmental effects would negligible at 35m. Although the level of enrichment was significantly reduced at the 35 m[1] point relative to 0 m, in this study moderate levels of enrichment and significant effects were often evident at the 35 m position. The impacts at 35 m appear to be greater than described previously in Tasmania (Ye et al. 1990; Crawford et al. 2002; Macleod et al. 2004). Whilst it is difficult to ascertain the specific drivers of this for each site given the differences in locations, study design, farm history etc., the results would seem to suggest that the stronger bottom currents and more dispersive nature of the many of the sites in this study could have increased the footprint. In addition, the bigger cages and associated higher fish numbers would have an increased organic loading which would likely contribute to a potentially larger footprint. In the state-wide metanalysis of benthic monitoring undertaken by Edgar et al. (2010) between 1997-2003, it was noted that cages were on average 26-36 m in diameter; however, the cage size in the current study was between 32-76 m diameter, with cages at the southern leases being between 53-76 m. Leases are also typically larger now, with provision for multiple cage grids to support better stock management and fallowing, and so were between 75-200 ha in area compared to 20-60 ha at the time Edgar et al. (2010) undertook their assessment.

Regardless of the drivers, the larger footprints shown in the study highlight the importance of lease size and the layout and proximity of cage grids to each other and lease boundaries for environmental management. To put this into context, the average distance from the cages to the lease boundary in this study was 70 m at Lippies and Storm Bay 1, 180 m at Trumpeter and in Macquarie Harbour it ranged from 100-300 m depending on the lease. Whilst impacts were often evident at the 35 m compliance points (or transect sites at the comparable distance) they were generally diminished and far more subtle than closer to the cages. This is consistent with the Edgar et al. (2010) meta-analysis where only minor effects were detected at the 35 m compliance points, which in this case were generally 50-150 m from the cages. The results of this study suggest that at least 100 m between cage grids is needed to minimise the overlap of enrichment footprints and further facilitate recovery. Similarly, a distance of at least 100 m between cage grids and the lease boundary is recommended to minimise effects that extend beyond the lease boundary.

A key aim of the local scale study was to better understand the recovery responses of the sediments to farm management strategies such as fallowing. However, obtaining reliable and

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[1] Noting 50 m rather than 35 m from the cage was sampled in Macquarie Harbour

comparable data was often difficult given uncertainty over the timing of fallowing periods and the logistics of coordinating benthic surveys around farm production timelines. As a consequence, we have had to rely on retrospective analysis of differences in farming intensity and management actions/ timings to identify sediment responses. In the cases where surveys did correspond directly to periods of fallowing, the responses were largely consistent with prior studies and expectations. Sediment chemistry and function (e.g., organic matter processing) appeared to respond relatively rapidly to cessation of farming, and there was typically a decrease in the abundance of opportunistic species and a concomitant increase in the abundance of a range of the less tolerant taxa closer to the cages.

The response to fallowing in Macquarie Harbour was more complex due to the influence of bottom water dissolved oxygen concentrations. This was most evident in the final survey conducted in Spring 2016 when dissolved oxygen levels had declined to the lowest levels on record (see Ross & Macleod 2017). Faunal abundance and the number of species had declined and the presence of *Beggiatoa* had increased significantly across the harbour, but the magnitude of this change was greater further into the harbour where oxygen levels reached lower levels. Recovery also appeared slower along the same gradient from the harbour mouth[2], and at leases that had a longer or more intense farm production history. This highlights the complex interplay between two key factors, farm production levels and oxygen supply, which are well known to be important in governing the responsiveness of macrofaunal communities and the assimilative capacity of the benthic environment (Findlay & Watling 1997; Keeley et al 2013). Interestingly in Macquarie Harbour it appeared that farmers were more inclined to fallow individual cages rather than the whole grid/ lease than elsewhere, potentially responding to targeted observations of conditions underneath specific cages. However, our findings would suggest that given the more extensive and overlapping cage footprints evident in Macquarie Harbour, fallowing is likely to be more effective if carried out at the scale of the entire grid or lease.

A major focus of this study was to address whether the suite of biotic and abiotic measures previously recommended for environmental monitoring and management remain fit for purpose in new growing regions. The results confirmed that the macrobenthic invertebrate community structure is still the most sensitive, reliable, and informative measure of sediment conditions, and as such should remain an essential element of both baseline assessments and ongoing monitoring to inform both farm management and regulation. Whilst not likely to be part of routine monitoring, the direct measurement of sediment biogeochemical functioning proved to be a highly sensitive measure of organic matter mineralisation and could clearly inform the spatial extent of the enrichment footprint. This measure, consequently, would be a powerful tool should more detailed assessments be required. Redox and sulphide also remained useful indicators of sediment health, but it was noted that redox potential appeared to be less variable across the range of conditions assessed and as such would be a more reliable measure of change than sulphide concentrations. Sulphide remained an informative measure of relative change across sites, therefore, is likely to be more useful for describing spatial patterns of enrichment than temporal change. In Macquarie Harbour, the results showed that both redox and sulphide needed to be interpreted carefully and that the broader

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[2] There are exceptions depending on the depth under the lease and proximity to the deeper holes that are recharged more rapidly e.g. the Strahan lease

background environmental conditions at the time of the survey needed to be taken into consideration. Whilst the southern regions might have differences that reflect the broad localised changes in sediment structure and conditions in each region, the environmental responses at leases within Macquarie Harbour are dependant not only on the prevailing sediment structure and biogeochemistry but also on the temporal changes in hydrography and oxygen exchange within the system. Consequently, measurement of redox and sulphide within this system should be limited to describing spatial patterns of enrichment rather than temporal change.

Of the other abiotic parameters, the various measures of C and N in the sediments (e.g.% content, molar ratio and stable isotope signature) and sediment particle size analysis were informative when establishing the background environmental conditions in all study regions, and provide important context when describing monitoring results, but are not sensitive or reliable enough to monitor the enrichment footprint beyond the immediate proximity of the cages. The exception was the $\delta^{15}N$ isotopic signature of sediments which now appears to be a reasonably sensitive indicator of enrichment levels, particularly in the more marine locations where the contrast with the background signature has increased due to changes in feed composition with greater terrestrial inputs. Historically, marine derived fishmeal was the dominant protein source, and as a result the signature could not be readily distinguished from the background signature of $\delta^{15}N$ which would be typical of marine organic matter (Edgar et al. 2005; Macleod et al. 2004). However, with the change to more terrestrially derived nutrient sources in fish feeds (e.g., plant proteins) the contrast against the background $\delta^{15}N$ signature is generally clearer; this was particularly evident in the new marine leases in southeast Tasmania. In Macquarie Harbour, where the background $\delta^{15}N$ signature is already depleted because of significant natural freshwater/ terrestrial inputs, the signal was somewhat reduced and $\delta^{15}N$ only appeared to be a suitable indicator of very high concentrations of organic matter immediately adjacent to the cage; beyond this it did not appear to be readily distinguishable from the background signature.

Importantly, one of the most significant findings was the validation of the visual assessment methods of sediment health developed by Macleod and Forbes (2004) in new growing areas. This study clearly showed that, where site-specific criteria and weightings were applied, the visual health scores could clearly discern different impact levels in all the study regions. This was particularly pertinent in Macquarie Harbour where i) the sediments and faunal responses have been shown to be quite distinct, and ii) the reliability of the visual indicators had previously not been assessed against macrofaunal assessments. In Macquarie Harbour, the visual health scores clearly demonstrated the different impact gradients across leases and revealed the deterioration in sediment conditions in response to the major decline in bottom water dissolved oxygen levels in Spring 2016. That said, the visual health scores were less able to differentiate the more subtle effects identified by analysis of the faunal communities or process rate measurements. However, this may be less of a concern for regulatory monitoring, given that decision making is generally triggered by moderate to major effects rather than subtle changes. At present, the compliance requirements for visual assessment of sediment conditions are focused on identifying unacceptable impacts (e.g. presence of *Beggiatoa*, feed pellets, numerous opportunists, gas bubbling), but the results suggest that scoring a broader suite of ecological features from the same footage can provide a more holistic and informative measure of sediment health, consistent with the findings of Macleod and Forbes (2004). This would incur very little extra cost but would provide important additional information and context when interpretating the broader significance of the metrics used for regulation.

Macleod and Forbes (2004) provided guidelines for classification of a range of different levels of impact and stages of sediment conditions. They highlighted regional differences in the key biotic and abiotic indicators of impact/recovery stages based on two different environmental condition and substrate types - exposed sandy and sheltered mud sites. The sites in the southeast region appeared to broadly align with previous recommendations for more exposed sandy sites, both in terms of indicators and level of impact. However, there were local (regional) exceptions, particularly in the characterising species and especially the transitory species that reflected subtle differences in background environmental conditions between leases. Macquarie Harbour is a good example of this; whilst the general response to enrichment was consistent with expectations, and the stages of impact and recovery could be clearly defined, the indicator species, response levels and thresholds were distinct.

Although the findings of Macleod and Forbes (2004) were used to inform regulatory monitoring and compliance requirements, the purpose of the guidelines from that study was principally to assist with farm management and the interpretation of monitoring data. Likewise, it is hoped that the findings of this research will play a critical role in interpreting monitoring data in new growing regions, for both farm management and regulation. As such it is important that the inherent variability observed both within and between the regions in this study be understood and the environmental standards be adjusted to focus on change relative to baseline and reference conditions rather than fixed parameter ranges (e.g. as provided for redox and sulphide). The findings also clearly warrant further discussion on the level of change that defines ‘unacceptable’ when measured at compliance locations 35 m from the lease boundary. Consistent with the findings of Edgar et al. (2005), this study noted that under current regulations, conditions at compliance sites need to be very close to that observed immediately adjacent to the cages to be considered “unacceptable”, and we would suggest that perhaps the definition of “unacceptable” needs to be reviewed.

Finally, the study once again highlighted the importance of establishing baseline conditions, and the need to assess change against reference conditions. The effects of farming were evident and measurable out to distances of 200 m at the leases in the south and 500 m in Macquarie Harbour. Whilst these effects were often quite subtle relative to those observed adjacent to the cages, they still highlight that both sediment function and species composition were affected within these spatial ranges. This would suggest that reference locations should be at least 500 m from farms in the south and 1000 m away from farms in Macquarie Harbour. We recognise that this can be difficult when reference sites need to account for similar depths and substrate and need to address proximity to other farms. However, it is important as the comparison of change from baseline conditions prior to farming needs to differentiate farm induced change from natural variability. Given the importance of understanding regional scale impacts, further consideration should also be given to identifying monitoring reference sites in regions without salmonid farms.

5.5.4 Summary of Recommendations

- Visual assessment methodology can be enhanced to provide additional context for interpretation of compliance and regulation. The scoring of footage for additional ecological features would provide a more holistic and informative measure of sediment health.
- The guidelines provided by Macleod and Forbes (2004) remain a useful basis to inform management on the level of impact and recovery. However, given differences in environmental conditions and responses between some sites and regions, notably in Macquarie Harbour, there needs to be a level of site and region specificity to the guidelines and their interpretation.
- The establishment of robust baseline conditions and ongoing monitoring of reference conditions remains critical, particularly considering the inherent variability observed both within and between regions in this study. Environmental standards should focus on change relative to baseline and reference conditions rather than a suite of standardised/ fixed parameters.
- Further consideration should be given to the level of change that defines ‘unacceptable’ when measured at compliance locations 35 m from the lease boundary.
- We suggest that providing a buffer of at least 100 m between cage grids, and from cage grids to the lease boundary will help minimise the overlap of enrichment footprints and reduce effects beyond the lease boundary.
- Fallowing remains an important management practice to facilitate the recovery of sediments. We recommend fallowing periods of at least 3-6 months undertaken at the scale of entire grids or leases, particularly in locations/situations where enrichment footprints are larger and overlapping.
- The biotic and abiotic parameters currently used for environmental monitoring of salmon farms would appear to still be generally quite sensitive and fit for purpose. Macrofauna continue to be the most reliable indicator of sediment conditions, with the sediment chemistry such as redox and sulphide providing detailed and location dependent measures of the enrichment footprint. Other parameters, such as the various measures of C and N in the sediments and sediment particle size analysis are informative when establishing the background environmental conditions and provide context when describing monitoring results at a given site, but with the exception of the $\delta^{15}N$ isotopic signature, are not reliable indictors of farm impacts.

Table 5.5-1 Comparison of the key biotic and abiotic measures (mean ± SE) of sediment condition at each of the study leases at the cage (0 m), 35/50 m from cage and the control (CO) sites.

	Total Abundance (ind. per grab)			**Richness (taxa. per grab)**			**Diversity (H')**			**Sulphide (µM)***			**Redox (mV)**			**δ^{15}N**		
	Cage	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**
Macquarie Harbour																		
THC	28 (6)	132 (15)	29 (3)	2.4 (0.3)	7.4 (0.4)	8.1 (0.6)	0.4 (0.1)	1.2 (0.1)	1.6 (0.1)	26 (5)	11 (2)	2 (7)	-5 (13)	25 (10)	36 (12)	4.9 (0.3)	4 (0.2)	4.1 (0.2)
Gordon	10 (4)	43 (14)	29 (3)	1.4 (0.2)	2.8 (0.4)	8.1 (0.6)	0.3 (0.1)	0.5 (0.1)	1.6 (0.1)	13 (3)	10 (2)	2 (7)	-73 (36)	-45 (28)	36 (12)	6.6 (0.4)	5.5 (0.4)	4.1 (0.2)
Strahan	108 (24)	47 (7)	6 (1)	5.3 (0.5)	6.9 (0.4)	2.9 (0.3)	0.9 (0.1)	1.3 (0)	0.8 (0.1)	11 (3)	10 (4)	4 (6)	-27 (9)	-31 (6)	-38 (9)	4.3 (0.3)	3.6 (0.1)	3.4 (0.1)
Franklin	15 (5)	35 (6)	5 (1)	1.6 (0.2)	2.5 (0.3)	2.2 (0.2)	0.3 (0.1)	0.4 (0.1)	0.6 (0.1)	28 (6)	20 (4)	4 (11)	-136 (30)	-38 (6)	-20 (10)	3.7 (0.2)	3.2 (0.1)	2.9 (0.1)
average	*37 (6)*	*65 (7)*	*19 (2)*	*2.6 (0.2)*	*4.8 (0.3)*	*4.9 (0.3)*	*0.5 (0)*	*0.8 (0)*	*1 (0)*	*21 (2)*	*15 (2)*	*2 (12)*	*-60 (14)*	*-22 (9)*	*-5 (6)*	*4.9 (0.2)*	*4.1 (0.2)*	*3.4 (0.1)*
Southern Channel																		
Lippies	*1059 (160)*	*186 (21)*	*130 (11)*	*26.5 (1.6)*	*37.4 (1.7)*	*34.1 (1.6)*	*1.1 (0.2)*	*2.9 (0.1)*	*2.9 (0.1)*	*60 (17)*	*23 (12)*	*12 (3)*	*230 (31)*	*259 (29)*	*387 (24)*	6.7 (0.2)	8.1 (0.1)	8.5 (0)
Storm Bay																		
Trumpeter	226 (25)	94 (9)	94 (9)	19.9 (1.5)	26.8 (1.9)	34.6 (2.5)	1.4 (0.1)	2.6 (0.1)	3.1 (0.1)	136 (34)	139 (29)	29 (14)	96 (24)	160 (18)	307 (25)	6.3 (0.1)	7.1 (0.2)	8 (0.4)
Storm Bay 1	374 (58)	148 (13)	115 (14)	18.1 (1.3)	33.4 (1.4)	33.6 (2.6)	1.1 (0.1)	2.6 (0.1)	2.8 (0.1)	104 (28)	136 (32)	32 (6)	57 (29)	162 (22)	327 (20)	6.3 (0.3)	7.1 (0.3)	8.8 (0.1)
average	*300 (33)*	*121 (9)*	*105 (8)*	*19 (1)*	*30.1 (1.2)*	*34.1 (1.8)*	*1.2 (0.1)*	*2.6 (0.1)*	*2.9 (0.1)*	*120 (22)*	*138 (21)*	*21 (10)*	*77 (19)*	*161 (14)*	*317 (16)*	*6.3 (0.2)*	*7.1 (0.2)*	*8.4 (0.2)*

	ROV Score			**Dissolved Oxygen** (µmol m^{-2} hr^{-1})			**Dissolved Inorganic Carbon** (µmol m^{-2} hr^{-1})			**Ammonium** (µmol m^{-2} hr^{-1})			**Nitrate** (µmol m^{-2} hr^{-1})		
	Cage	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**	**Cage**	**35/50m**	**Cont**
Macquarie Harbour															
THC	-4.4 (1.1)	6.3 (1.3)	9.9 (0.4)	-3530 (625)	-802 (64)	-418 (45)	8732 (1755)	59 (504)	127 (86)	477 (70)	31 (5)	1 (1)	-78 (10)	-2 (3)	-7 (2)
Gordon	-8 (1.5)	-4.2 (1)	9.9 (0.4)	-1311 (381)	-723 (104)	-418 (45)	2219 (954)	-258 (634)	127 (86)	61 (14)	21 (12)	1 (1)	81 (42)	63 (33)	-7 (2)
Strahan	0.6 (2.2)	5.1 (0.9)	8.1 (0.8)	-1585 (531)	-1326 (318)	-378 (37)	1739 (492)	1535 (242)	544 (164)	282 (85)	29 (8)	10 (2)	-35 (5)	-34 (10)	-16 (2)
Franklin	-7.6 (1.4)	-4.4 (0.7)	5.8 (0.7)	-1447 (246)	-839 (141)	-337 (26)	9490 (1208)	576 (354)	666 (106)	1038 (305)	61 (14)	9 (2)	-48 (12)	-29 (12)	-21 (2)
average	*-5.5 (0.9)*	*0.2 (1)*	*6.2 (0.4)*	*-1910 (265)*	*-913 (95)*	*-437 (27)*	*5545 (931)*	*478 (256)*	*689 (116)*	*528 (126)*	*39 (6)*	*15 (3)*	*-23 (15)*	*-3 (11)*	*-14 (2)*
Southern Channel															
Lippies	*-0.4 (0.9)*	*3.3 (0.5)*	*10.4 (0.4)*	*-3702 (536)*	*-1161 (202)*	*-542 (110)*	*3958 (970)*	*1048 (129)*	*436 (89)*	*174 (32)*	*41 (14)*	*2 (2)*	*-25 (8)*	*-12 (3)*	*4 (1)*
Storm Bay															
Trumpeter	-2.8 (1.1)	1.4 (0.5)	4.4 (0.7)	-5014 (1624)	-1333 (76)	-576 (73)	6681 (1693)	1792 (439)	414 (79)	481 (260)	71 (25)	9 (4)	-43 (24)	-8 (1)	2 (2)
Storm Bay 1	-4.9 (1.7)	2.1 (0.6)	4.5 (0.5)	-3214 (1001)	-1070 (141)	-531 (55)	3817 (1167)	1034 (172)	469 (21)	251 (81)	59 (10)	1 (2)	-14 (6)	-6 (4)	7 (1)
average	*-3.8 (1)*	*1.7 (0.4)*	*4.5 (0.4)*	*-4114 (949)*	*-1202 (86)*	*-553 (44)*	*5249 (1071)*	*1413 (252)*	*442 (40)*	*355 (128)*	*65 (13)*	*5 (3)*	*-27 (12)*	*-7 (2)*	*4 (1)*

Table 5.5-2 Mean abundance (indiv. m^{-2}) and percentage of total faunal abundance for the ten most common species in the three study regions at the cage (0 m), 35/50 m from the cage and the control (CO) sites

Storm Bay

Species	Phylum	0m	%	Species	Phylum	35m	%	Species	Phylum	CO	%
Capitella sp.	Annelida	3436	77.3%	*Capitella sp.*	Annelida	398	22.3%	*Edwardsia sp.*	Cnidaria	211	13.6%
Nassarius nigellus	Mollusca	164	3.7%	*Nassarius nigellus*	Mollusca	140	7.8%	*Euphilomedes sp.*	Crustacea	65	4.2%
Pectinaria antipoda	Annelida	148	3.3%	*Edwardsia sp.*	Cnidaria	122	6.8%	*Aorid sp.1*	Crustacea	51	3.3%
Jassa sp.1	Crustacea	82	1.8%	*Pectinaria antipoda*	Annelida	112	6.3%	*Ampelisca cf. australis*	Crustacea	44	2.8%
Mytilus galloprovincialis	Mollusca	54	1.2%	*Perinereis sp.*	Annelida	58	3.2%	*Isaeid sp.2*	Crustacea	43	2.7%
Perinereis sp.	Annelida	48	1.1%	*Isaeid sp.1*	Crustacea	57	3.2%	*Pectinaria antipoda*	Annelida	36	2.3%
Isaeid sp.1	Crustacea	27	0.6%	*Nemertean sp.1*	Nemetoda	48	2.7%	*Apseudid sp.1*	Crustacea	31	2.0%
Phoxocephalid sp.2	Crustacea	25	0.6%	*Paraonis sp.1*	Annelida	43	2.4%	*Paraprionospio coora*	Annelida	27	1.8%
Edwardsia sp.	Cnidaria	19	0.4%	*Echinocardium cordatum*	Echinodermata	38	2.1%	*Leptocuma obstipa*	Crustacea	27	1.7%
Ianiropsis sp.	Crustacea	17	0.4%	*Mediomastus sp.*	Annelida	37	2.1%	*Onuphid sp.2*	Annelida	26	1.7%

Southern Channel

Species	Phylum	0m	%	Species	Phylum	35m	%	Species	Phylum	CO	%
Capitella sp.	Annelida	13831	88.2%	*Capitella sp.*	Annelida	390	14.2%	*Byblis mildura*	Crustacea	377	19.6%
Mediomastus sp.	Annelida	316	2.0%	*Mediomastus sp.*	Annelida	336	12.2%	*Ampelisca cf. australis*	Crustacea	243	12.6%
Pectinaria cf. dodeka	Annelida	187	1.2%	*Ampelisca cf. australis*	Crustacea	222	8.1%	*Paraprionospio coora*	Annelida	90	4.7%
Corbula gibba	Mollusca	126	0.8%	*Ampharetid sp.1*	Annelida	200	7.3%	*Pratulum thetidis*	Mollusca	88	4.6%
Nebalia sp.1	Crustacea	108	0.7%	*Prionospio kulin*	Annelida	127	4.6%	*Prionospio kulin*	Annelida	72	3.8%
Phyllodocid sp.2	Annelida	103	0.7%	*Paraprionospio coora*	Annelida	124	4.5%	*Scalibregma cf. inflatum*	Annelida	69	3.6%
Paraprionospio coora	Annelida	72	0.5%	*Byblis mildura*	Crustacea	119	4.3%	*Mediomastus sp.*	Annelida	64	3.3%
Theora lubrica	Mollusca	70	0.4%	*Scalibregma cf. inflatum*	Annelida	106	3.9%	*Ampharetid sp.1*	Annelida	50	2.6%
Nassarius nigellus	Mollusca	64	0.4%	*Nemertean sp.1*	Nemetoda	102	3.7%	*Nemertean sp.1*	Nemetoda	48	2.5%
Prionospio kulin	Annelida	54	0.3%	*Ennucula obliqua*	Mollusca	68	2.5%	*Amphiura elandiformis*	Echinodermata	42	2.2%

Macquarie Harbour

Species	Phylum	0m	%	Species	Phylum	50m	%	Species	Phylum	CO	%
Nebalia sp.	Crustacea	230	38.5%	*Schistomeringos loveni*	Annelida	357	37.7%	*Pista australis*	Annelida	42	16.7%
Schistomeringos loveni	Annelida	204	34.1%	*Pista australis*	Annelida	257	27.2%	*Samythella sp.*	Annelida	42	16.7%
Pista australis	Annelida	58	9.6%	*Pseudopolydora cf. paucibranchiata*	Annelida	162	17.1%	*Pseudopolydora cf. paucibranchiata*	Annelida	31	12.2%
Pseudopolydora cf. paucibranchiata	Annelida	26	4.3%	*Samythella sp.*	Annelida	50	5.2%	*Euchone varibilis*	Annelida	29	11.5%
Ophryotrocha shieldsi	Annelida	20	3.4%	*Nebalia sp.*	Crustacea	39	4.2%	*Byblis mildura*	Crustacea	19	7.4%
Capitella sp.	Annelida	18	3.1%	*Capitella sp.*	Annelida	20	2.1%	*Parathyasira resupina*	Mollusca	14	5.4%
Charcotia australiensis	Crustacea	15	2.6%	*Echinocardium cordatum*	Echinodermata	13	1.4%	*Branchiomma sp.*	Annelida	11	4.4%
Echinocardium cordatum	Echinodermata	8	1.3%	*Branchiomma sp.*	Annelida	11	1.2%	*Philomedid sp.*	Crustacea	9	3.4%
Samythella sp.	Annelida	7	1.1%	*Charcotia australiensis*	Crustacea	7	0.7%	*Echinocardium cordatum*	Echinodermata	7	2.9%
Euchaetid sp.	Crustacea	3	0.5%	*Euchaetid sp.*	Crustacea	4	0.5%	*Amaena cf. trilobata*	Annelida	5	2.1%

Table 5.5-3 Comparison of the sediment particle size distribution at each of the study regions at the cage (0 m), 35/50 m from cage and the control (CO) sites.

Storm Bay	**pebble 4mm**	**granule 2mm**	**very coarse sand 1mm**	**coarse sand 0.5mm**	**medium sand 0.25mm**	**fine sand 0.125mm**	**very fine sand 0.063mm**	**silt <.063mm**
0m	-136.0%	30.0%	-38.0%	6.0%	-20.0%	10.0%	0.0%	3.7%
35m	-60.0%	14.0%	-22.0%	9.0%	-5.0%	6.0%	0.0%	4.9%
CO	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Southern Channel								
0m	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
35m	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
CO	96.0%	24.0%	160.0%	18.0%	307.0%	25.0%	0.0%	6.3%
Macquarie Harbour								
0m	77.0%	19.0%	161.0%	14.0%	317.0%	16.0%	0.0%	6.3%
50m	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
CO	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%

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6 Reef Interactions

Assessing interactions between temperate reef ecosystems and salmon farming in south-east Tasmania: Development of fit-for-purpose monitoring tools

6.1 General introduction

The salmon industry in Tasmania is currently in a phase of expansion, with plans to increase to a $1 billion a year industry by 2030 (DPIPWE 2017). To meet this target, the industry is considering a range of options including alternate farming approaches and expansion into new areas. Maintaining strong environmental performance under new production scenarios is a key priority for both government and industry. As the industry increases its presence in Tasmanian coastal regions, both in terms of location and total production volume, increasing importance has been placed on better understanding the broadscale interactions of farming with potentially sensitive habitats, including Tasmania's iconic rocky reefs. Intensification of aquaculture activities in the new farming areas in south-east Tasmania (southern D'Entrecasteaux Channel and northern Bruny Island) has also led to requests from the abalone and rock lobster industries and commercial and recreational fishing communities to review the risks of adverse interactions with these important industry sectors.

In response to these concerns, this project sought to specifically inform understanding of the potential for interactions between local reef systems and salmon farming, with two key objectives: 1) to examine the main environmental risk factors for temperate reefs in southern Tasmania from salmon farming, and 2) to develop risk appropriate monitoring and assessment approaches for reefs in these environments.

To meet these objectives, this project comprised six main components:

1) **A global literature review on the effects of organic enrichment on temperate rocky reefs (with a particular focus on aquaculture where possible):** Here the aim was to address major pathways for interaction between temperate reef ecosystems and organic enrichment sources (Section 6.2).

2) **A literature review on the methods for assessing change in temperate rocky reefs**: The aim was to examine existing monitoring and assessment approaches used to assess environmental impacts on reef systems worldwide, and to provide the basis for selecting the approaches to be tested within this project (Section 6.3).

3) **A baseline biodiversity assessment of rocky reef ecosystems in the southern Channel and north Bruny Island**: This provided a detailed census of all flora and fauna found on rocky reefs in the two proposed salmon expansion areas and a means to directly evaluate salmon farming interaction risks within a local context (Section 6.4).

4) **Development of a novel Rapid Visual Assessment technique for the assessment of organic enrichment on temperate reef ecosystems**: This component sought to develop a practical and robust technique to directly assess functional change that could be attributed to organic enrichment in temperate reef ecosystems and used for ongoing management of these systems (Section 6.5). The method was developed based on the findings of outputs 1-3**.**

5) **Assessment of towed video and photo quadrats as a method for assessing change in temperate reef ecosystems**: This sought to provide a cost-effective and robust, rapid assessment alternate to diver-based surveys. Based on the findings of outputs 1-4 a sub-set of remote techniques (towed video and photo-quadrats) were evaluated (Section 6.6).

6) **Assessing the capacity of targeted indicators for monitoring health of temperate reef ecosystems**: Here the robustness of potential reef health indicators (abalone plates, epiphyte plates, cryptic invertebrate assemblages, and sediment traps) was examined for suitability for incorporation into ongoing monitoring programs (Section 6.7).

The general discussion synthesises the findings of each component into a series of recommendations regarding risk appropriate assessment and monitoring of potential interactions between salmon farming and Tasmania’s temperate rocky reefs.

6.2 Global significance of temperate reefs: An ecosystem under pressure

Temperate reef ecosystems have exceptionally high importance, providing numerous ecological, recreational and commercial services on a global scale (Teagle et al. 2017; Smale et al. 2013; Bennett et al. 2016). Temperate reefs are classified as hard-bottom marine ecosystems existing between the tropics and the poles, with a high diversity of form and function (Bennett et al. 2016; Steneck et al. 2002). Forests of macroalgae tend to dominate these ecosystems, providing both habitat and the foundation for a wide range of ecosystem services (Teagle et al. 2017; Krumhansl et al. 2016). As these forests are dependent on light for productivity, they are prevalent in shallow or coastal waters. The nearshore coastal zone is also where the majority of human activity is located, and subsequently, temperate reefs are particularly vulnerable to impacts from anthropogenic activities (Crain et al. 2009; Halpern et al. 2008; Vitousek et al. 1997).

Despite high levels of human interactions, temperate reef ecosystems are some of the most productive and diverse habitats on earth (Teagle et al. 2017). They possess disproportionately high biodiversity compared to other habitats, along with high rates of endemism (Oh et al. 2015; Kerswell 2006). Additionally, temperate reef ecosystems provide a number of critical ecosystem services, underpinning many fisheries worldwide (Bertocci et al. 2015) and providing substantial carbon sequestration, nutrient cycling (Teagle et al. 2017), tourism and recreational services (Rees et al. 2015). Despite the high ecological and functional significance of these environments, they are often seen as the “lesser cousin” of tropical reef ecosystems, with limited public perception of the broader value of these ecosystems and consequently smaller research effort (Figure 6.2-1; Coleman & Wernberg 2017; Bennett et al. 2016). This low public perception may increase the vulnerability of temperate reef ecosystems to anthropogenic impact as management and protection strategies are not as demanding. The tendency to place a greater value on tropical systems also means we have more gaps in our knowledge regarding how temperate ecosystems respond to increasing anthropogenic pressure, both at the local and global spatial scales. Given the broad diversity and high degree of endemism that exist in temperate reef ecosystems, responses to anthropogenic pressures are likely to vary significantly at local scales with a potential for ecosystem consequences that are not easy to reverse.

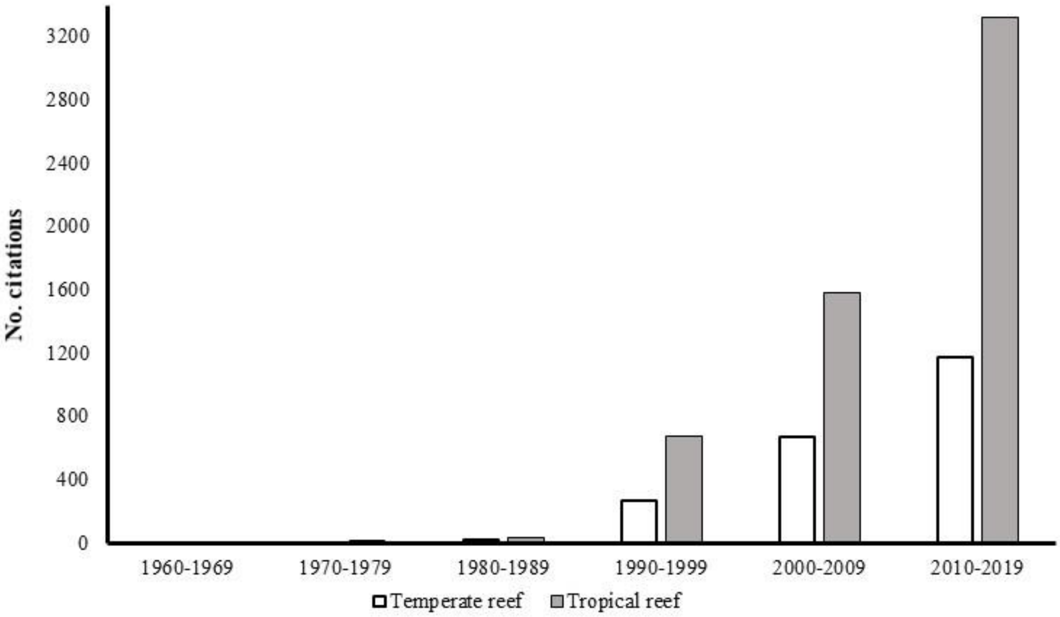

Figure 6.2-1 Comparative research effort on temperate vs tropical reef ecosystems characterised through the number of citations of "temperate reef" vs "tropical reef" from a Web of Science search on these terms.

Like many marine ecosystems, temperate reefs are under pressure from a variety of anthropogenic stressors. These stressors can be short or long-term (i.e. episodic or sustained) and local to global in nature. Climate change is a good example of a global issue where impacts on temperate reef systems have been clearly demonstrated. For example, increases in sea temperature linked to global climate change have been shown to result in range-shifts of keystone temperate reef species, which can in turn alter ecosystem function (e.g. Robinson et al. 2015; Johnson et al. 2011). There are numerous examples of resource exploitation affecting both the biodiversity and functional ecology of specific reef systems (Teagle et al. 2017), such as proliferation of prey species or primary producers through the loss of predators (Ling et al. 2009; Edgar et al. 2009). There are also many examples of localised impacts, such as point source discharges from industry or human settlement (Stuart-Smith et al. 2015; Campbell et al. 2014; Connell et al. 2008; Airoldi & Beck 2007). Each of these stressors can impact temperate reef ecosystems in different ways, but all have the capacity to cause phase-shifts in ecosystem function. In addition, stressors can interact, and where effects are synergistic the overall outcome can be greater than the sum of the individual effects (Strain et al. 2014; Crain et al. 2008; Steneck et al. 2002).

While climate change is a significant emerging challenge, organic enrichment from human activity has been identified as a threat to ecosystem function in many areas (Russel et al. 2009; Vitousek et al. 1997). Climate change has been shown to heighten the impacts of organic enrichment on temperate reef ecosystems (Falkenberg et al. 2014; Strain et al. 2014), with nutrient enrichment compounding the effects of factors such as enhanced CO_2 concentrations and temperature increases (Russell et al. 2009). Unlike global stressors, impacts from organic enrichment are often easier to address at the local level. By managing organic enrichment locally, there is potential to improve the resilience of reef ecosystems not just at the local level, but also with respect to those global stressors more commonly associated with climate change (Russell et al. 2009). However, for such management to be effective, there needs to be a detailed understanding of the ways in which organic enrichment can affect temperate reef ecosystems, and how the local biotic and abiotic factors might interact to enhance or mitigate these effects. Understanding the potential effects of organic enrichment on temperate reef ecosystems is critical both to establishing the sensitivity or resilience of systems, and to developing appropriate management strategies.

6.2.1 Processes of organic enrichment in temperate reef ecosystems: A "multi-use" challenge

The traditional view of organic enrichment in the marine environment is that inputs are generally point source, and given the dilution capacity of the oceans, unlikely to cause any negative broad-scale response. However, with approximately half the global population living within 60 km of the coast (UNEP 2016), and urban population growth continuing to climb, there is potential for multiple point-sources of coastal pollution to overlap each other resulting in compounded effects over a broader area spatially. Pollution associated with coastal urbanisation has long been regarded as one of the most consistent and widespread pressures to temperate reef ecosystems (Halpern et al. 2007; Krause-Jensen et al. 2008). Increasingly, industries such as open-cage aquaculture are providing additional and significant levels of organic inputs into coastal environments (Wang et al. 2012; Carrol et al. 2003; Brooks et al. 2002). In many areas, aquaculture is now operating on a scale far greater than any previous point-source organic enrichment (Taranga et al. 2015; Trujillo et al. 2012), with farmed species often relying on nutrient-rich external food sources (Tacon & Metian 2008). While aquaculture may be an important industry to provide increasing global protein needs, its long-term growth and expansion must consider the environmental sustainability in multi-use systems, such as temperate reef environments. As organic inputs from coastal urbanisation and coastal aquaculture are likely to grow into the future, pressures on temperate reef ecosystems will be exacerbated. Consequently, it is essential to develop management strategies at the ecosystem level. The management of inputs independently runs the risk of missing the collective impacts.

Understanding the various pathways where organic enrichment can enter coastal waters is critical for management of multi-use systems. Organic enrichment of temperate reef ecosystems can occur through either dissolved or particulate inputs which are seldom independent (Figure 6.2-2). Increases in dissolved nutrients like nitrogen will tend to boost productivity at the base of the food web. In contrast, particulate organic enrichment sources associated with inputs such as sewage outfalls or aquaculture waste can directly increase food resources to opportunistic invertebrates and fish, potentially causing top-down effects (White et al. 2018; Dempster et al. 2009). Increases in particulate loads onto reef ecosystems can also influence lower trophic order processes, potentially shading encrusting flora and fauna or smothering filter-feeders (Krause-Jensen et al. 2008, 2009). Dissolved and particulate enrichment pathways are not independent, as elevated dissolved nutrients can contribute to increased sedimentation of organic material by boosting water column productivity. This can in turn influence light penetration with a negative feedback to the primary productivity of reef ecosystems (Guinder et al. 2009; Diehl 2002). Thus, both dissolved and particulate inputs have the capacity to drive change on temperate reef ecosystems, albeit through different but interlinked pathways. Tools such as biogeochemical models to inform management need to understand both these pathways and their synergisms.

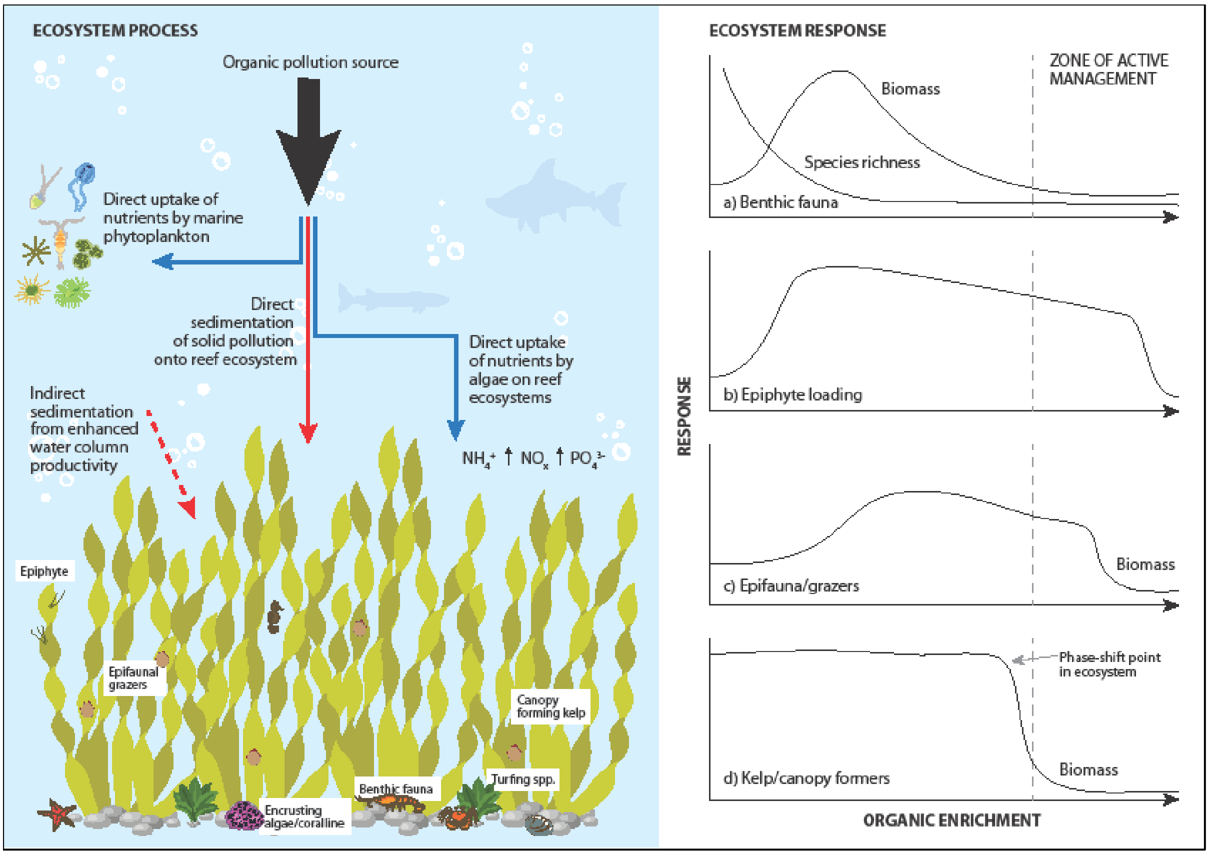

Figure 6.2-2 Schematic outlining both the process and response to organic enrichment in temperate reef ecosystems. The process pane outlines the direct and indirect pathways of organic enrichment of temperate reef ecosystems. The response pane highlights the complexity of response in a multi-level ecosystem, with A) an adaptation of the standard Pearson-Rosenberg (1978) model for organic enrichment for benthic infauna; and B-D representing the hypothetical response of different functional groups in a three-dimensional/multi-trophic temperate reef system.

In general, temperate coastal ecosystems are relatively resilient to low levels of organic enrichment, particularly where the mode of delivery simulates natural processes, such as “pulses” of nutrients mimicking event-based inputs such as high rainfall. While there may be a short-term acute response to organic material after an event such as this, in the absence of ongoing inputs, the ecosystem tends to revert to an original state (Bender et al. 1984; Keough & Quinn 1998; Gillanders & Kingsford 2002). However, organic enrichment associated with anthropogenic activities is more likely to be presented as a “press” input, where organic enrichment levels remain elevated over time (Gillanders & Kingsford 2002). At relatively low levels such inputs can be assimilated by the marine food web, however, if sustained long-term, then there is the potential for phase shifts in ecosystem function. For example, long-term eutrophication of the Baltic Sea has caused major shifts in phytoplankton communities resulting in what was termed a “silent spring in the sea" (Rosenberg et al. 1988), whilst long-term eutrophication driven hypoxia, overfishing and invasive species in combination are believed to have been responsible for drastic reductions in demersal fisheries in the Black Sea (Diaz & Rosenberg 2008). The latter example shows the local environment (e.g., ambient nutrient concentrations, exposure, current flow, seasonal variation, and presence of other stressors) influences ecosystem resilience and the potential for temperate reef systems to withstand such impacts. Thus, it is important to understand the process of organic enrichment at a local scale, to characterise the challenges for management.

6.2.2 The common ground: Global responses to organic enrichment

For management to be most effective it requires both reliable indicators that a system is under stress and clear management actions. There have been many studies examining the impacts of organic enrichment on temperate reef ecosystems, and as a result there are several common globally recognised responses (Table 6.2-1). Of all the potential enrichment impacts on temperate reefs, bottom-up effects on algae are the most widely reported and understood (Table 6.2-1). Organic enrichment, associated with urbanisation, has been linked to region-scale losses of canopy forming perennial brown macroalgae, and its replacement with fast growing turfing or filamentous species (Benedetti-Cecchi et al. 2001; Eriksson et al. 2002; Connell et al. 2008). This phase-shift occurs after an initial disturbance, such as a storm event, that removes large patches of the canopy forming algae. Combined with enhanced nutrient conditions to support persistent growth of opportunistic algae, regrowth of the canopy is inhibited (Alestra & Schiel 2015; Connell et al. 2008; Gorgula & Connell 2004). Large scale loss of canopy forming macroalgae is not always associated with areas of organic enrichment, and forests of macroalgae have been shown to be quite resilient to organic enrichment in isolation (e.g., Bokn et al. 2003). In southern Tasmania, Oh et al. (2015) observed consistent nutrient-associated increases in epiphytic and ephemeral algae in proximity to salmon farms (Table 6.2-1), but the foundation species that formed canopies were not affected. While the presence of large amounts of epiphytic and ephemeral algae indicates the initial system response to nutrient enrichment, how the ecosystem would respond should a large-scale disturbance occur to remove the canopy is unknown. Clearly, there is still a need for further investigation into the factors that may enhance either the vulnerability or resilience of temperate reef ecosystems.

Table 6.2-1 Common responses to organic enrichment (dissolved nutrient and sedimentation effects) by algae, invertebrates and fish in temperate reef ecosystems.

		Response	**Location (reference)**
Dissolved nutrient enrichment	**Algae**	Increase growth of turfing algae	South Australia (Connell et al. 2008; Falkenberg et al. 2012)
		Increased density of epiphytic cover on macroalgae	SE Australia (Oh et al. 2015)
		Increased density of filamentous green algae	Southern Norway (Bokn et al. 2003) SE Australia (Oh et al. 2015)
		Loss of perennial/canopy forming species	Baltic Sea (Worm et al. 1999), South Australia (Gorman et al. 2009)
		Loss of species diversity	Baltic Sea (Worm et al. 1999), SW Atlantic (Scherner et al. 2013)
		Increased biomass of rapid growth macroalgae	Japan (*Saccharina*, Agatsuma et al. 2014), SE Australia (*Undaria*, Carnell & Keough, 2014), California (*Macrocystis*, Hernandez-Carmona et al. 2001)
	Fish & mobile invertebrates	Increases in herbivorous species	Baltic Sea (Hillebrand et al. 2000), NW Atlantic (Lotze et al. 2001), South Australia (Russel & Connell 2005, Falkenberg et al. 2014)
Sedimentation	**Algae**	Inhibition of growth or recruitment of sensitive species	South Australia (Irving and Connell 2002, Connell et al. 2008), Mediterranean (Claudet & Fraschetti, 2010), Chile (Muth et al. 2017)
		Mortality through smothering of sensitive species	Western Mediterranean (Airoldi et al. 1995, Airoldi & Cinelli 1997), South Australia (Irving and Connell, 2002).
		Trapping of sediment by turf species	Western Mediterranean (Airoldi et al. 1995, 1996)
	Fish & mobile invertebrates	Increases in opportunistic species	NE Atlantic (Echavarri-Erasun et al. 2007), Mediterranean (Fernandez-Jover et al. 2008), Western Norway (White et al. 2018)
		Community changes in filter feeders	NE Atlantic (Echavarri-Erasun et al. 2007), Australia (Schonberg, 2016)
		Community changes in fish species	SE Australia (Smith et al. 1999), Mediterranean (Guidetti et al. 2003), Norway (Dempster et al. 2009)
Combined effects	**Algae**	Loss of perennial/canopy forming macroalgae	NW Mediterranean (Benedetti-Cecchi et al. 2001; Mancuso et al. 2018), Sweden (Eriksson et al. 2002), South Australia (Connell et al. 2008), SE Australia (Coleman et al. 2008), SW Atlantic (Scherner et al. 2013)
		Increases in turfing algae	NW Mediterranean (Benedetti-Cecchi et al. 2001), Sweden (Eriksson et al. 2002), South Australia (Gorgula & Connell, 2004; Connell et al. 2008)

	Fish & mobile invertebrates	Increases in abundance of smaller, faster growing species	SE Australia (Stuart-Smith et al. 2015; Ling et al. 2018)
		Reduced biomass	SE Australia (Stuart-Smith et al. 2015)
		Reduced species richness	SE Australia (Stuart-Smith et al. 2015), Global (McKinley & Johnston, 2010)

Foundation species, such as canopy forming perennial kelps, inherently stabilise ecosystems, potentially buffering the effects of many impacts, including nutrient enrichment and global climate change (Falkenberg et al. 2012, 2015). However, not all canopy forming species are equal and variation in the canopy can have flow on effects with respect to ecosystem sensitivity/ resilience. Carnell & Keough (2014) found nutrient enrichment allowed the proliferation of an introduced canopy species. Whilst the overall canopy remained intact, the species composition of the canopy was altered, with potential flow on effects to ecosystem function. Likewise, Irving et al. (2004) found species composition within the canopy plays a large role in the structuring of benthic communities, with the effects of organic enrichment likely to vary between mono-specific and multi-species canopies. The ambient nutrient concentration can also affect outcomes, with high anthropogenic inputs into oligotrophic systems likely to have the most extreme results (Gorman et al. 2009; Connell & Irving 2008; Russell et al. 2005). Gorman et al. (2009) found that the extent of turf establishment on South Australian coasts was largely dependent on the inherent nutrient availability, with locations where the differential between input and existing nutrient concentrations was high experienced the most severe effects. Timing of nutrient inputs can also influence outcomes, particularly where anthropogenic inputs (either as a result of quantum or timing) override natural seasonal variation in nutrient limitations. Where this occurs, ecosystems may be vulnerable to organic enrichment effects, with turfing algal species able to proliferate at the expense of canopy forming algae as turfs are better able to exploit the constant supply of nutrients that feed their more rapid growth patterns (Connell et al. 2008; Gorgula & Connell 2004; Benedetti-Cecchi et al. 2001). Further study is needed to determine how resilience to organic enrichment may be influenced by factors such as spatial and temporal availability of nutrients, along with species richness of the canopy, as this would underpin development of regionally tailored management strategies.

While organic enrichment can elicit bottom-up effects on algal communities, these may be wholly or partially mitigated by the related response of the primary consumers (Table 6.2-1). Herbivores can provide top-down control for turfing or filamentous algae stimulated by organic enrichment (Worm et al. 1999). This process will, once again, likely depend on the inherent natural characteristics of the region, particularly ambient nutrient concentrations. In naturally eutrophic systems nutrient enrichment (bottom-up) effects are likely to dominate however, grazer (top-down) effects may be much more pronounced in oligotrophic conditions (Burkepile & Hay 2006; Lotze et al. 2001). Trophic interactions are complex, however, with factors such as the life-history strategy, functional group of the dominant species, and timing (season) likely to impact the result (Alestra & Schiel 2015; Burkepile & Hay 2006; Lotze et al. 2001). Furthermore, not all species exert equal influence on ecosystems. Keystone species such as herbivorous sea urchins are renowned for overgrazing on kelp forests, with wide-reaching top-down effects which may act synergistically with nutrient enrichment to cause a functional shift in reef ecology (Piazzi & Ceccherelli 2017). In the Western Mediterranean Sea, overgrazing by sea urchins caused the initial disturbance which removed the kelp communities, but nutrient enrichment created a positive feedback loop that promoted the growth of turfing algae and limpets which impeded subsequent kelp bed recovery, even once urchin numbers had subsided (Piazzi et al. 2016). In this instance, it would have required a return to oligotrophic conditions to allow the kelp beds to recover (Piazzi & Ceccherelli 2017). This clearly shows that at a regional level, organic enrichment works in combination with local conditions (i.e., species structure and diversity and environmental conditions) to influence the scale of impact.

Interactions between fish and organic enrichment are even more complex. Organic enrichment can affect fish assemblages either directly, through consumption of the solid fraction (Fernandez-Jover et al. 2008), or indirectly, through interactions with other species influencing prey items. In areas affected by urban sources of enrichment, increases in the abundance of smaller, faster growing species, reduced biomass and reduced fish species richness have been observed (Ling et al. 2018; Stuart-Smith et al. 2015). However, where the organic enrichment also provides an additional food source, as is the case with aquaculture (i.e. excess feed and faecal material), the biomass of opportunistic fish species using this resource as a trophic subsidy may increase significantly (Fernandez-Jover et al. 2011). Clearly interactions with organic enrichment are complex at this trophic level, and there is potential for cascading effects that may affect the commercial and recreational values of temperate reef ecosystems with implication for regional economies. Higher trophic interactions and the potential for adverse impacts are likely to be dependent on the intrinsic environmental conditions. Consequently, a firm knowledge of these factors is vital to the understanding of the ecosystem response.

6.2.3 Mapping process into response: The inherent complexity of reef ecosystems

Reef ecosystems are complex, varying widely according to local or regional environmental conditions. Although the expected response to organic enrichment of a particular species or taxonomic group may be understood (Table 6.2-1), it is still difficult to predict the broader ecosystem response. An understanding of both process and response is critical for sustainable management of temperate reef ecosystems (Figure 6.2-2). Conceptually, the process of organic enrichment can be mapped, even in complex multi-use ecosystems, where there is good understanding of anthropogenic inputs. It is understanding the response that presents the more significant challenge.

The environmental and functional responses of the ecosystem to organic enrichment are relatively well understood for soft sediments (Figure 6.2-2; Keeley et al. 2012; Kalantzi & Karakassis 2006; Pearson & Rosenberg 1978), and whilst some of that understanding may translate across to temperate reef ecosystems, the multi-dimensional structure of reefs represents quite a fundamental difference. In this case, organic enrichment may have different interactions and impacts at each functional level within the reef ecosystem, as the prevailing environmental conditions (e.g., nutrient concentration, temperature, and disturbance) may elicit quite different responses for each level (Figure 6.2-2). For example, the temporal response profile for epiphytic and/or turfing algal species (i.e., fast growing and opportunistic species) would likely follow the expected “Pearson-Rosenberg” response to organic enrichment (Figure 6.2-2; Benedetti-Cecchi et al. 2001; Eriksson et al. 2002; Connell et al. 2008). Initially, these species would proliferate, potentially replacing the canopy forming species as the dominant primary producers in the reef ecosystem. These opportunistic algal species are relatively robust and so can tolerate high levels of eutrophication. They may be controlled in some measure by epifauna/herbivores (Figure 6.2-2; Burkepile & Hay 2006; Lotze et al. 2001; Worm et al. 1999), which would also adapt to take advantage of this new resource, peaking in biomass further along the enrichment gradient. However, epifauna/herbivores are likely to be more susceptible to eutrophication and so would not be able to survive very high nutrient levels. The reef response model is not wholly inconsistent with that proposed for soft sediments, but one key difference is the presence of the kelp canopy, which stabilises the ecosystem and induces inherent resilience by providing physical, chemical and biological buffering (Connell & Ghedini 2015). However, somewhere along the organic enrichment gradient, there will be a phase-shift in the ecosystem, where there is rapid kelp loss (e.g., Gorman et al. 2009), either due to secondary enrichment effects (i.e. smothering of the macroalgae with epiphytic flora, fauna or sediments), or as a result of physical disturbance.

These processes interact simultaneously. The point at which management actions are most commonly initiated, the "zone of active management," is often after the critical phase-shift point for ecosystem function, as such catastrophic loss of canopy forming algae has already occurred (Figure 6.2-2). While management strategies such as the replanting of canopy algae or the removal of turfs have had some success in re-establishing kelp communities in specific circumstances (e.g., Campbell et al. 2014; Gorman & Connell 2009), it would be preferable to act before such large-scale kelp loss occurs. Consequently, identifying clearer indicators of when the system is under stress from organic enrichment is a key challenge.

6.2.4 Challenge for the future: Incorporating science into management

There is no "one size fits all" rule for management of temperate reef ecosystems. Understanding the regional ecosystem structure and function, and prevailing environment conditions is crucial for predicting the potential risk of organic enrichment, and for determining the likely response, vulnerability and resilience of temperate reef systems. Existing management of these systems tends to be reactive, where the impacts and intervention strategies are not investigated until a clear and current problem (e.g., widespread kelp loss) has occurred.

Large-scale kelp loss remains one of the most ubiquitous responses of temperate reef ecosystems to organic enrichment. It is very hard to manage once it has occurred. Science is needed to support proactive management. Whilst it requires a large-scale disturbance (e.g., severe weather, urchin grazing) to facilitate a phase-shift from kelp to turfing algae in isolation (Mancuso et al. 2017; Connell et al. 2008), once such a disturbance has occurred nutrient enrichment can entrench an alternate stable state (Connell & Ghedini 2015), with "press" inputs of nutrients sustaining the growth of turfing communities over slower growing canopy forming species. As it is very difficult to reverse this process and return an ecosystem back to its original state, it would be better to avoid this phase-shift in the first place (Gorman & Connell 2009). Sustained growth of epiphytic and ephemeral species is potentially a good early indication the ecosystem may be subject to pressure from organic enrichment (Oh et al. 2015). However, it is still not known at what point the presence of these species shifts from simply being a useful proliferation to soak up excess nutrient, to being indicative of an ecosystem under stress and therefore vulnerable.

As tipping points are likely to be regionally specific, there is growing need for an assessment approach that can effectively quantify impacts of organic enrichment along with subsequent trophic consequences on a regional basis. Effective management will require good local knowledge of the environmental conditions, inherent biodiversity, and functionality of the ecosystem. Characterising regional biodiversity provides valuable information on ecosystem structure and trophic interactions. Ongoing monitoring, targeted on the impacts of organic enrichment, will provide the understanding of the functional ecology and impact responses to establish ecosystem tipping points. Reef functionality and biodiversity are interlinked and an understanding of both is necessary for good management. Baseline data, being data collected prior to the introduction of a potential source of enrichment, is also essential. The absence of baseline data in many highly urbanised/ impacted areas, such as the Mediterranean, northern Europe, or even southern Australia is a hindrance to effective management of reef systems (Claudet & Fraschetti 2010; Connell et al. 2008; Eriksson et al. 2002). Without baseline data, identifying cause and effect can be extremely difficult, rendering it impossible to know how to adjust management strategies accordingly (Duarte et al. 2009).

There is a real opportunity to “think global but act local” (Strain et al. 2014; Russell et al. 2009) when translating ecological knowledge into sound management practice for organic enrichment. Given the increasing pressure on temperate reef ecosystems, and the high value, but general under-appreciation of the ecosystem services these reefs provide, the challenge for the future will be determining how to respond proactively to ensure their long-term sustainability. Mitigating the effects of organic enrichment on a regional level has the potential to make temperate reef ecosystems more resilient to a broader suite of “global” environmental pressures. To that end, local knowledge of ecosystem processes and impact responses is key to successful management, particularly in regions where there are multiple anthropogenic enrichment sources.

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6.3 Methods for assessing change in temperate reef ecosystems

6.3.1 Introduction

Approaches for assessment of anthropogenic impact on temperate marine reefs have evolved markedly over the last decades, informed by experience and influenced by technological advances. There are now a number of different scientific approaches that can be used to assess change in temperate reef ecosystems (Van Rein et al. 2009). However, as temperate reefs are multi-dimensional, multi-trophic systems with a high degree of inherent variability, unless change is catastrophic, the ability to detect and identify anthropogenic driven change is challenging. Consequently, it is important to be very clear about the aims and objectives in undertaking assessments, and to clearly define the type and magnitude of change to be detected. Given the complexity of temperate reef ecosystems, not all anthropogenic pressures will elicit the same response (Shahidul Islam & Tanaka 2004; Arevalo et al. 2007; Fowles et al. 2018; Fowles et al. 2018), and a study examining a once-off disaster such as an oil spill or an extreme weather event will clearly require a different design to an ongoing monitoring program which seeks to understand the potential impacts of industry or urbanisation over time (Juanes et al. 2007; Castege et al. 2014; Valentine et al. 2016). A thorough understanding of methods detect change is therefore necessary to facilitate sustainable management of temperate marine reefs and the resources they support into the future.

This chapter will 1) explore different rocky reef survey designs; 2) describe the different metrics used to assess rocky reef environments; and 3) compare sampling methods. In all cases, the benefits and limitations of the different approaches in terms of their ability to detect human-induced change in temperate rocky reefs over time, but most especially their ability to identify impacts **before** ecosystem-level phase shifts occur, will be considered.

6.3.2 Survey techniques for rocky reef environments

Quantitative assessments of temperate rocky reef ecosystems are generally undertaken using transects and/or quadrats (Kushner et al. 1994; Edgar & Barrett 1997; Edgar & Barrett 1999; Parravicini et al. 2010; Zahn et al. 2016). While transects are necessary for assessing larger mobile species (e.g. fish and macro invertebrates), quadrats are better suited to quantify the coverage of non-mobile organisms such as sessile invertebrates and algae (Edgar & Barrett 1997; Van Rein et al. 2009). The standardised quantitative approach outlined by Edgar & Barrett (1997) has been used in a number of subsequent temperate rocky reef studies (Edgar & Barrett 1997; Edgar et al. 1997; Stuart-Smith et al. 2008). However, this adoption is not universal, with many studies designing surveys to accommodate specific research questions and the characteristics of the ecosystem (Table 6.3-1).

Table 6.3-1 Approaches to Rocky Reef Health Monitoring - Summary

Method														Habitat type	Paper
Survey Technique				Collection method					Parameter				Survey depth design		
Transects	Quadrats	Photo mosaic	Video footage	In-situ Scuba	Scuba + camera	AUV	ROV	Drop camera [1]	Fish	Invertebrates	Macroalgae	Other			
✓	✗	✗	✗	✓	✗	✗	✗	✗	✓	✓	✓	✗	Contour	TRR	Tenera Environmental. (2006)
✓	✓	✗	✗	✓	✗	✗	✗	✗	✓	✓	✓	✗	Contour	TRR	Edgar and Barrett (1997)*; Edgar and Barrett (1999)*; Stuart-Smith, Barrett et al. (2008)*; Edgar, Moverley et al. (1997) *
✓	✓	✗	✗	✓	✓[r]	✗	✗	✗	✓	✓	✓	✗	Contour	TRR	Sala, Ballesteros et al. (2012); Valentine, Jensen et al. (2016)
✓	✓	✗	✗	✓	✓[r]	✗	✗	✗	✓	✓	✓	✗	Variable	TRR	Kushner, Lerma et al. (1994)
✓	✗	✗	✗	✓	✗	✗	✗	✗	✓	✓	✗	✗	Contour	TRR & SRR	Edgar and Stuart-Smith (2009)
✓	✗	✗	✗	✓	✗	✗	✗	✗	✗	✓	✗	✗	Contour	TRR	Zahn, Claisse et al. (2016)
✓	✓	✗	✗	✓	✗	✗	✗	✗	✗	✓	✓	✗	Contour	TRR	Barrett, Buxton et al. (2009)
✗	✓	✗	✗	✓	✗	✗	✗	✗	✗	✓	✓	✗	Con. + Var.	TRR	Parravicini, Micheli et al. (2010)
✓[q]	✓	✗	✗	✓	✗	✗	✗	✗	✗	✗	✓	✗	Contour	TRR	Crawford, Thompson et al. (2006)
✓	✗	✗	✓	✓	✓	✗	✗	✓	✓	✗	✗	✗	Unknown	TRR	Watson, Harvey et al. (2005)
✓	✗	✗	✓	✓	✗	✗	✗	✓	✓	✗	✗	✗	Variable	TRR	Colton and Swearer (2010)
✓	✗	✗	✓	✓	✗	✗	✗	✓	✓	✗	✗	✗	Unknown	TR	Willis and Babcock (2000)
✗	✗	✗	✓	✗	✗	✗	✗	✓	✓	✗	✗	✗	Contour	TRR, SB	Zintzen, Anderson et al. (2012)
✓	✗	✗	✓	✗	✗	✗	✗	✓	✓	✗	✗	✗	Variable	TRR	Logan, Young et al. (2017)
✗	✗	✗	✓	✗	✗	✗	✗	✓	✓	✗	✗	✗	Variable	TRR, SRR	Harasti, Malcolm et al. (2015), Walsh, Barrett et al. (2016); Harasti, Davis et al. (2018)
✗	✗	✓	✗	✗	✗	✓	✗	✗	✗	✗	✗	✓	Variable	TRR	Williams, Pizarro et al. (2010)
✓	✗	✓	✗	✗	✗	✓	✗	✓	✗	✓	✓	✓	Variable	TRR	Barrett, Seiler et al. (2010)

Method														Habitat type	Paper
Survey Technique				Collection method					Parameter				Survey depth design		
Transects	Quadrats	Photo mosaic	Video footage	In-situ Scuba	Scuba + camera	AUV	ROV	Drop camera [1]	Fish	Invertebrates	Macroalgae	Other			
×	✓	✓	×	×	×	✓	×	×	×	✓	✓	×	Variable	TRR, CR	Smale, Kendrick et al. (2012)
✓	×	×	×	×	×	×	×	✓	✓	✓	✓	×	Variable	TRR, SB	Nichol, Anderson et al. (2009)
×	✓	×	×	×	✓	×	×	×	×	✓	✓	×	Variable	TRR	Garrabou, Riera et al. (1998); Dunmore, Keeley et al. (2015); Ling (2008)
×	✓	×	×	×	✓	×	×	×	×	✓	✓	×	Contour	TRR	Glasby (1999); Balata, Piazzi et al. (2007)
✓[q]	✓	×	×	×	✓	×	×	×	×	✓	✓	×	Contour	TRR	Oh, Edgar et al. (2015)
✓	✓	✓	×	×	✓	×	×	×	×	×	✓	×	Variable	TRR	Guinda, Gracia et al. (2014)

* These papers follow the Edgar Barrett (1997) methodology closely.

✓[r] These surveys included video for reference only but did not rely on recorded video or photographs for analysis.

1 Includes stationary and towed cameras.

✓[q] Transects used purely for placement of quadrats/photo-quadrats

TRR Temperate Rocky Reef; SRR: Subtropical Rocky Reef, TR: Temperate Reef; SB: Soft Bottom; CR: Coral Reef

6.3.2.1 Survey design

Defining appropriate and standardized survey methods for temperate rocky reef surveys is a challenge. Survey design can vary substantially from survey to survey depending on research questions, restrictions imposed by reef geography and biology, and/or resource limitations (Table 6.3-1, Table 6.3-2, Table 6.3-3, and Table 6.3-4).

Transects are effective tools for collecting data on temperate rocky reefs. How they are applied is often largely dependent on which species or functional element of the rocky reef ecosystem is being characterised (Kushner et al. 1994; Edgar & Barrett 1997; Edgar et al. 1997). Broader and longer transects are commonly used to record larger and mobile functional groups. These groups often exhibit a relatively low and heterogeneous population density (e.g., predatory fish) and may therefore be missed when using techniques which cover less area (Murray 2001; Parravicini et al. 2009). In contrast, more constrained transects or quadrats are most commonly used to collect data on sessile and slow moving organisms, cryptic fish, benthic cover (e.g. algae) and other aspects of the benthos (e.g., substrate type) which have a higher population density, spatial complexity and/ or potentially cryptic nature (Edgar et al. 1997; Tenera Environmental. 2006; Van Rein et al. 2009; Guinda et al. 2014). Smaller transects or quadrats are inherently limited in their ability to characterise mobile invertebrates and fish (Garrabou et al. 1998; Parravicini et al. 2009; Van Rein et al. 2009; Oh et al. 2015).

Transects characterise a broad survey area, usually delineated by a line of a set length (e.g. Stuart-Smith et al. 2008; Barrett et al. 2009; Edgar & Stuart-Smith 2009; Sala et al. 2012; Zahn et al. 2016), whereas quadrats define a much smaller, discrete area, and provide a higher resolution snapshot of the substrate and associated fauna and flora (Parravicini et al. 2010; Sala et al. 2012; Mantelatto et al. 2013; Guinda et al. 2014; Dunmore et al. 2015). Survey area, transect/quadrat placement, assessment methods used, and whether or not transects or quadrats are fixed to the seabed (permanent), may all vary between studies (Table 6.3-2, Table 6.3-3, Table 6.3-4). These differences in design will influence biodiversity estimates and assessment of ecosystem function and may have implications for the accuracy and reliability of the resultant data. When designing projects, it is important to clearly define the research questions and to ensure that the survey approach provides sufficient statistical power to adequately answer those research questions. It is also essential to consider how the survey was designed when comparing data between surveys, to make sure you are comparing “apples with apples.”

It is important to consider the survey area and how that relates to the species being targeted in the survey. For example, transects used to monitor sparse populations of mobile reef fish are often wider than those used to survey species with a higher population density, commonly 5m wide for the former, and 1-2m for the latter (see Table 6.3-2 and Table 6.3-3). However, 0.25m^2 quadrats are considered to be appropriate when examining small scale changes in abundance and distribution of sessile organisms such as foliose and filamentous algae (Table 6.3-4; Murray 2001). The size and nature of the survey area are also important in terms of providing robust biodiversity estimates and assessments of reef health (functionality) over time. Adjusting either transect/quadrat size or the number of replicates sampled per site (Guinda et al. 2014) can affect the statistical strength and reliability of data (Brown et al. 2004, Houk & Van Woesik 2006; Parravicini et al. 2009; Guinda et al. 2014). While increasing the number of replicates (and therefore the area) is likely to increase the accuracy of the data, the optimal survey area will be unique to the target environment (Table 6.3-4 and Table 6.3-5; Guinda et al. 2014) and research questions. Few temperate rocky reef studies have directly addressed the optimisation of survey design, or considered how a given sampling area may adequately represent one reef but not another due to differences in heterogeneity and diversity (Guinda et al. 2014). It is also important to assess the optimal design in order to ensure that the metrics employed (e.g., biodiversity, functionality/ change over time) are suitable for the research aims. A pilot study to capture the natural variance within the temperate reef system and provide data for power analysis is a good way to inform this process (Carey & Keough 2002).

Healthy temperate rocky reefs tend to exhibit high biodiversity and substantial functional heterogeneity (Connell & Irving 2008; Gunderson et al. 2008; Ling 2008; Wernberg et al. 2009). As a result, the greater the area surveyed within a reef system, the greater the species richness that will be identified, until true species richness is reached. Where sampling area is constrained, this may lead to underestimation of species richness (Gotelli & Colwell 2001; Guinda et al. 2014). Consequently, where understanding whole reef biodiversity is the main focus, increasing the survey area through a greater number of randomly positioned sample sites for each survey may be the best way to capture the reef heterogeneity and produce credible estimates for extrapolation to broader scales (Rogers et al. 2000; Guinda et al. 2014).

Table 6.3-2 Transect Surveys-Reef Fish

Length	Width	Number of Replicates	Area	Depth placement	Permanent (yes/no)	Assessment methods	References
25m	5m	3	375m^2	?	No	Visual census	Sala et al. (2012)
25m	5m	12	1500m^2	Random	No	Visual census	Watson et al. (2005)
25m	5m	7-12	875 – 1500m^2	?	No	Visual census	Colton and Swearer (2010)
25m	5m	9	1125m^2	Random	No	Visual census	Willis and Babcock (2000)
30m	2m	12[a]	720m^2	Stratified random	No	Visual census	Tenera Environmental. (2006)
50m	(2x) 5m	4T	2000m^2	Stratified haphazard	No	Visual census	Edgar et al. (1997), Edgar and Barrett (1997), Edgar and Barrett (1999), Stuart-Smith et al. (2008)
50m	(2x) 5m	6 [b]	3000m^2	Stratified haphazard	No	Visual census	Valentine et al. (2016)
50m	(2x) 5m	4.4 (average)	2200m^2 (average)	Stratified haphazard	No	Visual census	Edgar and Stuart-Smith (2009)
100m	3m	1	300m^2	?	Yes	Visual census	Kushner et al. (1994)

NB: The aim of all of the above-listed studies was to characterise fish biodiversity or community assemblage.

a 4x replicates per depth zone; 240m^2 per depth zone

b 4x replicates at 5m depth, x2 replicates at 2m or 10m depth; 2000m^2 at 5m depth, 1000m^2 at 2m or 10m depth

c replicates following different depth contours; 500m^2 at each depth contour

Table 6.3-3 Transect Surveys-Invertebrates, Macroalgae and Cryptic Fish

Length	**Width**	**Number of Replicates**	**Area**	**Placement**	**Permanent (yes/no)**	**Assessment methods**	**References**
5-20m	1m	2-3	10-40m^2 to 15-60m^2	?	No	Cover mapping	Guinda et al. (2014)
30m	2m	2	120m^2	Stratified random	No	Visual census	Zahn et al. (2016)
30m	2m	6[a]	360m^2	Stratified random	No	Visual census	Tenera Environmental. (2006)
50m	1m	4	200m^2	Stratified haphazard	No	Visual census	Edgar et al. (1997), Edgar and Barrett (1997), Edgar and Barrett (1999), Stuart-Smith et al. (2008)
50m	1m	6[b]	300m^2	Stratified Haphazard	No	Visual census	Valentine et al. (2016)
50m	(2x) 1m	4.4 (average)	440m^2 (average)	Stratified haphazard	No	Visual census	Edgar and Stuart-Smith (2009)
50m	1m	4[c]	200m^2	Stratified	No	Visual census	Barrett et al. (2009)
20m	3m	12	720m^2	?	yes	Visual census	Kushner et al. (1994)
4.5-12m	?	8-122[e]	?	Perpendicular + parallel to coast	No	Percentage cover estimation + visual census	Nichol et al. (2009)

NB: The aim of all of the above surveys was to characterise benthic communities

a 2x replicates per depth zone; 120m^2 per depth zone

b 6x replicates at 5m depth, x2 replicates at 2m or 10m depth; 200m^2 at 5m depth, 100m^2 at 2m or 10m depth

c 4x replicates in one extended 200m transect

d replicates following different depth contours; 100m^2 at each depth contour

e 4.5 – 12m sub-transects were assessed every 30sec for a period of 15sec along transects varying in length from 200m to 1100m (speed: 0.3-0.8m/s). The number of transects assessed varied from between 8 and 22 on 200m long transects to between 45 and 122 for 1100m long transects.

Table 6.3-4 Quadrat details

Assessment methods	No. per site (most common)	Size range (m^2) (most common)	Purpose	Reference
% macro algae cover	4 – 20 (20)	0.25 – 2.00 (0.25)	Biodiversity	Edgar and Barrett (1999); Stuart-Smith et al. (2008); Kushner et al. (1994)
			Impact	Valentine et al. (2016); Crawford et al. (2006); Ling (2008)
			Biodiversity and impact	Edgar and Barrett (1997); Barrett et al. (2009)
% encrusting algae cover	5	0.04	Biodiversity and impacts	Sala et al. (2012)
% biota cover: algae and inverts (and/or other quantitative/ qualitative assessment)	2-129 (4)	0.024-1.95 (0.25,1.5)	Assessment method development	Parravicini et al. (2010)
			Biodiversity	Barrett et al. (2010); Smale et al. (2012); Garrabou et al. (1998);
			Impact	Dunmore et al. (2015); Oh et al. (2015); Balata et al. (2007)
% encrusting invertebrate cover	20	2.00	Biodiversity	Kushner et al. (1994)
% sessile organism cover	8	0.034	Impact	Glasby (1999)
Substrate characterisation	20 – 129 (20)	1.5 – 2.00 (-)	Biodiversity	Kushner et al. (1994); Barrett et al. (2010); Smale et al. (2012)
CFR index[1]	5-9 (-)	0.25 – 2.5 (-)	Impact	Guinda et al. (2014)
Sea urchin density	30	0.25	Biodiversity and impacts	Sala et al. (2012)

[1] CFR Index: characteristic macroalgae coverage (C), fraction of opportunistic species (F) and characteristic macroalgae richness (R).

When examining temporal trends, such as those needed for the establishment of a long-term monitoring program, ensuring the survey area is adequate is critical. Subtle temporal changes in reef health can be masked by variation in spatial heterogeneity (Rogers et al. 2000). For logistical reasons it may be desirable to reduce the survey area to minimise survey time; however, this may amplify variability observed between samples (Murray 2001; Parravicini et al. 2009; Guinda et al. 2014). If sample size is not large enough, small-scale variability between replicates may mask patterns of interest and overestimate habitat heterogeneity, resulting in reduced accuracy of results (Parravicini et al. 2009). Ensuring a thorough understanding of the experimental design and any limitations in statistical power to detect change over time (in areas of high patchiness and variation) will be vital for ensuring the long-term success of monitoring programs (Brown et al. 2004; Teagle et al. 2017). Where the aim is to determine temporal change, it may also be necessary to increase the power of the sampling design by increasing the size of the survey area, or alternatively by limiting the random variation within the data collected (Underwood & Chapman 2003). Variation can be minimised in a number of ways; for example, by choosing to only monitor indicator species (Juanes et al. 2008; Smith et al. 2008) or to focus on the survey of a particular habitat only.

Defining particular habitats for assessment will reduce the potential for other sources of variability to influence temporal change investigations (Rogers et al. 2000). However there needs to be a clear acknowledgement that the questions and conclusions are thereby limited to these design choices.

Fixing transects or quadrats permanently to the seabed, or using techniques that enable exact locations to be revisited during successive surveys can significantly reduce the ‘noise’ in data due to spatial heterogeneity, while eliminating the need to greatly increase sampling size (Rogers et al. 2000; Williams et al. 2010). This approach has been routinely used when examining temporal change in long term monitoring, or where resources may be limited as a cost-effective method of decreasing random variation and increasing statistical power in both tropical (e.g. Brown et al. 2004; Jokiel et al. 2015) and temperate systems (e.g. Dunmore et al. 2015). As part of a survey to assess the impact of salmon farming on temperate rocky reefs in New Zealand, Dunmore et al. (2015) repeatedly surveyed four permanent quadrats to achieve the statistical power necessary to detect specific impacts over time. While fixed surveys can reduce the necessity for larger survey areas, care must be taken in extrapolation of data where there is no contextual information such as follow up investigations over the broader spatial scale. Furthermore, it is important to ensure the selection of sites adequately represents the area of interest, as this will increase the likelihood of detecting change from selected indicators over time (Keough et al. 2007). Whichever method is chosen, understanding the implications of fixed versus random quadrats on the interpretation of data and the statistical methods chosen for analysis is critical.

Another source of variation in data is placement of transects on rocky reefs. Benthic communities vary with depth due to changes in the physical, chemical and biological environment (Juanes et al. 2008). Studies which use transects that run across a depth gradient will provide a more holistic picture of the diversity of the reef as a whole (Nichol et al. 2009), but will likely have a much greater inherent variability. Transects which follow depth contours will reduce variability due to depth and therefore increase the potential for characterising change over time or between multiple sites but only <u>at that depth </u>(Juanes et al. 2008). It is possible to examine the effect of depth on variability by using a stratified approach with multiple transects over multiple depth contours (Table 6.3-2, Table 6.3-3).

Determining which quadrat size, what number of replicates, where to position quadrats or whether to use a fixed versus random design is critical in survey development and should take into account the study aims, resources and reef characteristics (Murray 2001; Brown et al. 2004; Guinda et al. 2014). If long-term monitoring programs for temperate reefs are being designed around a specific impact, a robust baseline against which to compare change over time is also vital. Ideally, a baseline would include multiple control and impact sites, as well as data prior to the commencement of the impact (i.e., before, after, control, impact (BACI) design; (Benedetti-Cecchi 2001; Hewitt et al. 2001).

Most importantly, when designing a study or monitoring program with the intent of detecting change over space or time, having clear objectives is vital to frame an effective design (Keough et al. 2007). For example, if a 50% decline in canopy forming algae will extensively reduce ecosystem function, a monitoring program which can only reliably detect an 80% loss of canopy forming algae will not be fit for purpose. Defining “acceptable” levels of change is crucial to ensure the experimental design is robust and will achieve the outcomes required to inform management. This can be a challenge in heterogeneous temperate reef habitats; a pilot study can provide an understanding of the system variability and how that might affect the sensitivity of proposed techniques to address the specific management requirements. This may include power analysis to accurately determine the design parameters required to detect a significant difference across a broad range of indicators and criteria.

6.3.2.2 Assessment techniques

Assessment technique will be determined by the research question, which is in turn influenced by the survey focus (i.e. biodiversity vs functionality) and subsequent data analysis (Kushner et al. 1994; Garrabou et al. 1998; Murray 2001; Stuart-Smith et al. 2008; Parravicini et al. 2010; Sala et al. 2012; Guinda et al. 2014). Most studies assess transects through underwater visual census to provide density estimates of individual species or to score a variety of parameters (Table 6.3-2, Table 6.3-3). As transects are generally used to survey large, mobile species, it is recommended an acclimatisation period is included prior to undertaking a visual census after the transect line has been laid (Smith et al. 2008). Quadrats are generally used to collect either percentage cover or abundance estimates for less mobile benthic organisms so an acclimatisation period may not be necessary (Table 6.3-5).

Abundance estimates are generally obtained by simply counting all the individuals of specific taxa within the quadrat area (Kushner et al. 1994; Sala et al. 2012), but there are a number of other approaches available to estimate proportional cover of algae and encrusting organisms, including point count and visual cover estimation (Table 6.3-5). The point count technique is the most applied method to estimate cover; parameters of interest are identified and scored beneath regular or randomly placed points within the quadrat. Point counts can be done both in-situ using traditional quadrats, or through subsequent photo analysis in the laboratory. In biodiversity assessments, more points per quadrat will enable more evenness in the data and will allow for smaller and less common organisms to be counted. Visual cover estimation can also provide a means to assess benthic cover both in-situ and through post-processing, although it may be more subject to observer bias than other forms of assessment. A number of approaches have been adopted to try and overcome this bias, including the use of scoring systems or categories to assess cover, or by simply estimating the percentage cover of each organism within a subdivision of the quadrat area (Mantelatto et al. 2013). Where visual cover estimates are used, using multiple assessors can help to quantify and address any observational variability between assessors.

Table 6.3-5 Quadrat Assessment Methods

Method	Description	References
Point count	Percentage cover is estimated by counting the number of times each species is located underneath a points across the quadrat area.	Balata et al. (2007), Barrett et al. (2010), Edgar and Barrett (1997), Edgar and Barrett (1999), Glasby (1999), Ling (2008), Oh et al. (2015), Smale et al. (2012), Stuart-Smith et al. (2008), Valentine et al. (2016)
Cover mapping	Area covered by different benthic organisms is assessed by tracing around organisms and converting to area and percent cover.	Dunmore et al. (2015), Garrabou et al. (1998), Guinda et al. (2014)
Cover estimation	Percentage cover of benthic organisms is estimated by an observer.	Sala et al. (2012)
	Cover estimated by an observer who assigns a score to each species within a quadrat subdivision (0 = absent – 4 = 100 % cover) and summing the scores across all subdivisions.	Parravicini et al. (2010)
Frequency count	Benthic organism occurrence frequency, expressed as a percentage of the number of quadrat subdivisions is estimated by counting	Parravicini et al. (2010)

	the number of quadrat subdivisions within which each species occurred.	
Quadrat count	The abundance of targeted species is estimated by counting all specimens of within the quadrat area.	Kushner et al. (1994); Sala et al. (2012)

6.3.3 Metrics for monitoring change on temperate rocky reefs

Several metrics have been used to assess change in temperate rocky reefs. These fall into two main categories: biodiversity and ecosystem function. Biodiversity metrics include taxonomic diversity, species richness and other metrics based on the number and variety of species (Edgar and Barrett 1997; Ling 2008; Somerfield et al. 2008; Stuart-Smith et al. 2015), while ecosystem function metrics focus on the services provided by the organisms within the study area and how changes in species/ groups of species might affect those services (Airoldi et al. 2008; Somerfield et al. 2008; Stuart-Smith et al. 2013; Laureto et al. 2015). For instance, canopy forming taxa provide a dynamic and complex habitat which supports biological, functional, and structural diversity. If that community changed to one dominated by ephemeral species, this would support lower levels of biological and functional diversity which could be observed with many of the ecosystem function metrics. Ecosystem function metrics have been used effectively at both a local and global scale (Eriksson et al. 2002; Lotze et al. 2006).

6.3.3.1 Biodiversity

Understanding the biodiversity of a rocky reef may be the study objective, for example, in assessing conservation or remediation efforts. Many studies have demonstrated that healthy marine environments tend to exhibit higher biodiversity and food web complexity, while impacted areas are characterised by reduced biodiversity and shorter food webs (Reish 1955; Pearson & Rosenberg 1978; Graham 2004; Lotze et al. 2006; Edgar et al. 2010; Stuart-Smith et al. 2015; Fowles et al. 2018; Ling et al. 2018). For example, Stuart-Smith et al. (2015) and Ling et al. (2018) demonstrated that at sites heavily impacted by urbanisation, decreased biodiversity and shorter food webs with fewer fish and invertebrate species were common, and smaller fish species with fast life history strategies (i.e. *r*-selected) were likely to dominate. Similarly, within algal communities, small, fast growing opportunistic species, such as turfing and filamentous algal communities were generally associated with highly urbanised sites (Fowles et al. 2018a; Fowles et al. 2018b). Anthropogenic pressures such as urbanisation and organic pollution are often not the main cause of species extinctions in temperate reef environments but have been linked to declines in habitat forming species, such as kelp, which may result in reduced resilience of the system (Lotze et al. 2006).

Biodiversity data is best collected by undertaking a full visual census, with species identified to the lowest taxonomic level and counted (e.g. Edgar & Barrett 1997; Stuart-Smith et al. 2015). This information can then be used to calculate a number of diversity indices, from a simple number of species present (i.e. species richness) to more complex measures such as the Shannon diversity index (H), which takes into account abundance and evenness of species (Ling 2008; Colton & Swearer 2010; Stuart-Smith et al. 2015). Overall, biodiversity assessments are the most informative way to capture robust information on ecosystem structure that can be used to detect broad shifts in community assemblages over space and time.

6.3.3.2 Ecosystem function

Functional traits of rocky reefs have been shown to change in a systematic way in response to environmental pressures. Healthy resilient reefs are associated with high functional diversity, largely due to the presence of perennial canopy forming and coralline algae (Eriksson et al. 2002; Graham 2004; Claudet & Fraschetti 2010; Coleman & Wernberg 2017; Fowles et al. 2018; Fowles et al. 2018a; Fowles et al. 2018b). The removal of patches of canopy forming species, either as a result of natural processes or anthropogenically driven events, can initiate a phase-shift to patches dominated by ephemeral or turf algae (Bokn et al. 2003; Graham 2004; Russell et al. 2009; Falkenberg et al. 2012). Ephemeral turfing, filamentous, and epiphytic algal growth has been associated with a general decline in reef health and a corresponding loss of functional diversity (Bokn et al. 2003; Airoldi et al. 2008; Connell & Irving 2008; Campbell et al. 2014). A shift from canopy forming algae to turfing ephemeral mats creates a functional change within the ecosystem and influence biodiversity. There is a need for reliable assessment methods to monitor change on reefs over time and provide warning of such potential functional shifts in ecosystems.

Ecosystem function can be assessed by detailed visual census surveys or targeted surveys aimed at identifying changes in functional groups or functional performance. There is no “one size fits all” approach for this, as key assessment indicators will be regionally specific. Surveys to assess ecosystem condition often include some measure of the diversity of functional groups, or percentage cover of perennial macroalgae or other target species (Connell & Irving 2008; Connell et al. 2008; Gorman et al. 2009; Coleman & Wernberg 2017). They may also include a measure of degradation such as change in cover or presence of ephemeral algae (Gorman et al. 2009). Some indicators are better at detecting environmental deterioration than others (Ling et al. 2018); consequently, the choice of indicator is important and should be tailored to the aims of the assessment. Empirical evidence suggests lower functional diversity, decreases in canopy cover, and increases in ephemeral algae are all indicative of impacted reefs; however, measures of ecosystem function are not used as widely as measures of biodiversity (Airoldi et al. 2008). There is scope to further employ functional approaches to detect change in temperate reef ecosystems, particularly in the development of monitoring programs which seek to provide early warning of ecosystem stress.

In summary, metrics of ecosystem function provide different information to biodiversity metrics. Whilst these can be applied independently, in some circumstances they can be used as a proxy for biodiversity. It may be possible to refine these techniques to specifically inform our understanding of temperate reef resilience, and to identify critical stages in impact assessment. Importantly, functional and biodiversity assessments are not mutually exclusive. In a management context, the two approaches are complementary, with change in functionality triggering the need for a full biodiversity assessment to assess the ramifications of the functional change.

6.3.4 Collection methods in rocky reef monitoring

6.3.4.1 Background

Data collection for rocky reef monitoring is no longer synonymous with in-situ visual assessment by taxonomically trained scuba divers (Table 6.3-5). Underwater photographic and video technologies allow footage of the benthic environment to be collected for later assessment off site (Shortis et al. 2007; Mallet & Pelletier 2014). While in-situ scuba techniques are still the most common approach for reef surveys (Table 6.3-1), the use of remote assessment techniques will increase as technology advances. In-situ and remote collection methods present different advantages and disadvantages depending on the reef environments, resource limitations and/or data requirements.

6.3.4.2 In-situ assessment approaches

In-situ assessment by scuba divers is a long-standing method for collecting data on rocky reefs and is especially important where a comprehensive understanding of reef biodiversity and dynamics is required (Watson et al. 2005; Mallet & Pelletier 2014). In-situ scuba methods allow for detailed species identification and observation of cryptic fish and mobile invertebrates. This approach generally involves a scuba diver undertaking a visual census (i.e., recording observations from a quadrat and/or transect; (e.g. Edgar et al. 1997; Watson et al. 2005). The biodiversity surveys developed by Edgar & Barrett (1997) and the Reef Life Survey (Edgar & Stuart-Smith 2014) have been adopted widely in Australia and elsewhere as a standardised approach to collect high quality data on the biodiversity of temperate reef ecosystems (e.g. Edgar & Barrett 1999; Stuart-Smith et al. 2008; Edgar & Stuart-Smith 2009; Valentine et al. 2016).

Where research questions are specific, a full biodiversity assessment may be unnecessary, and the scope of a visual census may be narrowed to focus on particular elements of ecosystem function or certain indicators (Parravicini et al. 2009; Guinda et al. 2014; Zahn et al. 2016). This may be an appropriate approach to examine anthropogenic impacts, as by focusing on key response parameters the power to detect impact can be increased. For example, reducing the dive time per site can free up resources to increase sample size and/or replication.

There are, of course, limitations associated with in-situ assessments. Single diver observation and data recording can introduce bias into data. Diver to diver differences can skew data, leaving no permanent visual record of reefs for future interpretation, calibration, or alternative assessment. The very presence of divers during in-situ assessments can also influence organism behaviour, and the depth of observation is restricted (Mallet & Pelletier 2014). Remote assessment methods can be used in place of or in conjunction with in-situ observations to try to address these challenges.

6.3.4.3 Remote assessment approaches

Underwater cameras can be used to capture underwater transect and quadrat footage, either by diver or via remotely operated and autonomous underwater vehicles (ROVs and AUVs respectively), or with towed cameras. This technology has enabled much greater seafloor coverage than in-situ observations (Nichol et al. 2009; Van Rein et al. 2009; Barrett et al. 2010; Guinda et al. 2014; Mallet & Pelletier 2014). In addition, unaccompanied underwater cameras, developed specifically to monitor fish assemblages (e.g. baited or unbaited underwater remote video cameras ((B)RUVs), remotely controlled or autonomous drop cameras) provide a means of observing reef fish within their habitat without the potentially disturbing presence of divers, ROVs or AUVs (e.g. Willis & Babcock 2000; Harasti et al. 2015; Walsh et al. 2016).

The data which can be generated by the different remote assessment approaches will differ in its quality, sensitivity and application, and the techniques themselves have different operational requirements (Nichol et al. 2009; Barrett et al. 2010; Langlois et al. 2010; Guinda et al. 2014; Mallet & Pelletier 2014). Thus, it is important to consider the suitability of each approach to address the question when determining which to include in a survey.

6.3.4.3.1 Hand-held cameras

Using hand-held cameras requires divers in-situ to film or photograph the area for characterisation at each site. This approach offers many of the benefits of in-situ assessment while minimising the dive time required per site by having the assessment conducted after the fact (Mallet & Pelletier 2014), thus allowing for a greater area to be surveyed. It also provides the ability to review footage multiple times, which allows for different analyses and cross-checks without additional field time. While this approach will still introduce an element of bias as a function of the potential change in fish behaviour around divers, there are many benefits including the ability to measure fish length more accurately (Watson et al. 2010).

Interestingly, the ability to undertake species identification and observe cryptic species is more limited in photographs or video footage than using in-situ assessment as it removes the interaction with the survey area (i.e., the diver is not able to observe and interact with species for identification, or to search complex habitats). The quality and resolution of footage is also sensitive to water clarity and the specifications of the camera. These features can reduce the ability to identify to species level, with concomitant effects on species richness values and estimates of biodiversity (Gotelli & Colwell 2001; Guinda et al. 2014).

6.3.4.3.2 Autonomous underwater & remotely operated vehicles

Autonomous underwater and remotely operated vehicles (AUVs and ROVs respectively) offer robust survey tools for quantifying temperate reef habitat at depths beyond what is possible for a diver (Sward et al. 2019). AUVs can collect images of benthic data on temperate rocky reefs using a grid or photo mosaic approach, whereas ROVs record and stream video back to the surface which can be analysed (Table 1; Barrett et al. 2010, Williams et al. 2010; Smale et al. 2012; Williams et al. 2012; Pizarro et al. 2013; Guinda et al. 2014; Guinda et al. 2014).

AUVs are primarily used in benthic assessments, with photographs or videos stitched together to provide an uninterrupted image of the reef at the broadscale, but with sufficient quality that any individual image can be magnified to enable analysis at the finer scales. Images can be assessed as either traditional quadrats or transects (Pizarro et al. 2013; Guinda et al. 2014). Collecting data in this way provides flexibility to test the pros and cons of multiple assessment methods and statistical approaches. AUVs can carry a variety of monitoring and surveying equipment giving them the capacity to take video/ photographic footage as well as physicochemical or geophysical samples/measurements (Smale et al. 2012). These vehicles can be programmed to follow predefined routes or grids thereby photographing large areas of seabed at varying scales depending on the resolution required (Pizarro et al. 2013). Technology now allows sufficient accuracy in positioning and image matching to enable footage of the same area of seabed to be collected on subsequent surveys, and compared over time (Williams et al. 2010).

Unlike AUVs, ROVs are operated live enabling increased control in navigation. This enables the ‘pilot’ to respond as needed, offering greater flexibility and improving the ability to capture footage of the more cryptic species than when using AUVs. ROVs are of particular use in quantifying fish assemblages and reef structure as they are more interactive and responsive on a 3-dimensional scale than AUVs. Many ROVs also have the capacity to record still images in addition to video footage, extending their application to both benthic and pelagic aspects of temperate reef communities at depth.

AUVs and ROVs provide an ability to increase the area and depth of reef surveys (Sward et al. 2019). However, as with other remote assessments, there will be a potential loss of resolution and flexibility for species identification (Guinda et al. 2014). The ability to identify to species level and/or identify cryptic species, highly motile species, or those which are masked by canopy forming macroalgae is reduced as identification relies on photographs or videos. As with any method relying on imagery, the success of AUV or ROV surveying depends in part on water clarity and visibility. Though AUVs and ROVs are valuable tools, they cannot be used in very shallow water in case of damage, and similarly, they may not offer a practical solution in environments with a dense canopy (Barrett et al. 2010) due to the possibility of the thrusters becoming entangled (Guinda et al. 2014). Where entanglement is a significant concern, ROVs would have to navigate above the canopy, introducing issues associated with the canopy masking sub-canopy species and photograph resolution. The initial outlay in terms of cost and expertise required to operate this technology may also be prohibitive for many situations. Decisions to use AUVs and ROVs must also consider post-processing costs, as it is easy to generate hundreds of hours of video or thousands of images which then need to be analysed by skilled personnel, which often amounts to many hours of processing. Despite this, technological advances may help to overcome several of the issues, and it is expected their use will continue to grow into the future.

6.3.4.3.3 Stationary camera

Stationary cameras are primarily used to collect data on the distribution, species richness and abundance of reef fish (Willis & Babcock 2000; Colton & Swearer 2010; Harasti et al. 2015). This method generally consists of an underwater camera being deployed on the seabed where it films and records any fish which pass the camera lens. Cameras are set to record video footage within a frame attached to the camera housing for a fixed period. These systems have been used for long term monitoring of reef environments (Chabanet et al. 2012; Mallet & Pelletier 2014).

Stationary cameras can be either baited or unbaited (Watson et al. 2005). Because unbaited camera systems rely on fish randomly passing by the camera lens, a greater number of sampling sites are required to get the same statistical power as that of baited cameras (Watson et al. 2005). Baited cameras have been shown to record a greater abundance and species richness, including observations of rare predatory fish; however, unbaited cameras have the advantage of allowing observation of natural behaviour (Watson et al. 2005; Hardinge et al. 2013). While baited stationary cameras can provide detailed information on species richness, estimating relative densities poses a greater challenge due to difficulties in discerning the area from which fish have travelled to access the bait (Watson et al. 2005; Shortis et al. 2007; Colton & Swearer 2010).

A key advantage of stationary cameras is the ability to observe reef fish assemblages as they swim into the camera view without the influential presence (and associated bias) of scuba divers or other remote monitoring equipment (Watson et al. 2010). This enables biases associated with other techniques to be quantified, improving the interpretation of results (Colton & Swearer 2010). In addition, stationary cameras can be deployed at depths much greater than possible by diver sampling (Logan et al. 2017). While stationary cameras have been used as a means of overcoming some of the limitations associated with the more traditional visual census techniques, they are subject to their own suite of biases. Suitability of method is largely dependent on the type of fish being targeted (Willis & Babcock 2000; Watson et al. 2005; Colton & Swearer 2010). For example, predatory and scavenging fish assemblages tend to be better quantified using stationary cameras than in-situ diver surveys (Watson et al. 2005; Colton & Swearer 2010; Murphy & Jenkins 2010). However, there are difficulties in using baited stationary cameras for monitoring cryptic, herbivorous and territorial fish (Colton & Swearer 2010; Logan et al. 2017). Where aggressive species of fish are present, deployment times beyond 5-10 minutes have been shown to limit species richness estimates as some fish species are prevented from accessing the bait (Willis & Babcock 2000; Colton & Swearer 2010). Therefore, to characterise the full reef fish assemblage, it is recommended that stationary cameras are deployed alongside visual census techniques (Murphy & Jenkins 2010; Logan et al. 2017).

6.3.4.3.4 Towed camera

Using a vessel at the sea surface, towed cameras are pulled at a set height across a reef enabling video footage of the reef to be recorded (Mallet & Pelletier 2014). Unlike ROVs, towed cameras do not require thrusters, reducing the risk of entanglement and making them a more durable alternative in areas with a thick canopy. The towing speed and height of the camera are controlled and can be varied from the towing vessel depending on the requirements of the footage (a balance between video quality and spatial coverage), reef geometry and structure (Barrett et al. 2010; Mallet & Pelletier 2014; Logan et al. 2017).

As they are towed at speeds generally around 1 knot, towed video can cover a far greater area than would be possible by diver or ROV. The recent advent of high definition footage and large capacity memory cards has allowed for large blocks of sampling to be undertaken during a day (Mallet & Pelletier 2014). However, the capacity to sample a large area needs to be offset against the reduced resolution in comparison to in-situ techniques. Because control is from the surface via an umbilical cord, towed cameras do not have the high-accuracy navigation (vertical and horizontal) that is possible with an ROV. Furthermore, like other video techniques, in areas where there are dense algal canopies, the capacity to assess substrate, cryptic fish and invertebrates is limited. Thus, sampling using towed video works best for research questions focused on changes in broad parameters (i.e., canopy cover).

6.3.4.4 In-situ vs remote techniques

The collection of data through in-situ assessment has been the prevailing method for quantitatively characterising temperate rocky reefs ecosystems since snorkel and SCUBA were first developed. However, technological developments, from relatively simplistic methods such as hand-held cameras to highly technical ROVs, have dramatically expanded the scope and capacity for remote data collection (Van Rein et al. 2009; Mallet & Pelletier 2014). As technology continues to evolve, remote methods for collecting temperate reef data are likely to become more widespread.

Despite advances in technology, in-situ methods remain the most effective way to collect high-quality, high-resolution data across all components of temperate reef ecosystems (i.e., macroalgae, invertebrates and fish; Table 6.3-1). The level of detail that can be obtained from in-situ assessment by divers cannot be matched by remote assessment methods, however a key constraint for diving surveys is bottom time and depth, which in turn may potentially limit the research scope of surveys (Table 6.3-5). Increasing the potential survey area has been a driving influence for the development of remote technologies. Towed cameras, AUVs, and ROVs have dramatically increased the spatial area and depth that can be covered while reducing dive time and providing a permanent record of the sampling (Preskitt et al. 2004; Smith et al. 2008; Edgar et al. 2009; Parravicini et al. 2009; Underwood & Jackson 2009; Guinda et al. 2014). As the power to detect change is often reliant upon the total area (sample size) surveyed, this is a compelling reason to incorporate remote methods into temperate reef studies (Benedetti-Cecchi 2001). However, substituting in-situ assessment entirely with remote methods will result in a loss of detail, with accurate data on cryptic species and substrate characterisations generally difficult to obtain using these techniques (Barrett et al. 2010; Guinda et al. 2014). Indeed, it is a sliding scale of offset between accuracy and coverage, with methods such as towed camera, being able to cover the greatest spatial area in the least amount of time, but also resulting in the greatest loss of detail.

From a logistical perspective (i.e., personnel requirements and outlay on equipment), there are pros and cons of both in-situ and remote methods (Table 6.3-6). All methods require the use of skilled personnel in data collection and analysis. In-situ methods are generally more resource intensive in terms of personnel, requiring a team of divers that are highly trained in both SCUBA techniques and collection methods. However, beyond scuba equipment and a boat, in-situ methods require a relatively small outlay in terms of equipment. Remote methods also require skilled personnel to operate the technology; though it is worth noting it is not imperative for the person operating the technology to also have the skill set needed to collect the data from the footage. While in-situ methods may require more personnel to complete surveys, all the data is generally collected on the day of the survey, whereas a lengthy post-processing period is often associated with remote survey methods (Table 6.3-6). This post-processing period may negate any time-saving advantage gained during the collection of footage. Remote methods have the advantage of overcoming the diver bias associated with in-situ assessment (Preskitt et al. 2004; Houk & Van Woesik 2006; Parravicini et al. 2009), although diver bias may be somewhat addressed using in-situ techniques, either through rigorous diver training and/or side-by-side assessment allowing the variation between divers to be accounted for (Bernard et al. 2013). Furthermore, while the effort required to train a team of divers may be costly, there is intrinsic value in having the research team immersed in their study environment, a benefit not gained when using remote assessment methods.

Ideally, a combination of both in-situ and remote methods would be used to complement each other, with the remote assessment allowing greater spatial coverage, and the in-situ assessment allowing a greater level of detail at selected sites. This approach would result in a more robust assessment of conditions across multiple scales, and a thorough comparison of collection methods. A direct comparison of the differences in information obtained from remote versus in-situ collection techniques would be extremely useful to refine experimental design, optimise data collection methods, balance the need for detail versus total area assessed, and target resources most effectively towards achieving the survey aims. This would be particularly valuable for long-term monitoring and research projects with a lifespan of several years.

Unfortunately, the reality of finite resources means that running parallel studies using both in-situ and remote techniques is rarely possible. Maximising effective time spent in the field collecting data is key to successful outcomes, whether that be for research or to inform management. As noted at the start of this review the level of detail required for a survey and thus selection of methods for data collection will largely depend on the question being investigated. Using current technology, in-situ diver assessment techniques provide the greatest taxonomic resolution and therefore the most accurate representation of biodiversity. As such, this approach is still routinely used in rocky reef assessments where conservation values, baseline conditions and global pressures are the focus (e.g., Barrett et al. 2007, Barrett et al. 2009, Day et al. 2018). In contrast, where studies focus on broad ecosystem function (e.g., effects of anthropogenic pollution), remote techniques may offer the potential to obtain a more rapid assessments across a greater spatial area. A technique such as towed video has the capacity to cover a large spatial area, so if a key question for the monitoring or research program regards the spatial extent of the impact, then towed video is a valid technique to employ, provided the comparative coarseness of the data is acknowledged. In a well-designed reef assessment survey with clear objectives, remote and in-situ collection methods, along with biodiversity and functional indicators can be used to complement and inform each other.

Table 6.3-6 Summary of positives and negatives of in-situ vs remote assessments.

	Positives	**Negatives**
In-situ	• Higher level of detail possible across all ecosystem parameters • Identification of cryptic and smaller species possible • Increased accuracy of substrate characterisation • Relatively small outlay on equipment (beyond SCUBA requirements) • No further data processing required in the laboratory	• Can cover a relatively small area spatially • Limit to depth of study sites • Large team of divers often required
Remote	• Possible to cover a much larger area spatially • Able to cover a much greater depth range • Permanent record of transects/quadrats maintained • Fewer field personnel required • Can remove biases associated with multiple personnel collecting data.	• Data often coarse compared with in-situ techniques, particularly at sites with dense algal canopies. • Data quality highly dependent on water clarity/visibility • Lengthy post-processing procedures necessary in the laboratory • Large financial outlay for equipment

6.3.5 Conclusions

Temperate reefs are largely undervalued but highly important ecosystems that support significant ecological, social, and economic values. As anthropogenic pressure on coastal environments increases, it is increasingly important to understand, monitor and manage those impacts. The ability to accurately detect change in temperate reef ecosystems will underpin successful management into the future. However, reefs are complex multi-dimensional habitats with an inherently high degree of spatially and temporal variability, which makes designing assessment surveys challenging, with anthropogenically-driven change often difficult to clearly distinguish from other reef processes. However, failing to detect a loss of resilience in temperate reef environments will impair successful management. There are a number of ways to increase the likelihood of successfully detecting change in temperate reef ecosystems; i) ensuring the study design is robust and has the power to detect change, ii) broadening the metrics beyond biodiversity and incorporating ecosystem functionality into survey parameters to improve capacity to detect change, and iii) continuing to develop and adopt remote technology in order to broaden the spatial area of surveys and compliment high accuracy diver collected in-situ data. In combination this will enable change to be assessed across multiple levels and provide a much better understanding of ecosystem condition.

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6.4 Characterising biodiversity of rocky reefs in Southeastern Tasmania

6.4.1 Introduction

Tasmanian salmon aquaculture accounts for 99% of the total Australian salmonid production with a current gross value of $898 million (ABARE 2020). The industry is in a phase of expansion, with new lease areas established around North Bruny Island (Storm Bay 1-4) and the Southern D'Entrecasteaux Channel (Lippies). The establishment of these new leases has caused some stakeholder groups to question the potential for the farms to impact nearby rocky reefs. Many of these reefs are associated with other high-value commercial seafood production such as abalone and rock lobster, which together are worth over $150 million to the state per annum (Johnson et al. 2005; Valentine et al. 2016). As aquaculture production biomass increases in any given area, the risk of broadscale effects on sensitive habitat, such as rocky reefs, may also increase. Intensive aquaculture may increase nutrients in nearby reef ecosystems, and if so this can have ecological consequences, with worse case outcomes being functional change in habitat structure and the macro-algae community and adverse impacts on the overall diversity (Worm et al. 1999; Connell et al. 2008; Oh et al. 2015). A robust baseline assessment of rocky reefs in areas prior to the introduction of farming will help to assess salmon farming in system-wide context and thus help inform both farm based and regulatory monitoring.

Baseline data prior to disturbance is critical to understanding the way an ecosystem will respond to change. Anthropogenic disturbance can influence the biological dynamics of the various organisms within an ecosystem (Dayton 1971; Petraitis et al. 1989; Goodsell & Connell 2005). Algal and sessile invertebrate communities will respond to changes in nutrient, heavy metal and sediment dynamics that are commonly associated with anthropogenic impact. These impacts may also affect mobile fish and invertebrate species, both directly, or indirectly through food-web alteration (Eriksson et al. 2002; Goodsell & Connell 2005; Connell et al. 2008; Ling et al. 2018). Altered environmental conditions such as changes to light availability and exposure will have a major influence on reef community composition (Alexander et al. 2009; Fowles et al. 2018b). Features such as reef rugosity influence the topographical heterogeneity of the ecosystem, which in turn influences the effects of varying types of disturbance (Witman & Dayton 2001; Balata et al. 2007). Thus, understanding the biodiversity of rocky reef habitats prior to disturbance is critical for establishing a baseline of the ecosystem.

Traditionally, assessments of rocky reefs are undertaken by diver surveys using underwater visual census (UVC) techniques (Edgar et al. 1999; Edgar & Stuart-Smith 2014; Ling et al. 2018). Assessing overall reef biodiversity by implementing the 'Edgar-Barrett' method (Edgar & Barret 1997; Edgar et al. 1999), allows the abundance and diversity of all macroalgae, fish and invertebrates to be captured (Valentine et al. 2016). UVC methods were designed to maximise detection of (i) changes in population numbers and size-structure (ii) cascading ecosystem effects associated with disturbance and (iii) long term change and variability in reef assemblages. The 'Edgar-Barrett' methodology is detailed and consistent across different types of reef; it allows for both predicted and unanticipated impacts on rocky-reef communities to be examined, and provides a valuable reference to measure future change, making it applicable for broad-scale monitoring programs into the future (Valentine et al. 2016).

In this study the ‘Edgar-Barrett’ approach was used to assess the overall reef biodiversity in the North Bruny Island and Southeast D’Entrecasteaux Channel (SE Channel) regions. The aims were to characterise the biodiversity at sites within each region and investigate any broad spatial trends that might exist within the two regions. The relationship between reef biodiversity and two of the main natural environmental drivers of temperate reef ecology, exposure, and reef rugosity, were also explored. Overall, these data will provide a benchmark from which to evaluate any future change and insights into the key variables of change in this region.

6.4.2 Methods

6.4.2.1 Field methods

Diver surveys were undertaken following the “Edgar-Barrett” underwater visual census (UVC) method, to provide a quantitative assessment of fish, large mobile invertebrates and macroalgae on rocky reef (Edgar & Barrett 1997; Edgar et al. 1997). At each site 4 x 50 m transects were haphazardly laid along the 5 m or 10 m depth contour, with all fish, invertebrate and algae species along the transect counted and recorded by a team of 2 – 4 divers. Details of each site and sampling event, including the date and GPS location, were recorded (Figure 6.4-1). The surveys were undertaken during late summer and autumn in 2016 (North Bruny Island) and 2017 (Southeast D’Entrecasteaux Channel) at 13 sites surveyed in the North Bruny Island region (hereafter North Bruny) and 15 sites surveyed in the Southeast D’Entrecasteaux Channel region (hereafter SE Channel). Surveys were conducted at 5 m depth at all sites, with 2 x 50 m transects undertaken at 10 m depth at selected sites (Table 6.4-1).

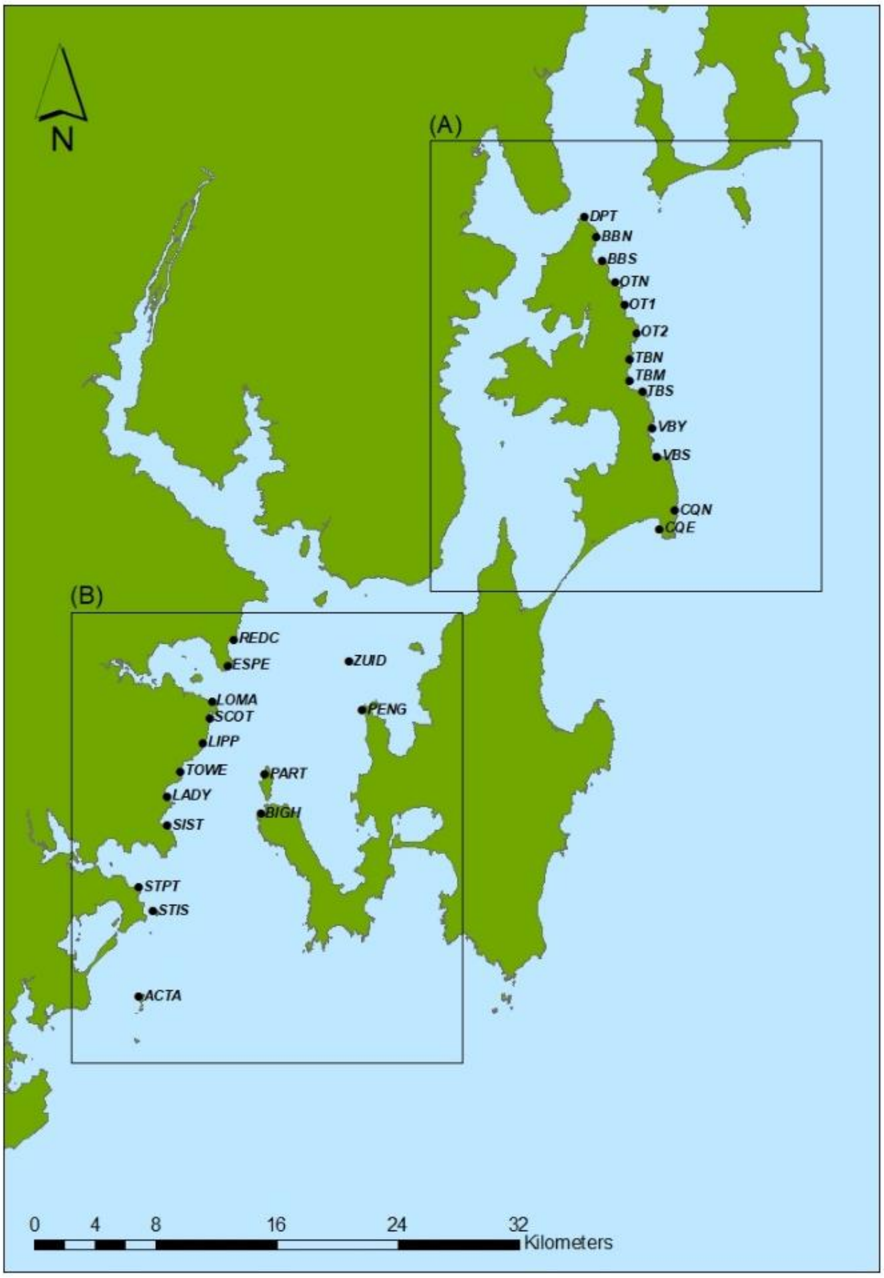

Figure 6.4-1 Map of Southeastern Tasmania showing temperate reef sites for biodiversity surveys within the two regions, being A) North Bruny and B) Southeast D'Entrecasteaux Channel. See Table 6.4-1 for details of site coding and transects conducted.

Table 6.4-1 Abbreviation codes and depth strata surveyed for all sites in the North Bruny Island and Southeast D'Entrecasteaux Channel regions.

North Bruny Island			***Southeast D'Entrecasteaux Channel***		
Site name	**Site code**	**Depth strata surveyed**	**Site name**	**Site code**	**Depth strata surveyed**
Dennes Point	DPT	5 m	Zuidpool Rock	ZUID	5 m, 10 m
Bull Bay North	BBN	5 m, 10 m	Penguin Point	PENG	5 m
Bull Bay South	BBS	5 m	Redcliffs	REDC	5 m
One Tree North	OTN	5 m	Esperence Point	ESPE	5 m

One Tree South 1	OT1	5 m	Lomas Point	LOMA	5 m
One Tree South 2	OT2	5 m	Scott Point	SCOT	5 m, 10 m
Trumpeter Bay North	TBN	5 m, 10 m	Lippies Point	LIPP	5 m, 10 m
Trumpeter Bay Mid	TBM	5 m	Tower Bay	TOWE	5 m, 10 m
Trumpeter Bay South	TBS	5 m, 10 m	Lady Bay	LADY	5 m
Variety Bay	VBY	5 m	Sisters Bay	SIST	5 m
Variety Bay South	VBS	5 m, 10 m	Partridge Island	PART	5 m
Cape Queen North	CQN	5 m	Southerly Bight	BIGH	5 m
Cape Queen Elizabeth	CQE	5 m, 10 m	Southport	STPT	5 m
			Southport Island	STIS	5 m, 10 m
			Actaeons	ACTA	5 m, 10 m

6.4.2.2 Macroalgae

Macroalgae were surveyed at 10 m intervals along the transect line using a 0.25 m^2 quadrat with a grid of 7x7 wires crossing perpendicularly. Macroalgal cover was assessed by identifying and counting algae species occurring directly under the 50 (49 plus one corner) grid positions. The point-count value for each species was converted to percentage of the total area for all algal species. Algae were surveyed in layers, with percent cover of canopy species recorded first. These were then pushed aside exposing the understorey species for counting. Following assessment, understorey was also pushed aside to allow for a substrate assessment. Unknown or unidentifiable species were either collected for later identification in the laboratory or recorded at genus level. In cases where the taxonomic level was more uncertain (such as for many sessile invertebrates including sponges), taxa were recorded at the highest practicable level (e.g. order). As percentage cover of giant kelp, *Macrocystis pyrifera*, is very difficult to accurately obtain from benthic transects, stype counts for this species were undertaken along each transect line.

6.4.2.3 Invertebrates and cryptic fish

To survey invertebrates and cryptic fish, divers searched the seabed in a 1 m wide swathe along the transect line. This included visual assessment of all crevices and overhangs but not overturning boulders. Algae were swept away from the transect to obtain a clear view of the substratum. Mobile invertebrates (including rock lobsters, abalone, sea urchins, octopus etc.) were counted, as well as cryptic fishes (which were also estimated for size). We did not count invertebrates under rocks or < 2.5 cm due to dive time constraints. The maximum shell length of abalone and the carapace length of rock lobsters were measured underwater using Vernier callipers.

6.4.2.4 Fish

The abundance and estimated size-class of all fish species were recorded by divers within 5 m swathes either side of the transect line and 5 m above the substrate. Calibration of size estimates were based on comparison of observed fish lengths with a scale-bar on the slates carried by divers. Fish were categorised into fish length (size) classes as 25, 50, 75, 100, 125, 150, 200, 250, 300, 350, 375, 400, 500, 625, 750, 875 and 1000+ mm. Fish >1000 mm length were individually estimated. Care was taken not to record more than once any species that was obviously attracted to the divers (e.g *Notolabrus tetricus*, *Notolabrus fucicola*, *Scorpis lineolata*).

6.4.2.5 Data analysis

All UVC data were checked before analysis, and where species identification was uncertain, taxa were pooled with congeners or other similar species. Any cryptic fish identified by divers during invertebrate surveys were added to the fish identified during the fish surveys. This resulted in three distinct groups for analysis: Algae, Fish and Invertebrates. A combination of multivariate data visualisation techniques and hypothesis tests were performed using statistical software packages (PRIMER, PERMANOVA+ and R Studio). For each dataset (Fish, Invertebrates and Algae) a Bray-Curtis dissimilarity matrix was generated. Fish and invertebrate datasets were log-transformed ($\ln(x + 1)$) prior to generation of the dissimilarity matrix. A Principal Coordinates Ordination (PCO) analysis was undertaken to show the relationship between species and different sites within each region. Where appropriate, PERMANOVA, a multivariate permutational procedure, was also used to estimate components of variance for the algae, fish and invertebrate community data sets, with site and depth as a fixed factors.

The influence of exposure and reef rugosity on biodiversity was also examined. An exposure model using CAWCR Wave Hindcast (CSIRO/BOM) data series was used to estimate mean wave power (kW/m) at each site (Mummery 2016). Wave power values for the five years preceding the survey were averaged to provide a comparison in energy between sites. The influence of reef rugosity was examined by characterising both habitat type and cryptic habitat availability. As rugosity has been shown to most affect invertebrate communities, it was only applied to invertebrate data (Alexander et al. 2009; Alexander 2013). Five categories of habitat were scored for each 50 m transect (consolidated bedrock/sand/cobble, small boulders 0.2-0.5 m, medium boulders 0.5-1.5 m, large boulders >1.5 m, unscorable) following the conventions outlined in Alexander et al. (2013). A percentage estimate of observed cryptic habitat availability was determined for each transect, based on the following six categories (0%, 1-10%, 10-40%, 40-70%, 70%+, unscorable). Cryptic habitat was broadly defined as the number of cracks, crevices and ledges that could be observed (Alexander 2013).

6.4.3 Results

6.4.3.1 North Bruny Island

Overall, there was significant site level variation in algae, fish and invertebrate communities across the North Bruny region. Ordination plots showed very clear spatial differentiation in the algal data, and while clear site differences were also evident in the fish and invertebrate data, these were less defined over the longitudinal gradient (Figure 6.4-2). There was a reasonable level of within site fidelity in the fish communities, particularly in the fish communities with Cape Queen Elizabeth (CQE) standing out as having a more distinct community than elsewhere. Unfortunately, sampling at 10 m was insufficient to draw any definitive conclusions from this depth category alone. Therefore, the remainder of this section will focus on the 5 m contour, where there is sufficient spatial coverage to establish a regional baseline. Raw data for the 10 m contour can be found in Appendix 1.

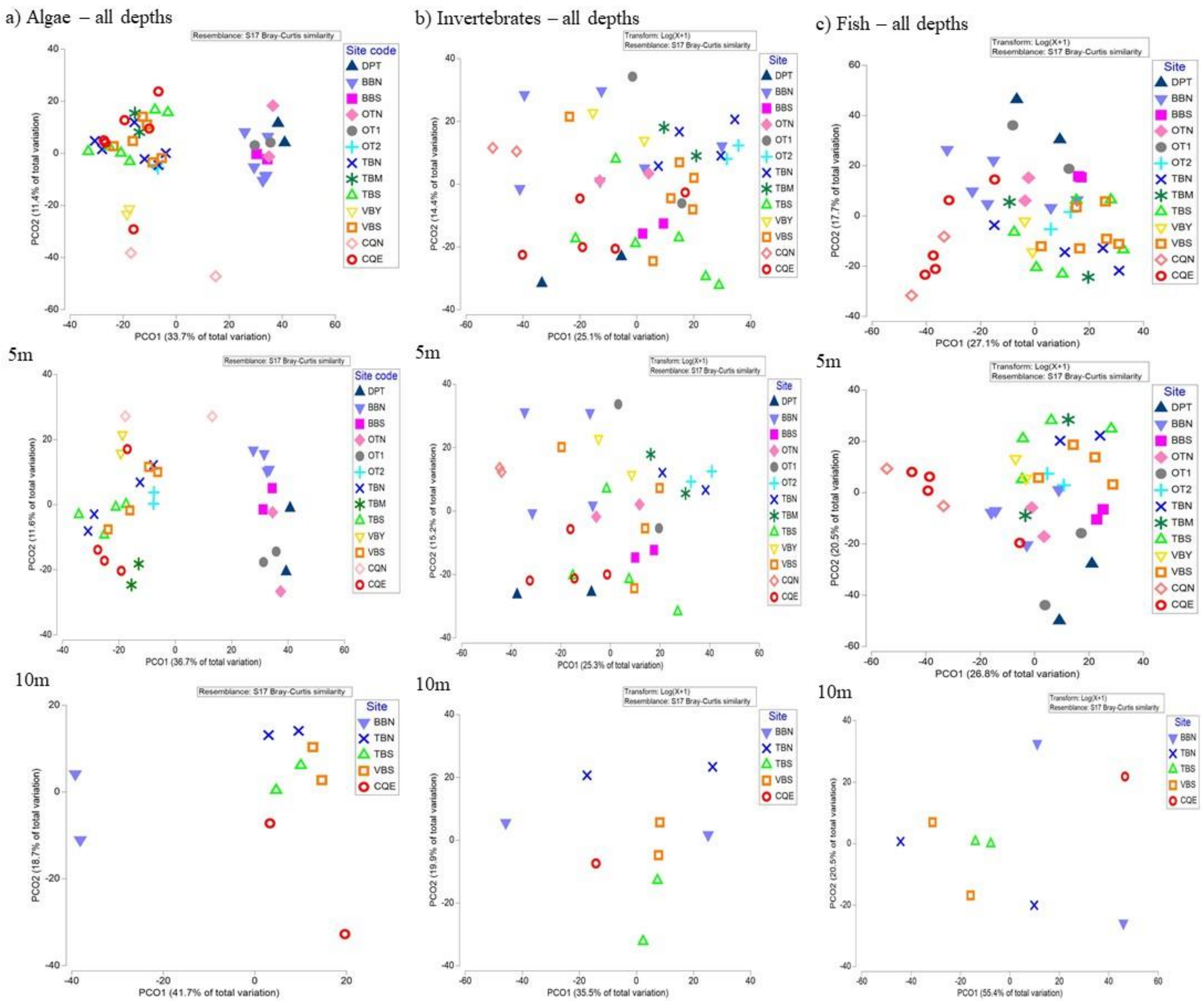

Figure 6.4-2 Principal Coordinate Analysis (PCO) on algal (% cover), fish and invertebrate communities (abundance) from 13 sites on North Bruny Island in March 2016.

6.4.3.1.1 Algae

There was a clear separation in algal species composition between northern and southern sites ($F_{(12, 33)}$ = 4.9385 , p = < 0.0001 (*MC*)), with the change apparently occurring around the One-Tree Point site (Figure 6.4-3, Table 6.4-2). Sites from OT2 south were dominated by the canopy forming species *Phyllosphora comosa* (crayweed), whereas *Ecklonia radiata* (common kelp) dominated the canopy from OTN north (Figure 6.4-3, Table 6.4-2). The canopy forming species *Durvillaea potatorum* (bull kelp) was more abundant at CQN and VBY than other sites, and was a distinguishing feature of these sites (Figure 6.4-3, Table 6.4-2). Total canopy cover was lower in the north (ranging from 13.4% at OT1 to 56.8% at BBS), compared to the south (ranging from 45.2% at OT2 to 90.2% at CQE; Table 6.4-2).

One-Tree Point was also the point at which the functionality of the system appeared to change. For example, northern sites had greater cover of the understory brown algae *Carpoglossum confluens* (2.4 - 36.6%) and understory red species *Callophyllis rangiferina* (1.2 - 6.0%) than sites in the south (0.0 – 11.2% and 0.0 – 0.2% respectively; Figure 6.4-3, Table 6.4-2). There was generally a higher percentage cover of brown algae understory across these northern sites, with a larger diversity of *Sargassum* and *Cystophora* species also observed (Table 6.4-2). While overall cover of red algae was relatively consistent between north and south, the composition differed, with the red algal taxa *Echinothamnion* and *Polluxfenia lobata* present in higher percentages in the northern sites (Table 6.4-2). Although individual algae species richness varied between sites, there was no evidence of any particular spatial trend for this parameter (Table 6.4-2).

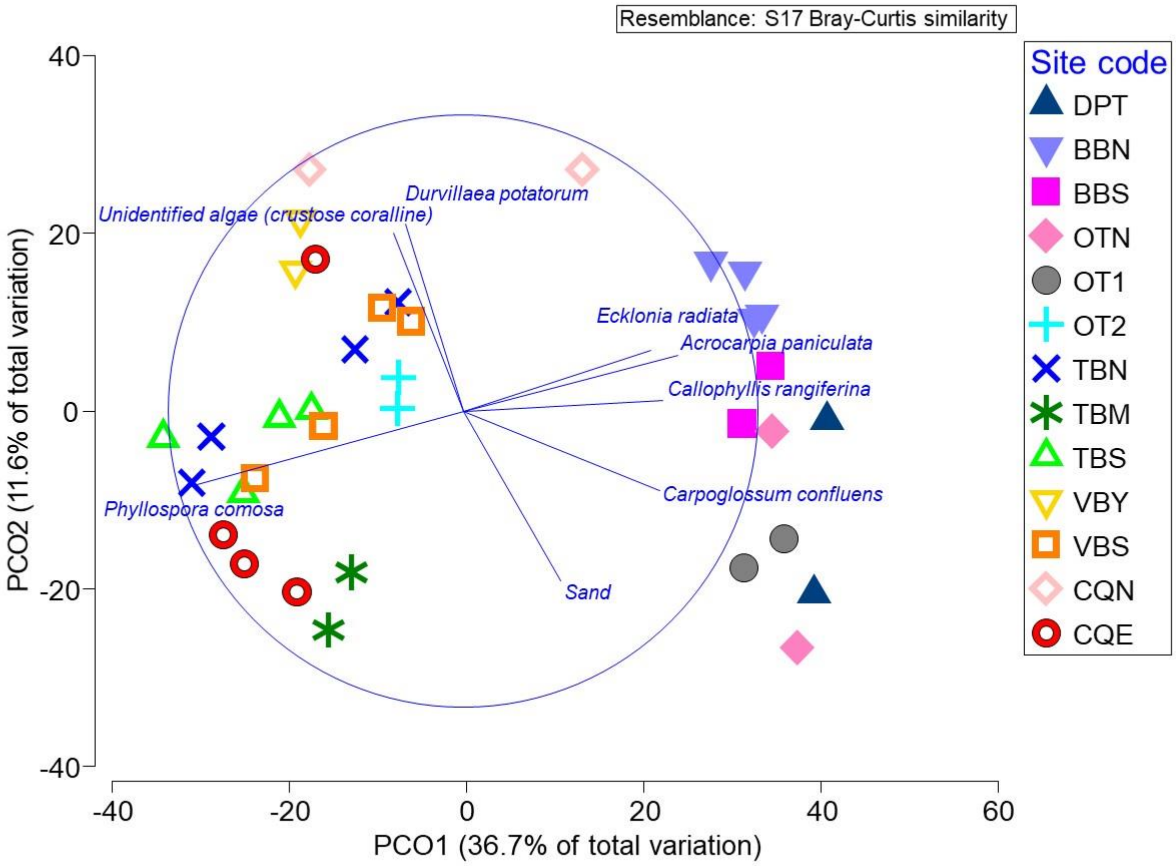

Figure 6.4-3 Principal Coordinate Analysis (PCO) on algae abundance at 5 m from 13 sites on North Bruny Island in March 2016. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates with a base variable comparison of > 0.6 included.

Table 6.4-2 Percentage cover of macroalgae and sessile invertebrate species from the North Bruny region averaged per 0.25 m^2 quadrat across site.

		Site												
Guild	**Species**	**DPT**	**BBN**	**BBS**	**OTN**	**OT1**	**OT2**	**TBN**	**TBM**	**TBS**	**VBY**	**VBS**	**CQN**	**CQE**
Canopy-forming algae	*Durvillaea potatorum*	0.0	0.0	0.0	0.0	0.0	0.0	4.5	0.0	0.0	18.6	0.0	23.2	1.7
	Ecklonia radiata	31.6	37.0	56.8	18.8	13.4	9.4	11.5	4.4	5.1	3.2	24.3	2.0	6.8
	Lessonia corrugata	0.0	3.5	0.0	1.2	0.0	0.0	3.0	0.0	0.0	2.0	0.0	2.2	0.0
	Macrocystis pyrifera	0.0	1.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Phyllospora comosa	0.0	0.0	0.0	0.0	0.0	35.8	70.1	77.0	71.9	44.2	54.5	20.6	81.7
	Canopy forming FG%	**31.6**	**41.5**	**56.8**	**22.4**	**13.4**	**45.2**	**89.1**	**81.4**	**77.0**	**68.0**	**78.8**	**48.0**	**90.2**
Understory brown algae	*Acrocarpia paniculata*	14.0	17.8	14.2	12.8	16.0	0.0	2.8	1.8	0.1	0.0	0.0	19.0	0.0
	Carpoglossum confluens	36.6	18.2	2.4	20.8	13.0	4.2	2.0	11.2	5.7	2.0	1.7	0.0	0.2
	Carpomitra costata	0.0	0.0	0.6	0.0	0.0	0.0	0.1	0.2	0.0	0.2	0.0	0.0	0.0
	Caulocystis uvifera	0.0	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Colpomenia sinuosa	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora moniliformis	0.0	1.0	5.2	0.0	0.6	0.0	0.0	2.8	0.0	0.0	0.0	0.0	0.0
	Cystophora platylobium	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	2.0	0.0	2.8	0.0
	Cystophora xiphocarpa	0.0	4.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Dictyopteris muelleri	0.6	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
	Dictyota spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4
	Dilophus marginatus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6
	Halopteris paniculata	0.0	3.1	1.2	0.0	4.6	0.0	0.0	0.0	0.4	0.0	0.0	0.4	0.0
	Lobophora variegata	2.0	0.2	1.2	1.2	1.0	0.4	0.6	0.6	2.5	0.0	1.5	0.0	0.1
	Perithalia caudata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.7	0.0	0.0
	Sargassum fallax	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum lacerifolium	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum spp.	0.8	0.5	0.2	1.2	2.0	0.0	1.2	1.8	0.0	0.0	0.0	0.0	0.0
	Sargassum verruculosum	9.2	0.0	0.0	1.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum vestitum	0.0	1.0	7.2	1.4	0.0	1.0	0.0	0.6	0.7	0.8	0.0	1.0	1.7
	Unidentified algae (filamentous brown)	17.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Xiphophora gladiata	0.0	0.4	0.0	0.0	1.6	0.4	2.0	7.8	0.0	0.0	0.0	6.2	0.0
	Zonaria turneriana/angustata	3.2	3.5	1.4	0.6	4.2	0.2	1.4	0.8	5.2	1.2	1.1	0.0	0.1
	Understory brown algae FG%	**84.0**	**50.3**	**33.6**	**39.2**	**48.8**	**6.2**	**10.3**	**27.6**	**14.6**	**6.2**	**7.6**	**29.4**	**3.1**

	Species	**DPT**	**BBN**	**BBS**	**OTN**	**OT1**	**OT2**	**TBN**	**TBM**	**TBS**	**VBY**	**VBS**	**CQN**	**CQE**
Understory green algae	*Bryopsis* spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
	Caulerpa flexilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0
	Caulerpa hodgkinsoniae	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa longifolia	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa scalpelliformis	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa simpliciuscula	0.0	0.6	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa spp. (rhizomes)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa trifaria	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Chaetomorpha coliformis	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
	Chaetomorpha spp.	0.0	0.0	0.0	0.4	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Cladophora spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
	Codium australicum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Codium dimorphum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Codium spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ulva spp.	0.0	0.0	0.0	0.2	1.4	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Understory green algae FG%		**0.6**	**1.5**	**0.2**	**0.6**	**1.8**	**0.0**	**0.0**	**0.0**	**0.0**	**0.4**	**1.6**	**0.0**	**0.0**
Understory red algae	*Areschougia* spp.	0.0	0.2	0.6	0.2	0.8	0.0	0.8	1.2	0.0	0.0	0.7	0.0	0.0
	Ballia callitricha	1.6	5.1	1.8	4.4	1.6	6.0	9.2	4.6	2.7	4.4	2.6	0.4	1.3
	Ballia scoparia	0.0	0.3	2.2	0.0	0.4	0.8	3.1	0.2	0.0	0.0	0.6	0.0	0.0
	Callophyllis rangiferina	1.2	4.1	6.0	4.4	1.2	0.2	0.1	0.0	0.0	0.2	0.2	0.0	0.0
	Callophyllis spp.	0.8	1.7	3.6	0.0	3.8	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0
	Champia spp.	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Chondria incrassata	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Corallina officinalis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.1	0.0	1.3
	Craspedocarpus ramentaceus	0.0	0.6	0.0	0.0	0.0	0.0	0.4	0.2	0.0	0.0	0.0	0.0	0.0
	Delisea plumosa	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	Delisea pulchra	0.0	0.0	0.0	0.0	0.0	2.2	0.0	0.0	0.0	0.6	0.0	0.0	0.0
	Dictyomenia harveyana	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
	Echinothamnion hystrix	0.4	0.2	0.0	3.0	1.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Echinothamnion spp.	14.4	2.1	7.8	1.2	3.6	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0
	Erythroclonium sonderi	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Euptilota articulata	0.6	1.2	0.6	0.2	0.0	2.6	0.0	1.0	0.0	0.2	0.0	0.0	0.0
	Gelidium asperum	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0

	Species	**DPT**	**BBN**	**BBS**	**OTN**	**OT1**	**OT2**	**TBN**	**TBM**	**TBS**	**VBY**	**VBS**	**CQN**	**CQE**
Understory red algae	*Gigartina* spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
	Gracilaria spp.	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Haliptilon roseum	0.4	0.9	0.0	0.0	0.0	0.0	0.3	0.4	0.0	0.6	0.1	8.8	10.4
	Hemineura frondosa	0.0	0.0	0.0	0.0	0.2	0.2	0.0	1.0	0.0	0.0	0.0	0.0	0.0
	Hypnea ramentacea	16.6	0.5	0.0	1.8	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Hypnea ramentacea epiphytic	0.0	2.0	0.0	0.0	0.0	0.0	0.0	7.4	0.0	0.0	0.0	0.0	0.0
	Laurencia spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Lenormandia marginata	3.0	1.5	0.4	4.4	3.2	3.2	2.6	1.2	0.0	0.0	0.7	0.4	0.4
	Mastophoropsis canaliculata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Nizymenia conferta	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Peyssonnelia novaehollandiae	0.4	0.1	0.8	0.0	1.2	0.0	0.9	0.4	2.8	1.6	1.0	0.0	0.8
	Phacelocarpus apodus	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Phacelocarpus peperocarpos	1.6	1.7	0.0	0.0	0.6	3.8	6.7	0.4	0.0	3.6	0.0	1.2	0.0
	Plocamium angustum	15.4	8.8	0.4	4.4	1.6	2.2	3.7	1.2	0.3	4.4	1.5	0.8	0.2
	Plocamium cartilagineum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.2	0.0
	Plocamium costatum	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
	Plocamium dilatatum	0.0	1.4	0.0	0.0	0.0	3.0	0.1	0.6	0.0	3.6	0.7	0.0	0.1
	Plocamium mertensii	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Plocamium patagiatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
	Plocamium spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Polluxfenia lobata	7.6	5.4	1.0	3.4	5.8	0.0	0.0	2.6	0.0	0.0	0.0	0.0	0.0
	Polyopes constrictus	0.0	4.7	0.0	6.0	0.6	6.4	1.2	4.6	2.7	3.8	0.9	17.0	0.6
	Polysiphonia spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ptilonia australasica	0.8	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0
	Rhodymenia prolificans	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	0.0	0.8	0.0	0.0	0.0
	Rhodymenia spp.	2.0	2.2	1.8	1.8	1.8	3.4	8.7	3.8	2.5	3.8	3.2	0.4	5.3
	Sonderopelta coriacea	0.0	0.1	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0	1.4	0.0	0.0
	Sonderopelta/Peyssonnelia	0.0	0.0	1.2	0.0	0.0	0.0	0.0	3.2	0.0	0.0	0.0	0.0	0.0
	Stenogramme interrupta	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.8	0.0	0.0	0.3	0.0	0.2
	Synarthrophyton patena	0.0	0.1	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Thamnoclonium dichotomum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0
	Unidentified algae (filamentous red)	10.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (foliose red)	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	Species	DPT	BBN	BBS	OTN	OT1	OT2	TBN	TBM	TBS	VBY	VBS	CQN	CQE
	Unidentified algae (red)	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Understory red algae FG%	**77.8**	**45.3**	**28.2**	**36.0**	**39.4**	**37.6**	**38.4**	**38.4**	**11.2**	**28.0**	**14.2**	**30.0**	**20.7**
Encrusting algae	*Peyssonnelia* spp. (encrusting)	22.8	11.2	13.6	8.8	7.8	8.2	7.6	8.4	14.4	10.0	11.7	19.6	6.7
	Unidentified algae (crustose coralline)	7.4	29.8	13.6	5.2	13.8	15.4	22.5	14.6	29.3	28.2	32.2	24.4	12.4
	Encrusting algae FG%	**30.2**	**41.0**	**27.2**	**14.0**	**21.6**	**23.6**	**30.1**	**23.0**	**43.7**	**38.2**	**43.9**	**44.0**	**19.1**
Sessile invertebrates	*Amathia wilsoni*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Corynactis australis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.4
	Culicia spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	2.6	0.0	0.0	0.0
	Erythropodium hicksoni	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Galeolaria spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Maoricolpus roseus	4.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Mytilus galloprovincialis	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ostrea angasi	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Pyura australis	0.2	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0
	Pyura gibbosa	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
	Pyura stolonifera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
	Unidentified ascidians	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
	Unidentified ascidians (encrusting)	0.0	0.4	0.0	0.0	0.6	1.0	0.0	0.0	0.4	0.0	0.1	0.0	0.0
	Unidentified bryozoans	0.0	0.8	0.0	0.0	0.0	0.0	5.4	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified bryozoans (hard)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.5	0.0	1.4	0.6	0.2
	Unidentified bryozoans (soft)	1.6	3.5	1.6	2.6	1.0	4.2	11.7	3.2	5.2	10.6	5.5	1.2	1.4
	Unidentified epiphytic bryozoans	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified hydroid	0.0	0.3	0.0	0.8	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.2
	Unidentified invertebrates (encrusting)	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0	9.8	17.2	7.3	1.2	0.2
	Unidentified sponge (encrusting)	25.2	8.2	8.0	10.6	3.8	8.4	7.1	3.6	3.9	4.0	6.6	2.6	13.3
	Unidentified sponges	2.4	1.2	1.2	1.8	0.0	1.4	0.3	0.4	1.8	0.2	1.2	0.4	0.4
	Sessile invertebrates FG%	**34.2**	**15.4**	**10.8**	**17.6**	**5.4**	**18.0**	**24.9**	**7.6**	**25.3**	**35.4**	**22.1**	**6.0**	**16.1**
Substrate	Bare rock (non - barrens)	0.0	2.3	0.0	1.6	2.0	3.6	9.0	1.2	1.9	7.2	3.7	10.6	1.7
	Heterozostera nigricaulis	0.0	0.0	0.2	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sand	9.4	3.2	2.8	25.6	19.4	6.0	0.4	17.4	5.2	0.0	7.8	0.0	0.7
	Turf/sand/sediment matrix	3.8	0.2	7.2	7.0	10.2	4.6	0.0	7.8	0.0	3.6	0.0	1.4	9.3
	Unidentified algae (brown turf)	10.6	3.4	3.2	2.4	4.4	35.2	0.0	0.4	0.0	14.6	1.2	6.0	1.8
	Unidentified algae (green turf)	0.0	0.0	2.2	0.4	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5

	Species	DPT	BBN	BBS	OTN	OT1	OT2	TBN	TBM	TBS	VBY	VBS	CQN	CQE
Substrate	Unidentified algae (red turf)	0.8	4.1	7.2	8.4	4.8	3.0	4.9	2.6	4.1	2.4	2.6	18.4	15.2
	Unidentified algae (turf)	0.0	5.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Substrate FG%		**24.6**	**18.5**	**22.8**	**45.4**	**43.8**	**52.4**	**14.3**	**29.4**	**11.2**	**27.8**	**15.3**	**36.4**	**29.2**
Algal species richness		**38**	**34**	**29**	**17**	**21**	**26**	**25**	**20**	**13**	**28**	**22**	**30**	**18**

NB: FG refers to Functional Group with the%FG being the summed percentage of all taxa within that group. Algal species richness is the sum of all algal species (only) found at any given site.

Stype counts for *Macrocystis pyrifera* were relatively low in the North Bruny region. Only BBN and OT2 recorded the presence of this species, and only very low numbers, i.e. 0.25 ± 0.13 and 4 ± 2.8 stypes/50 m transect respectively (Table 6.4-2, Figure 6.4-4). It is worth noting that data obtained through quadrat analysis and stype counts along the transects were relatively well-aligned for this region.

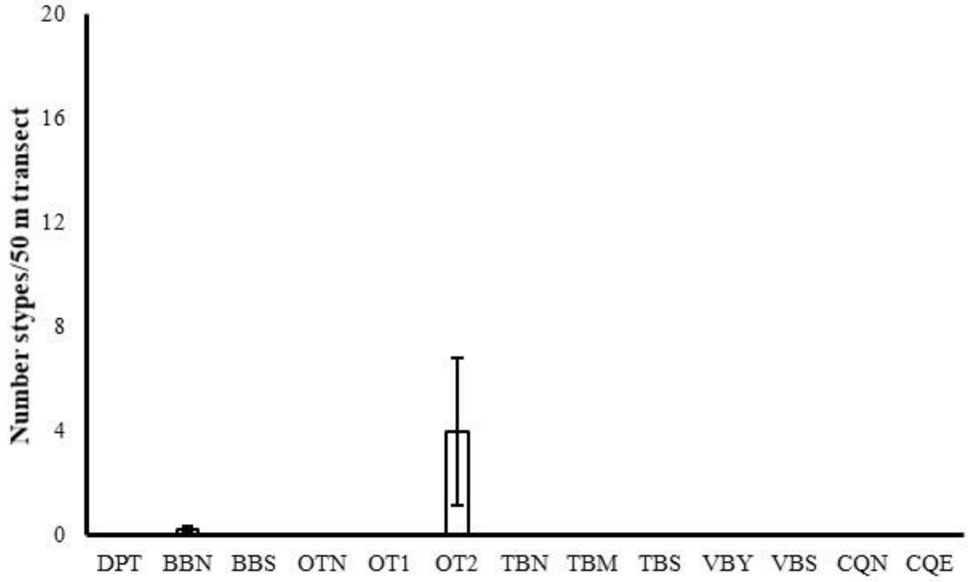

Figure 6.4-4 Average number of *Macrocystis pyrifera* stypes recorded per 50 m transect at each of the sites in the North Bruny region.

Overlaying the wave power (kW/m) data on the community ordination plots shows how well these features align with the algal community structure on the northern and southern North Bruny sites (Figure 6.4-5). Sites south of One Tree Point were relatively exposed, with the mean wave power >2.4 kW/m across a five-year period (Figure 6.4-5). The species community composition at these sites included species better adapted for higher exposure such as the canopy forming species *Phyllosphora comosa* (crayweed) and at CQN, which tended to be separate from the other site groupings due to the abundance of *Durvillaea potatorum*. The northern sites dominated by *Ecklonia radiata* had much lower mean wave power (⩽0.6 kW/m) (Figure 6.4-5).

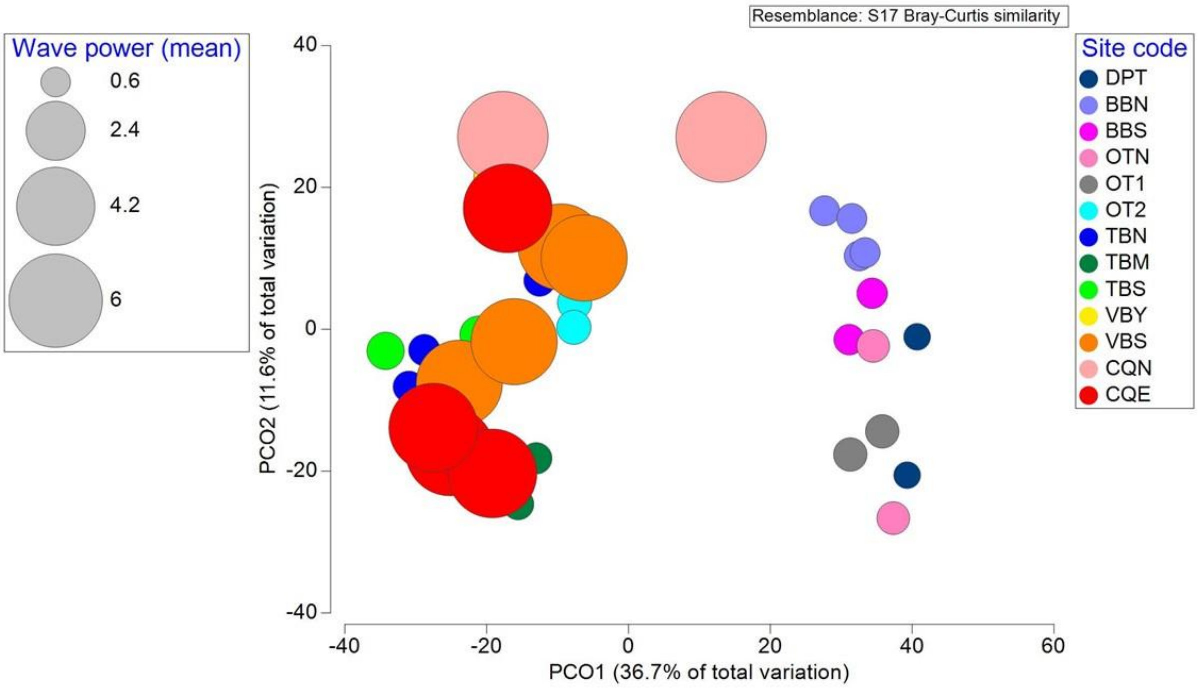

Figure 6.4-5 Principal Coordinate Analysis (PCO) on algae community data (5 m) with mean wave power (5-year average) overlain at 13 sites on North Bruny Island.

6.4.3.1.2 Invertebrates

The invertebrate communities also varied significantly between sites ($F_{(12, 30)} = 2.4311$, $p = < 0.0001$ (*MC*)). Unlike the algal data, this appeared to relate to individual site conditions, with no clear site clusters reflecting obvious spatial gradients in the data (Figure 6.4-6). The featherstar *Comanthus trichoptera* was clearly the dominant invertebrate species, and as a result, changes in the abundance of this species appeared to strongly influence the site community differences (Figure 6.4-6). Sites that recorded high abundances of *C. trichoptera*, such as VBS (124 individuals/50 m transect), TBN (105 individuals/50 m transect), OT2 (153 individuals/50 m transect) and BBS (426 individuals/50 m transect) were clearly distinguished along PCO1 (Table 6.4-3, Figure 6.4-6).

Blacklip abalone (*Haliotis rubra*), the purple urchin (*Heliocidaris erythrogramma*) and the New Zealand screwshell (*Maoricolpus roseus*) were also present in high abundances at particular sites (Figure 6.4-6, Table 6.4-3). For example, average black-lip abalone abundances at CQE were more than double that observed at all other sites (21.3 individuals/50 m transect, compared to <11.3 individuals/50 m transect), whereas TBS and BBS recorded particularly high abundances of purple urchin (68.5 and 36.5 individuals/50 m transect respectively; Table 6.4-4). Though the New Zealand screwshell was recorded at only two sites but was in high abundance at DPT (50.0 individuals/50 m transect) potentially differentiating this site on that basis.

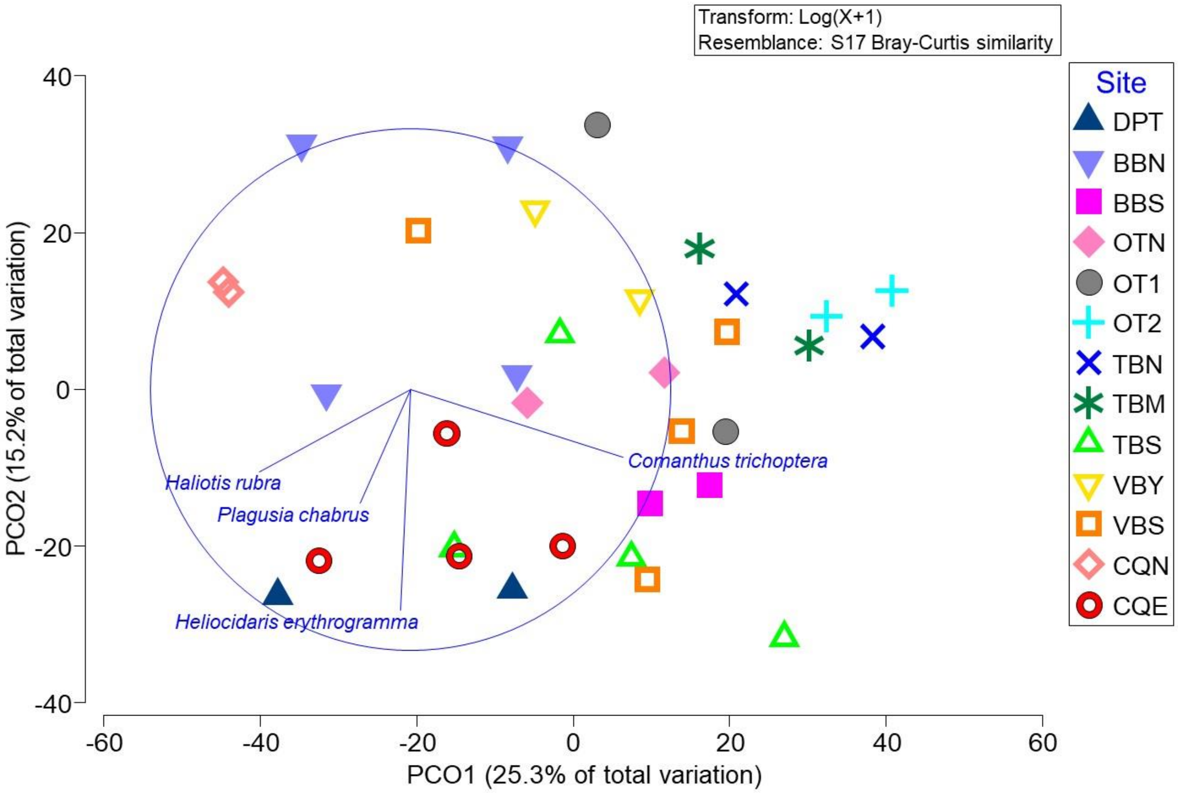

Figure 6.4-6 Principal Coordinate Analysis (PCO) on invertebrate abundance at 5 m from 13 sites on North Bruny Island in March 2016. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates with a base variable comparison of > 0.4 included.

Table 6.4-3 Abundance of invertebrate species (no individuals/50 m transect) from sites in the North Bruny region surveyed in March 2016.

	Site												
Species	**DPT**	**BBN**	**BBS**	**OTN**	**OT1**	**OT2**	**TBN**	**TBM**	**TBS**	**VBY**	**VBS**	**CQN**	**CQE**
Agnewia tritoniformis (Murex shell)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amblypneustes ovum (Short-spined urchin)	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amoria undulata (Wavy volute)	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0
Australostichopus mollis (Sea cucumber)	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.5	0.3	31.0	0.5
Cabestana spengleri (Triton shell)	0.0	0.5	0.0	0.5	0.5	0.5	0.0	0.0	0.3	0.0	1.0	0.0	0.0
Centrostephanus rodgersii (Long-spined urchin)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Comanthus tasmaniae (Feather star)	0.0	6.3	5.0	1.5	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.8
Comanthus trichoptera (Feather star)	0.5	11.0	425.5	33.0	49.5	153.0	105.0	44.0	51.3	53.0	123.5	0.0	18.5
Coscinasterias muricate (Eleven-arm star)	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Dicathais orbita (Dog whelk)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.3
Fromia polypore (Many-spotted seastar)	0.0	0.8	0.0	0.0	0.0	0.0	0.5	0.0	0.3	2.0	0.0	0.0	0.0
Goniocidaris tubaria (Pencil urchin)	0.0	0.3	4.5	2.5	1.0	0.0	0.0	0.0	0.3	1.0	0.3	0.0	0.0
Haliotis rubra (Black-lip abalone)	3.5	5.0	1.5	3.5	1.0	0.5	0.0	1.5	11.3	4.0	6.3	2.0	21.3
Heliocidaris erythrogramma (Purple urchin)	6.0	3.3	36.5	2.0	1.0	0.0	0.0	0.0	68.3	0.0	8.5	1.0	13.3
Jasus edwardsii (Southern rock lobster)	0.0	3.5	7.0	3.0	1.0	0.0	9.0	1.5	5.8	0.0	1.8	0.0	2.5
Maoricolpus roseus (New Zealand screw shell)	50.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nectocarcinus integrifrons (Red swimmer crab)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Nectocarcinus tuberculosus (Velvet crab)	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nectria ocellate (Ocellate seastar)	0.5	0.0	0.5	0.0	0.0	0.0	1.0	1.5	1.8	7.5	0.3	0.0	0.0
Octopus maorum (Maori octopus)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.3
Ostrea angasi (Native oyster)	6.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pagurid spp. (Hermit crab)	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.5	0.0	0.0	0.0
Paragrapsus laevis (Shore crab)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Patiriella regularis (Regular seastar)	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Penion maximus (Giant whelk)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Penion spp. (Whelk)	2.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Petricia vernicina (Velvet seastar)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Phasianotrochus eximius (Giant kelp shell)	0.0	0.0	1.0	2.5	0.5	1.0	2.0	2.5	1.0	13.5	0.0	0.0	0.5
Plagusia chabrus (Red bait crab)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0
Pleuroploca australasia (Tulip shell)	0.0	1.5	1.0	4.0	0.5	0.0	1.5	1.5	3.0	0.5	2.0	0.0	10.0
Ranella australasia (Australian Rock Whelk)	3.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Sassia parkinsonia (Trumpet shell)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Strigopagurus strigimanus (Rasping hermit crab)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Tosia australis (Southern biscuit star)	0.0	0.3	0.5	0.0	0.0	0.0	1.0	0.5	0.0	0.5	0.0	0.0	1.3
Tosia magnifica (Magnificent biscuit star)	0.0	0.5	0.0	0.0	0.5	0.5	0.0	3.0	0.3	2.0	0.0	0.0	0.3
Turbo undulatas (Turban shell)	0.0	0.0	0.0	2.5	0.0	0.0	0.5	0.0	0.0	0.5	0.0	0.0	0.0
Unidentified mollusc	0.0	4.5	3.0	0.0	0.0	0.5	0.0	0.0	0.3	0.0	0.5	0.0	5.3
Uniophora granifera (Granular seastar)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.3	0.0	0.0	0.0	0.0
Total abundance	**72.5**	**37.25**	**487.5**	**57.5**	**57**	**157.5**	**120.5**	**58.5**	**144**	**87**	**144.5**	**36**	**76**
Species Richness	**8**	**12**	**14**	**13**	**11**	**6**	**8**	**10**	**13**	**12**	**11**	**5**	**16**

NB: Total abundance refers to the total number of individuals on each transect (transect mean per site) and species richness refers to the sum of species recorded on each transect.

Exposure appeared to have little influence in shaping invertebrate communities, with no clear correlation between the community structure/ biodiversity and mean wave power (kW/m; Figure 6.4-7). Similarly, there was no clear pattern between species assemblage and site location overlain with reef rugosity and cryptic habitat at sites in the North Bruny region (Appendix 2).

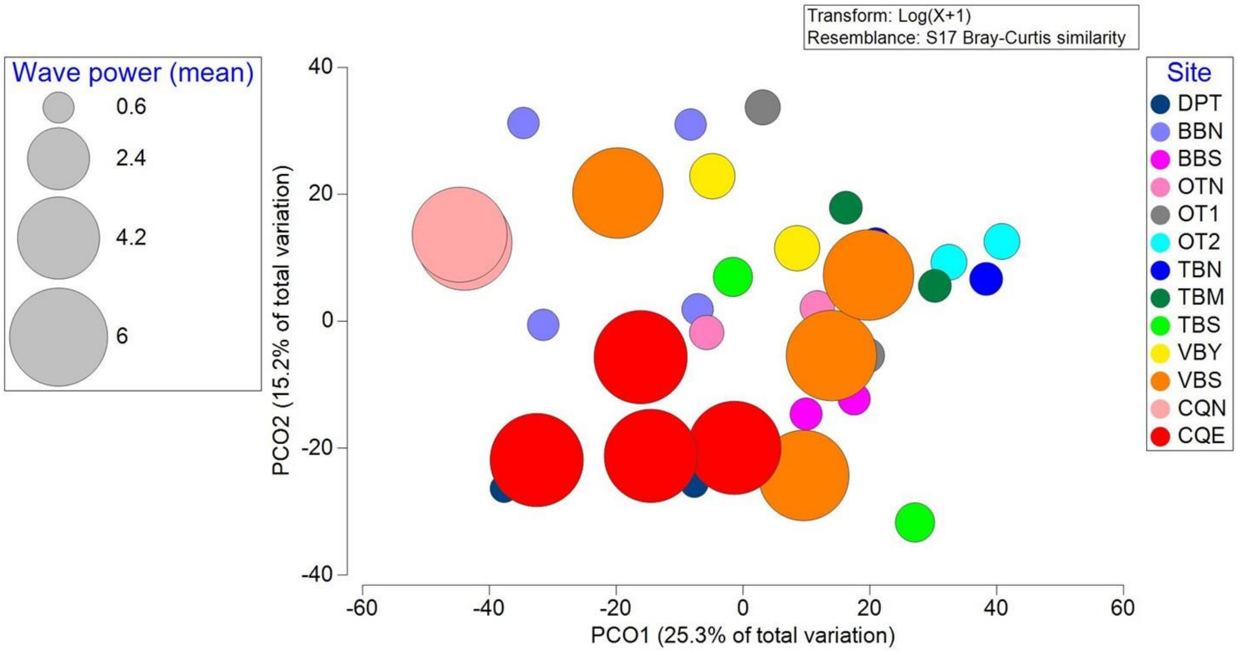

Figure 6.4-7 Principal Coordinate Analysis (PCO) on invertebrate community data (5 m) with mean wave power (5-year average) overlain at 13 sites on North Bruny Island.

6.4.3.1.3 Fish

There were significant differences in the fish communities at the North Bruny sites ($F_{(12, 30)}$ = 2.9132 , p = < 0.0001(*MC*)). Ordination plots suggest that the fish communities were different at the sites in the far south (CQE and CQN) (Figure 6.4-8), with higher abundances of purple wrasse (*Notolabrus fucicola*), and lower abundances of blue-throat wrasse (*Notolabrus tetricus*) and toothbrush leatherjacket (*Acanthaluteres vittiger*) generally recorded at these sites (Figure 6.4-8, Table 6.4-4). Conspicuously, hulafish (*Trachinops caudimactulatus*) were not observed at either of these sites (Table 6.4-4).

Northern sites, particularly those north of OT1, tended to have much higher numbers of blue-throat wrasse (11.5-19.5 individuals/50 m transect) compared to sites from OT2 south (3.3-12.0 individuals/50 m transect). Large numbers of hulafish characterised the mid-North Bruny sites, from VBS in the south (521.5 individuals/50 m transect) to OT2 in the north 252.5 individuals/50 m transect), with BBS also recording very high numbers of this species (332.5 individuals/50 m transect; Figure 6.4-8, Table 6.4-4). Toothbrush leatherjacket showed a very similar pattern of response to hulafish across sites (Figure 6.4-8, Table 6.4-4).

Total abundance of fish per 50 m transect was lowest at CQE (84.0 individuals/50 m transect); however, this site recorded the highest species richness value (24 species/50 m transect) (Table 6.4-4). Overall, total fish abundance and species richness were highly variable between sites and whilst there was a level of site specificity for some sites there was no clear pattern of response in either feature that could be related to spatial differences from north to south (Table 6.4-4).

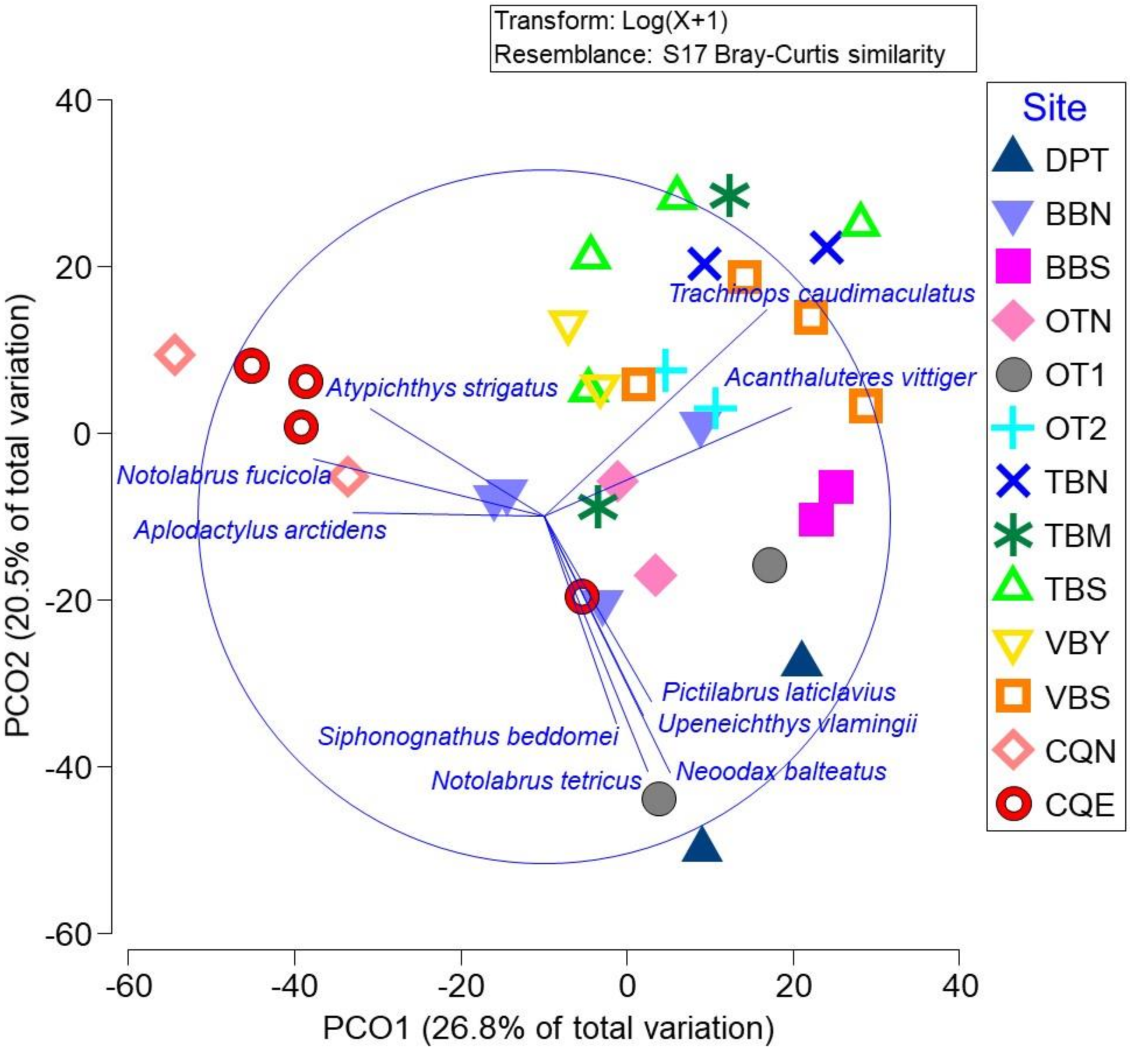

Figure 6.4-8 Principal Coordinate Analysis (PCO) on fish abundance at 5 m from 13 sites on North Bruny Island in March 2016. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates with a base variable comparison of > 0.6 included.

Table 6.4-4 Abundance of fish species (no individuals/50 m transect) from sites in the North Bruny region surveyed in March 2016.

	Site												
Species	**DPT**	**BBN**	**BBS**	**OTN**	**OT1**	**OT2**	**TBN**	**TBM**	**TBS**	**VBY**	**VBS**	**CQN**	**CQE**
Acanthaluteres vittiger (Toothbrush leatherjacket)	83.5	70.5	400.5	22.0	21.0	29.5	446.0	178.0	221.5	8.0	631.8	1.0	17.3
Aplodactylus arctidens (Marblefish)	0.0	0.3	0.0	1.0	0.0	0.5	0.0	0.0	0.0	1.0	0.0	2.0	0.8
Aracana aurita (Shaw’s cowfish)	1.0	0.3	0.0	0.0	0.5	0.5	0.0	0.0	0.0	0.5	0.8	0.0	0.3
Atypichthys strigatus (Mado sweep)	0.0	36.5	0.0	50.5	0.0	13.5	5.5	0.5	17.3	15.0	0.5	27.5	14.5
Caesioperca lepidoptera (Butterfly perch)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caesioperca razor (Barber perch)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cephaloscyllium laticeps (Draughboard shark)	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.5	0.0	0.0	0.3
Cheilodactylus spectabilis (Banded morwong)	0.0	1.3	0.0	1.0	1.0	0.0	0.0	2.0	1.0	0.0	0.0	0.5	0.8
Conger verreauxi (Southern conger)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Dasyatis brevicaudata (Smooth stingray)	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dinolestes lewini (Long-fin pike)	0.0	5.0	0.0	0.5	0.0	4.5	43.0	1.0	0.5	2.5	1.5	26.0	2.3
Diodon nicthemerus (Globe fish)	0.0	0.3	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
Dotalabrus aurantiacus (Castelnaus wrasse)	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Forsterygion varium (Variable threefin)	0.5	3.3	2.5	1.5	0.5	12.0	6.0	1.5	6.0	1.5	1.0	0.5	1.0
Gnathanacanthus goetzeei (Red velvetfish)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Helicolenus percoides (Reef ocean perch)	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heteroclinus perspicillatus (Common weedfish)	1.5	0.0	0.0	0.0	3.5	0.5	0.0	0.0	0.0	0.0	0.0	1.0	0.3
Heteroclinus tristis (Longnose weedfish)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0
Heteroscarus acroptilus (Rainbow cale)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Latridopsis forsteri (Bastard trumpeter)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.5	0.0	0.5	0.0
Lotella rhacina (Beardie)	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.3	0.0	1.0
Meuschenia australis (Brown-striped leatherjacket)	0.0	0.3	0.5	0.0	1.5	4.0	0.0	1.5	0.3	0.5	0.3	1.0	2.8
Meuschenia freycineti (Six-spine leatherjacket)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	1.0	0.0	0.0
Neoodax balteatus (Little rock-whiting)	26.5	0.5	7.0	3.5	4.5	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.3
Notolabrus fucicola (Purple wrasse)	1.0	6.8	3.0	2.5	1.5	3.0	0.5	6.0	4.3	4.5	5.3	23.0	8.0
Notolabrus tetricus (Blue-thorat wrasse)	12.5	14.3	13.5	11.5	19.5	5.0	3.5	6.5	1.0	5.0	3.5	3.5	3.0

Olisthops cyanomelas (Herring cale)	0.5	1.0	0.0	0.5	0.0	2.5	0.0	0.5	0.8	0.5	0.3	1.5	1.0
Parequula melbournensis (Silverbelly)	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Parma microlepis (White-ear)	0.0	0.0	0.5	1.5	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.8
Parma victoriae (Victorian scalyfin)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
Pempheris multiradiata (Common bullseye)	0.0	80.0	201.5	60.0	0.0	0.0	13.0	0.0	0.0	0.0	0.0	1.0	26.8
Pentaceropsis recurvirostris (Long-snouted boarfish)	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
Pictilabrus laticlavius (Senator wrasse)	3.0	1.5	3.0	1.0	1.0	1.0	0.0	0.0	0.8	0.5	0.0	0.0	0.3
Pseudocaranx dentex (White trevally)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudolabrus rubicundus (Rosy wrasse)	0.0	0.0	1.5	1.5	1.0	0.0	0.5	0.5	0.5	0.0	0.0	0.0	0.8
Scorpaena papillosa (Southern Rock Cod)	0.0	0.3	0.0	0.5	0.5	2.5	0.0	0.0	0.0	1.0	0.0	0.0	0.0
Scorpis aequipinnis (Sea sweep)	0.0	1.5	0.0	0.0	0.0	2.5	0.0	0.0	5.5	3.0	0.0	7.0	0.5
Scorpis lineolate (Silver sweep)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.0	0.0
Siphonognathus beddomei (Pencil weed whiting)	14.0	14.5	5.0	5.0	18.0	18.0	0.0	0.0	0.3	10.5	0.5	0.0	1.0
Tasmanoglobius gloveri (Glover’s Tasman goby)	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trachinops caudimaculatus (Hulafish)	1.5	26.3	332.5	15.0	60.0	252.5	495.0	262.5	713.8	155.0	521.5	0.0	0.0
Trinorfolkia clarkei (Common triplefin)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.0	0.0
Upeneichthys vlamingii (Southern goatfish)	10.0	0.0	1.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
Total Abundance	**155.5**	**264.2**	**972.5**	**180.5**	**137.0**	**354.0**	**1013**	**462.0**	**974**	**216.0**	**1171**	**101.0**	**84**
Species Richness	**12**	**20**	**14**	**19**	**17**	**19**	**9**	**12**	**16**	**21**	**17**	**15**	**24**

NB: Total abundance refers to the total number of individuals on each transect (transect mean per site) and species richness refers to the sum of species recorded on each transect.

Wave power (kW/m) did appear to have some influence on the fish communities across the North Bruny region (Figure 6.4-9). At sites where mean wave power and height were greater, *N. fucicola* tended to dominate assemblages, but this species was either not observed or present in lower numbers at sites with lower exposure (Table 6.4-4). However, this relationship was not observed at VBS which, although having similar exposure levels to CQE and CQN, had much lower numbers of *N. fucicola*, suggesting that there are other factors which may be driving the distribution of this species (Table 6.4-4).

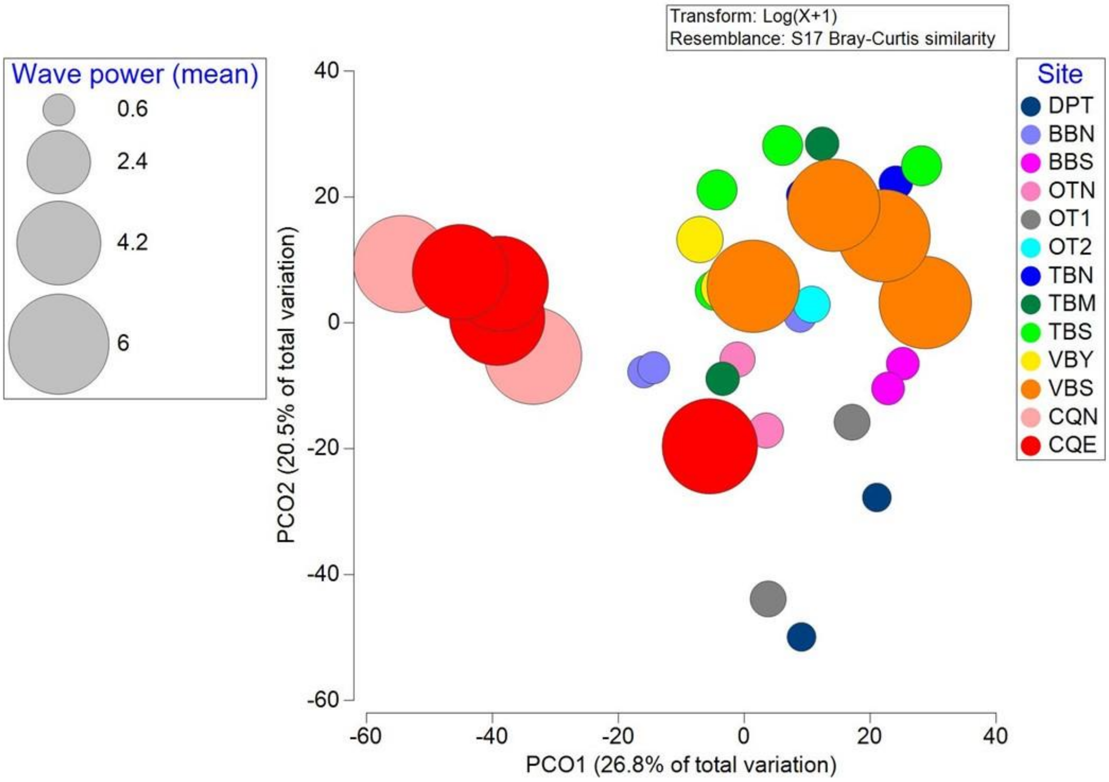

Figure 6.4-9 Principal Coordinate Analysis (PCO) on fish community data (5 m) with mean wave power (5-year average) overlain at 13 sites on North Bruny Island.

6.4.3.2 SE Channel

As in the North Bruny region, there was significant site level variation in algae, fish, and invertebrate communities across the SE Channel region. Ordination plots showed a clear spatial response pattern in the algal data with sites in the far south clearly differentiated from sites further north (Figure 6.4-10). Although there were site differences in the fish and invertebrate data these were not well defined over a longitudinal gradient (Figure 6.4-10). While surveys were undertaken at both 5 m and 10 m, due to a lack of sampling effort at 10 m limited conclusions can be drawn from this data and as such this analysis has focused on the 5 m data.

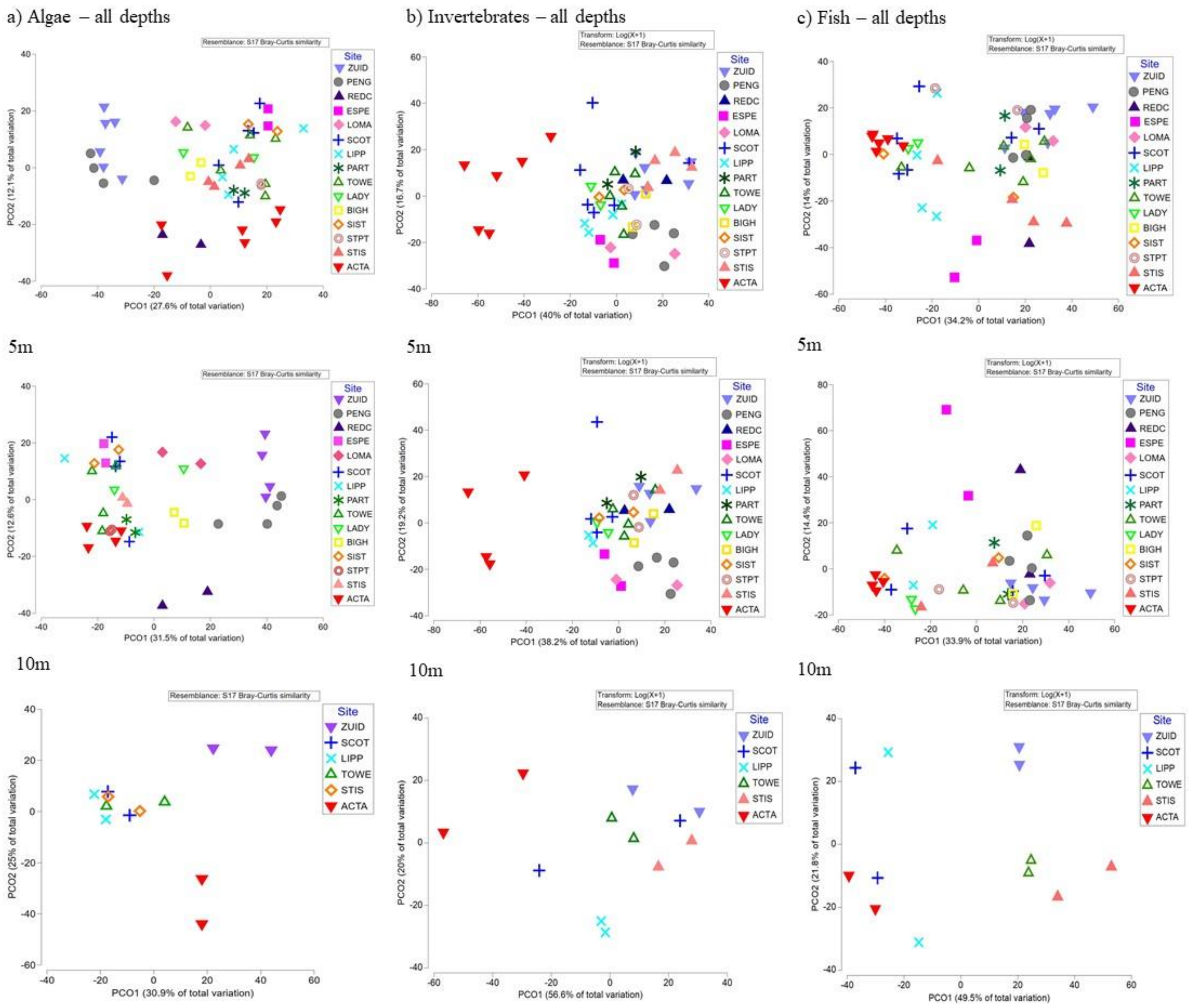

Figure 6.4-10 Principal Coordinates Analysis (PCO) on algal (% cover), fish and invertebrate communities (abundance) from 15 sites in the SE Channel region in February 2017.

6.4.3.2.1 Algae

Algal communities varied between sites ($F_{(14, 37)} = 3.5838$, $p = < 0.0001$ (*MC*)) in the SE Channel region. Ordination plots indicated two broad groupings within the region, differentiated by the dominant canopy forming species (Figure 6.4-11). Algal composition on transects from ZUID and PENG in the north appear to be more similar to each other than all other sites, with the canopy at these sites dominated by *E. radiata* (Figure 6.4-11). There were also differences in the sub-canopy at these sites, particularly red algal species with relatively high percentage cover of *Callophyllis rangiferina* (12.8% and 7.7% average cover respectively) and *Polluxfenia lobata* (5.8% and 1.9% respectively; Table 6.4-5). These two sites also included species that were absent or recorded in very low abundance from all other sites, such as the brown algae *Sargassum fallax* and the red algae *Myriogramme gunniana* and *Kallymenia* sp. (Figure 6.4-11, Table 6.4-5).

With the exception of REDC, all remaining sites recorded a higher percentage cover of the canopy forming species *Phyllosphora comosa* (crayweed), ranging between 32.2% cover at LOMA to 81.8% at cover ESPE (Figure 6.4-11, Table 6.4-5). The response of *P. comosa* was closely aligned with the vector responses for crustose coralline algae, such that a higher percentage cover for all these taxa was likely where *P. comosa* was dominant (Figure 6.4-11).

In terms of species composition, REDC is relatively distinct; whilst both *E. radiata* (10.6% average cover) and *P. comosa* (15.0% average cover) were important contributors to canopy at this site, cover of *Lessonia corrugata* was also significant (28.0% average cover) and only abundant at this site within the SE Channel (Table 6.4-5).

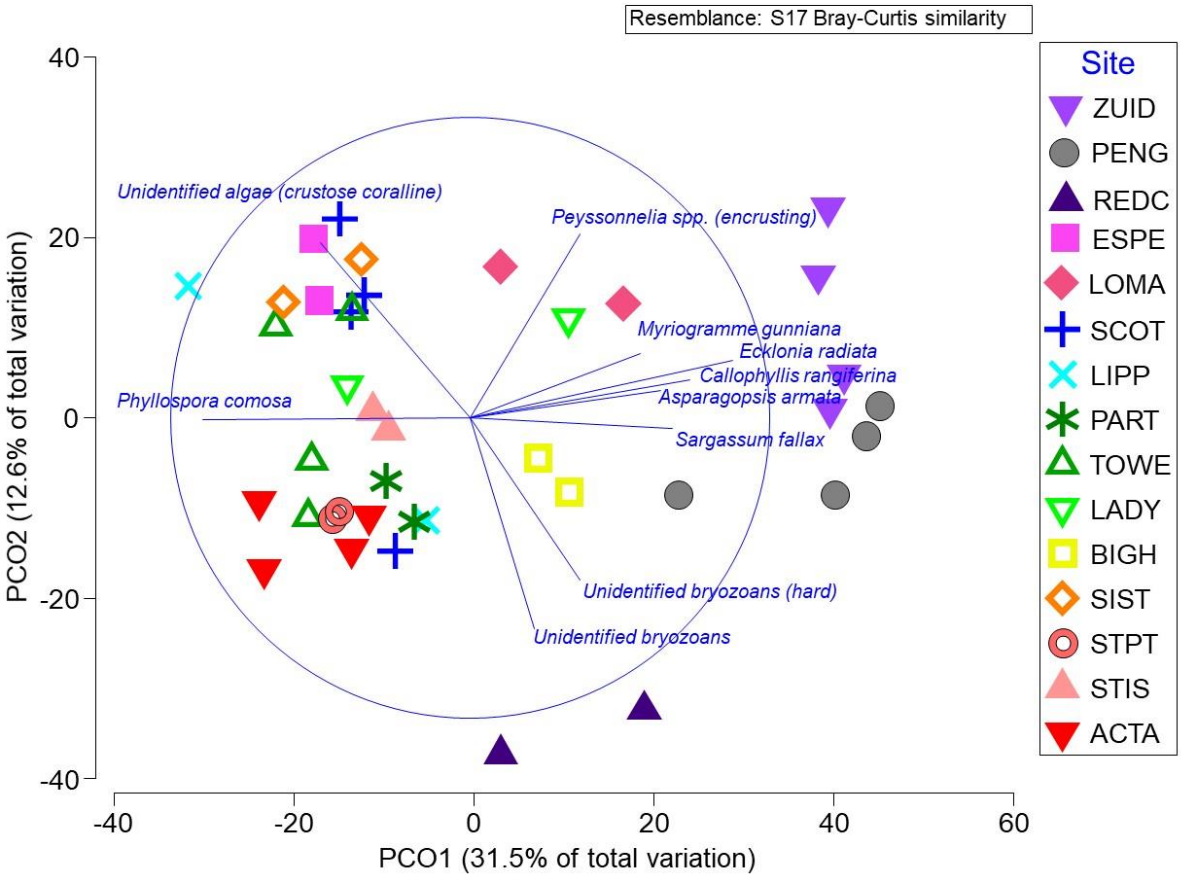

Figure 6.4-11 Principal Coordinate Analysis (PCO) on algae abundance at 5 m from 15 sites in the SE Channel region in February 2017. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates, with a base variable comparison of > 0.6 included.

In terms of functional groupings, ACTA had the highest abundance of understorey red algae across the region (78.0% average cover), with particularly high cover of *Plocamium dilatatum* and *Plocamium angustatum* (32.8% and 10.1% average cover respectively; Table 6.4-5). Regarding algal species richness, several sites had particularly high numbers of species, including TOWE (46 species/0.25 m^2) and SCOT (41 species/0.25 m^2; Table 6.4-5). In general, algal species richness was higher in the SE Channel compared to the North Bruny region.

Table 6.4-5 Percentage cover of macroalgae and sessile invertebrate species from the SE Channel region averaged per 0.25 m^2 quadrat across site.

		Site														
Guild	**Species**	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
Canopy-forming algae	*Durvillaea potatorum*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.8
	Ecklonia radiata	58.8	45.4	10.6	8.4	15.2	14.2	15.0	8.5	2.4	11.8	12.2	33.6	6.4	14.6	0.0
	Lessonia corrugata	0.0	0.0	28.0	0.0	0.0	0.0	6.0	5.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0
	Macrocystis pyrifera	0.0	2.4	0.8	0.0	2.0	0.0	0.0	0.7	0.0	0.0	2.0	0.0	0.0	0.0	3.4
	Phyllospora comosa	0.0	2.0	15.0	76.2	32.2	55.2	61.8	69.2	32.6	72.0	57.4	45.0	78.8	81.8	63.3
Canopy forming FG%		**58.8**	**49.8**	**54.4**	**84.6**	**49.4**	**69.4**	**82.8**	**83.4**	**35.0**	**83.8**	**71.6**	**78.6**	**87.6**	**96.4**	**67.5**
Understory brown algae	*Acrocarpia paniculata*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	1.2	1.6	2.8	7.3
	Carpoglossum confluens	7.3	6.4	10.6	17.6	9.6	2.4	0.4	3.2	16.0	4.0	3.6	10.0	2.4	1.8	3.5
	Carpomitra costata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0
	Cystophora monilifera	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora moniliformis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora platylobium	0.0	0.0	0.0	0.0	0.0	1.4	0.0	1.8	0.0	1.8	2.6	0.0	0.6	0.0	2.4
	Cystophora retorta	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora retroflexa	0.0	0.8	0.0	0.0	0.0	0.0	0.0	1.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0
	Dictyomenia tridens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
	Dictyopteris muelleri	1.8	0.2	0.0	0.0	0.0	0.5	0.0	0.0	0.6	0.4	1.4	0.0	0.0	0.0	0.0
	Halopteris paniculata	0.0	0.0	0.0	1.4	0.0	0.1	0.0	0.0	0.6	0.6	0.0	0.0	0.0	0.0	0.0
	Halopteris spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Lobophora variegata	0.0	0.5	0.0	1.2	1.0	0.0	0.0	0.1	1.0	2.4	3.4	1.4	1.0	3.6	0.3
	Perithalia caudata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.8	0.0	0.0	0.0	3.3
	Sargassum fallax	5.0	11.5	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	5.0	0.0	0.0	0.0	0.0
	Sargassum paradoxum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum spp.	0.2	0.2	0.0	2.2	1.2	1.5	1.0	0.7	1.2	1.6	0.4	0.0	4.0	2.4	0.0
	Sargassum spp. (subgenus *Arthrophycus*)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum verruculosum	0.8	0.4	0.0	0.0	8.4	0.0	0.0	0.2	2.6	0.0	0.0	3.0	0.0	3.8	0.0
	Sargassum vestitum	0.0	0.0	0.0	0.0	17.8	4.3	0.0	4.3	0.0	0.6	0.0	0.0	0.0	0.0	0.0
	Seirococcus axillaris	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.6	0.8	0.0	0.0	0.0
	Sporochnus comosus	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sporochnus radiciformis	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

	Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
Understory brown algae	Unidentified algae (filamentous brown)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	6.0	0.0	0.0	0.0	0.0	0.0	0.0
	Xiphophora gladiata	0.0	0.0	0.0	1.8	0.0	0.0	1.6	0.3	0.0	1.4	0.0	0.4	1.6	0.0	5.0
	Zonaria turneriana/angustata	1.3	0.8	0.0	1.8	2.4	0.9	3.4	1.3	5.0	3.8	1.4	1.0	1.8	2.2	1.3
	Understory brown algae FG%	**16.9**	**21.3**	**10.6**	**26.0**	**41.6**	**11.1**	**6.4**	**13.5**	**45.0**	**17.6**	**20.8**	**17.8**	**13.0**	**17.4**	**23.1**
Understory green algae	*Bryopsis* spp.	0.0	0.0	0.0	0.2	0.0	0.5	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa brownii	3.0	0.0	3.0	0.0	3.0	1.0	1.2	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.4
	Caulerpa flexilis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa geminata	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.8	0.0
	Caulerpa scalpelliformis	0.0	0.5	4.0	0.0	3.0	0.0	0.0	1.1	4.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa simpliciuscula	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.6	0.4	0.4	0.0	0.0	0.0	0.0	0.0
	Caulerpa spp. (rhizomes)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa trifaria	0.2	1.7	1.0	0.0	2.8	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Chaetomorpha spp.	0.0	0.1	0.8	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.4	0.0	0.0	0.0	0.0
	Cladophora spp.	0.3	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Codium fragile	0.0	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1
	Codium pomoides	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
	Codium spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Ulva spp.	0.0	0.2	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Understory green algae FG%	**3.5**	**4.6**	**10.8**	**0.2**	**9.8**	**1.6**	**1.8**	**3.3**	**4.4**	**0.4**	**2.0**	**0.0**	**0.2**	**0.8**	**1.5**
Understory red algae	*Areschougia* spp.	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.2	0.2	1.2	0.0	0.2	0.0	0.0
	Asparagopsis armata	1.5	4.7	0.0	0.0	5.4	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0	0.0	0.0
	Ballia callitricha	0.0	0.2	6.4	1.6	0.0	0.9	1.4	3.0	0.0	1.0	1.0	6.8	1.6	3.2	1.1
	Ballia scoparia	0.2	0.0	1.4	0.2	0.0	0.6	0.4	0.3	1.0	0.0	2.0	1.4	0.0	0.0	0.2
	Callophyllis lambertii	3.2	1.3	2.2	0.6	5.0	7.9	1.4	1.4	4.6	7.6	0.0	0.6	0.0	0.0	3.0
	Callophyllis rangiferina	12.8	7.7	1.2	2.0	2.4	1.0	1.8	1.9	1.6	1.4	0.0	0.2	0.0	0.2	3.0
	Ceramium spp.	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Champia spp.	0.3	0.2	0.0	0.0	0.0	0.1	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Chondria spp.	0.0	0.3	0.0	0.0	0.0	0.5	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Corallina officinalis	0.0	0.4	0.6	2.6	0.0	2.3	1.4	0.9	2.2	7.4	6.2	0.0	2.8	5.6	4.1
	Craspedocarpus ramentaceus	1.8	0.0	0.2	0.6	1.2	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
	Delisea plumosa	0.6	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Delisea pulchra	0.0	0.0	0.0	0.0	0.0	1.8	0.0	3.1	0.0	0.4	0.0	0.0	0.0	0.0	0.0
	Echinothamnion hystrix	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.0

	Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
Understory red algae	*Echinothamnion* spp.	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Erythrymenia minuta	1.5	1.1	0.6	0.4	0.0	0.4	0.2	0.3	0.2	0.8	0.0	0.0	0.0	0.0	0.0
	Euptilota articulata	0.0	0.3	0.4	0.2	0.2	2.1	1.0	0.7	0.0	0.0	0.2	1.0	0.0	0.4	0.1
	Euptilota sp.2	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Gelidium australe	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Haliptilon roseum	0.2	1.9	4.6	2.4	0.0	6.3	2.8	3.1	2.4	3.6	6.8	3.2	6.2	1.4	3.2
	Haloplegma preissii	0.0	0.0	1.4	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Hemineura frondosa	2.6	0.4	4.8	1.0	0.8	2.2	1.4	1.9	1.6	3.0	0.4	0.4	0.2	0.4	0.8
	Hypnea ramentacea	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.4	3.8	0.0	0.0	1.6	1.8	0.0	0.0
	Kallymenia cribrosa	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Kallymenia spp.	4.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Kallymenia tasmanica	0.0	3.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Laurencia elata	0.0	0.0	1.0	0.2	0.0	0.6	0.2	0.0	0.0	0.0	0.6	0.0	0.0	0.6	0.9
	Laurencia spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Lenormandia marginata	1.2	2.7	3.4	2.2	0.0	3.4	0.2	3.1	0.2	1.0	0.4	7.8	0.8	1.8	0.0
	Mastophoropsis canaliculata	0.0	0.0	0.0	0.0	0.0	0.5	1.0	2.5	0.0	0.0	0.0	0.0	0.4	2.4	0.0
	Melanthalia obtusata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.4	1.1
	Metagoniolithon radiatum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	2.4	0.0	0.0	0.0	0.0	0.4	0.0
	Mychodea acanthymennia	1.2	0.0	0.0	0.0	0.0	0.2	0.2	0.1	0.4	1.8	0.0	0.0	0.0	0.2	0.5
	Mychodea acicularis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	4.4	0.0	0.0	0.0	0.0	0.0	0.0
	Myriogramme gunniana	2.7	2.1	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Nitospinosa tasmanica	0.2	2.5	2.6	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Nizymenia australis	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Peyssonnelia novaehollandiae	1.4	7.5	0.2	3.2	0.0	0.9	0.4	2.2	0.0	3.0	1.0	1.8	0.6	0.6	5.2
	Phacelocarpus apodus	0.0	0.6	0.0	0.2	0.0	1.8	4.0	0.5	0.0	0.6	2.0	0.8	1.0	0.0	1.4
	Phacelocarpus peperocarpos	0.0	0.0	0.0	0.2	0.0	0.5	10.2	3.3	0.0	3.6	0.0	5.2	0.0	0.0	1.0
	Phacelocarpus spp.	0.9	0.0	0.0	0.0	1.8	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0
	Plocamium angustum	3.4	3.8	0.8	0.2	0.0	3.5	8.2	7.7	0.8	1.2	1.0	3.2	2.2	1.4	10.1
	Plocamium costatum	0.5	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Plocamium dilatatum	0.0	0.0	0.0	2.0	0.0	4.8	6.2	8.1	1.8	6.4	12.2	1.2	2.0	0.6	32.8
	Plocamium mertensii	0.0	0.0	0.0	0.4	0.0	2.1	6.4	7.5	0.0	0.0	3.2	0.0	0.0	0.0	3.9
	Pollexfenia lobata	5.8	1.9	0.0	0.0	0.0	1.3	0.2	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.0
	Polyopes constrictus	0.3	0.0	0.2	0.6	0.0	0.4	0.6	0.7	1.2	0.0	2.8	0.8	1.0	0.6	0.2

	Species	ZUID	PENG	REDC	ESPE	LOMA	SCOT	LIPP	TOWE	LADY	SIST	PART	BIGH	STPT	STIS	ACTA
Understory red algae	*Ptilonia australasica*	0.5	0.0	0.0	0.0	0.0	0.6	2.2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.3
	Rhodymenia sonderi	0.1	0.0	1.6	1.8	0.0	0.9	0.0	1.6	0.0	0.4	0.0	0.0	0.4	0.0	0.0
	Rhodymenia spp.	2.3	1.5	5.4	1.4	0.6	3.1	6.6	5.3	2.4	0.0	2.0	5.4	4.0	5.0	1.5
	Sarcodia marginata	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
	Sonderopelta coriacea	0.0	3.3	0.0	1.6	0.0	1.9	0.8	6.2	6.2	0.8	2.4	7.8	0.0	0.0	2.0
	Sonderopelta/Peyssonnelia	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Stenogramme interrupta	0.8	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Synarthrophyton patena	0.0	0.0	0.0	0.2	0.0	0.2	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Thamnoclonium dichotomum	0.5	3.0	0.8	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.4	1.2	0.0	0.0	0.0
	Unidentified algae (filamentous red)	0.5	9.0	2.6	0.0	0.8	0.0	0.0	0.0	4.2	0.0	4.8	2.4	7.6	0.0	1.6
	Unidentified algae (foliose red)	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (red)	0.0	0.0	0.0	0.0	2.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Understory red algae FG%		**53.9**	**60.7**	**44.0**	**28.8**	**23.2**	**56.5**	**60.4**	**68.0**	**42.6**	**45.2**	**50.6**	**54.8**	**33.4**	**25.4**	**78.0**
Encrusting algae	*Peyssonnelia* spp. (encrusting)	24.5	16.8	3.6	23.4	27.4	15.9	8.4	11.4	12.4	20.0	14.0	24.0	13.6	20.4	4.0
	Unidentified algae (crustose coralline)	20.4	17.6	4.8	49.2	12.8	30.5	38.0	28.4	25.4	52.8	19.2	23.2	24.4	25.6	27.1
	Unidentified algae (encrusting brown)	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Encrusting algae FG%	**44.9**	**34.4**	**8.4**	**72.6**	**40.8**	**46.4**	**46.4**	**39.8**	**37.8**	**72.8**	**33.2**	**47.2**	**38.0**	**46.0**	**31.1**
Sessile invertebrates	Amathia wilsoni	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Erythropodium hicksoni	0.7	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
	Galeolaria spp.	0.4	4.4	0.0	0.4	0.0	0.0	0.0	0.0	0.2	0.2	0.0	3.0	0.0	2.8	0.0
	Ostrea angasi	0.3	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Parazoanthus spp.	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0
	Pyura australis	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.2	0.0	0.0
	Pyura stolonifera	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified ascidians	0.2	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0
	Unidentified ascidians (encrusting)	2.1	0.0	0.0	0.8	1.8	0.5	1.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified bryozoans	1.8	9.3	12.8	0.0	0.0	3.1	0.0	1.8	0.0	0.0	6.0	15.8	11.2	1.4	5.3
	Unidentified bryozoans (hard)	2.2	4.0	7.6	0.6	0.2	1.0	2.4	1.2	3.0	1.0	2.4	3.6	3.4	5.8	1.6
	Unidentified bryozoans (soft)	1.2	0.0	0.0	17.2	0.2	3.8	6.6	4.5	2.0	1.4	0.0	0.0	0.0	0.0	0.0
	Unidentified hydroid	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified invertebrates (encrusting)	1.3	0.0	0.0	2.4	0.0	7.5	3.6	6.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sessile invertebrates	Unidentified sponge (encrusting)	11.7	5.6	16.2	5.4	2.2	10.3	12.4	6.6	5.2	4.4	10.4	11.2	14.0	14.4	7.9
	Unidentified sponges	2.8	5.0	3.8	1.8	0.8	1.3	1.2	2.0	0.0	0.2	5.0	1.8	2.4	1.4	1.5

	Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
	Sessile invertebrates FG%	**24.7**	**29.5**	**40.4**	**28.6**	**5.6**	**27.5**	**29.0**	**23.4**	**10.8**	**7.4**	**23.8**	**36.2**	**31.2**	**25.8**	**16.3**
Substrate	Bare rock (non - barrens)	0.9	0.0	0.0	1.8	0.8	2.1	1.8	0.1	0.2	1.6	0.0	0.0	0.0	0.0	0.0
	Cobble	0.0	0.0	0.0	0.0	0.0	0.2	0.0	4.0	0.0	8.0	0.0	4.0	0.0	0.0	0.0
	Gravel	0.0	0.0	0.0	0.0	0.0	4.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Sand	9.6	0.2	8.0	0.0	8.8	2.4	0.0	5.3	30.6	6.0	7.0	1.4	3.4	12.2	6.3
	Shell	0.0	0.5	0.0	0.0	1.2	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0
	Turf/sand/sediment matrix	4.0	16.0	8.0	3.6	4.0	2.2	5.6	3.5	7.0	1.4	9.2	12.2	4.2	8.4	11.6
	Unidentified algae (brown turf)	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (red turf)	4.3	3.3	8.4	1.2	3.2	2.2	4.6	3.5	3.6	2.6	11.2	12.8	5.4	4.0	6.5
	Substrate FG%	**19.1**	**20.0**	**24.4**	**6.6**	**18.0**	**13.4**	**12.0**	**16.4**	**41.4**	**20.4**	**27.4**	**30.4**	**13.0**	**24.6**	**24.4**
	Algal species richness	**36**	**34**	**31**	**31**	**23**	**41**	**32**	**46**	**36**	**31**	**30**	**28**	**26**	**26**	**32**

NB: FG refers to Functional Group with the%FG being the summed percentage of all taxa within that group. Algal species richness is the sum of all algal species (only) found at any given site.

Quadrat analysis suggested that *M. pyrifera* was present in low abundances at several sites throughout the SE Channel region (0.5 – 2.7% average cover where present; Table 6.4-5). However, *M. pyrifera* stype counts along transects in this region indicated that *M. pyrifera* was actually the dominant canopy forming species at ACTA (162.5 ± 81.1 stypes/50 m transect), and that there was moderate cover at PART (15.5 ± 11.0 stypes/50 m transect; Figure 6.4-12). Stypes were recorded at several sites where *M. pyrifera* was determined to be absent using quadrat analysis, including BIGH, SIST, STPT and STIS. However, quadrat analysis detected *M. pyrifera* at REDC, LOMA and TOWE, where stype counts suggested otherwise (Table 6.4-5, Figure 6.4-12).

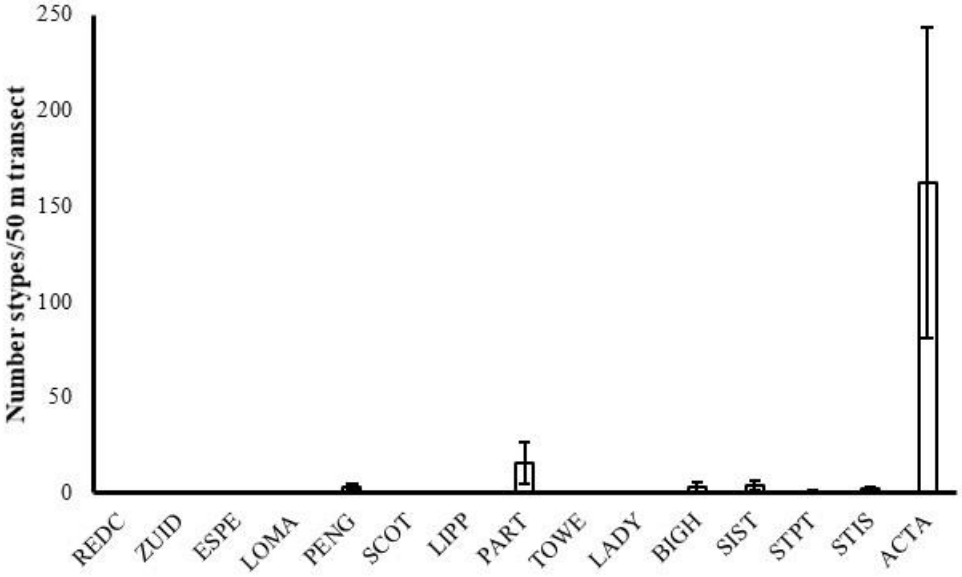

Figure 6.4-12 Average number of *Macrocystis pyrifera* stypes recorded per 50 m transect at each of the sites in the SE Channel region.

There appears to be a strong relationship between exposure and composition of the algal community in the SE Channel. Overlaying these factors on the ordination plots suggests that wave power (kW/m) shows a good correlation with certain sites and as such may explain some of the variability in the data (Figure 6.4-13). The northern-most sites (ZUID, PENG) stand apart in the ordination plot, with *E. radiata* as the dominant canopy species, and these sites were clearly subject to lower mean wave power (<2 kW/m) and significant wave heights (<0.2 m). The wave power and height were greater at most other sites in the SE Channel, which in turn had a markedly different algal community composition to ZUID and PENG. At these sites, *P. comosa* was the dominant canopy forming species (Table 6.4-5).

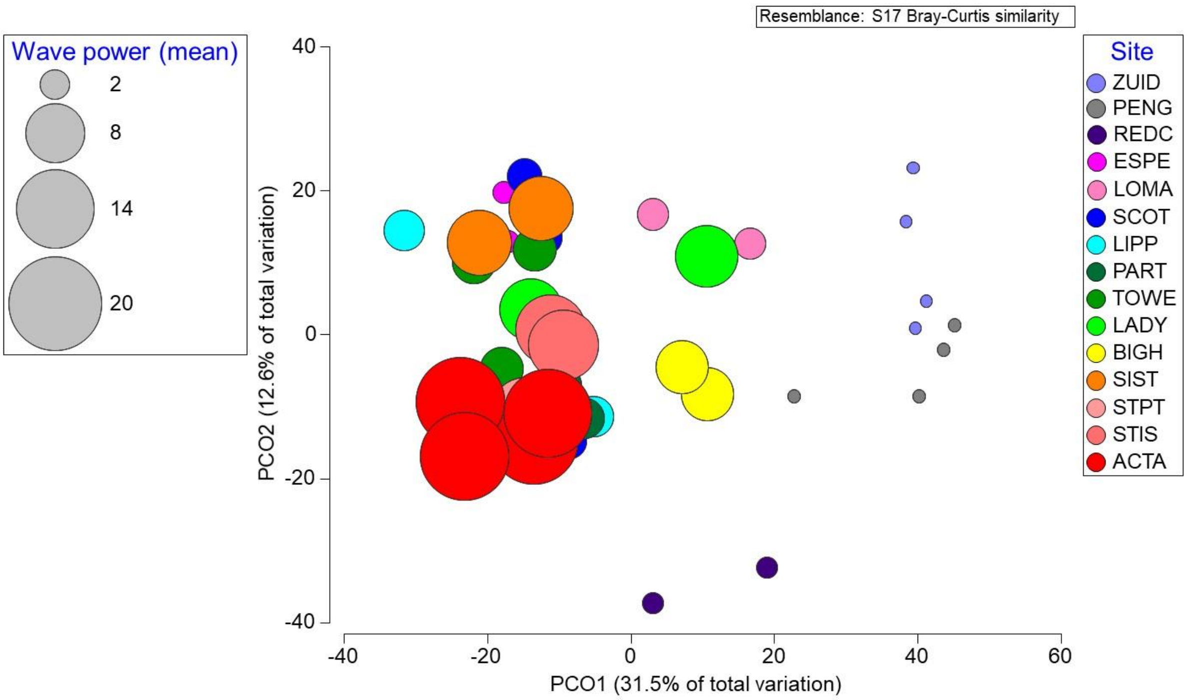

Figure 6.4-13 Principal Coordinate Analysis (PCO) on algae community data (5 m) with mean wave power (5-year average) overlain at 15 sites in the SE Channel.

6.4.3.2.2 Invertebrates

There were significant differences in the invertebrate abundances between sites ($F_{(14, 37)}$ = 4.1142, p = < 0.0001 (*MC*)) in the SE Channel region, with the far southern site ACTA in particular differing from all other sites. Invertebrate assemblages at this site were dominated by the blacklip abalone, *Haliotis rubra* (11.3 individuals/50 m transect), with very low abundance (<1.3 individuals/50 m transect) of any other species recorded (Figure 6.4-14, Table 6.4-6). As in the North Bruny region, the featherstar (*Comanthus trichoptera*) dominated invertebrate assemblages across the SE Channel region, with all sites other than ACTA recording high abundances of this species (Figure 6.4-14, Table 6.4-6).

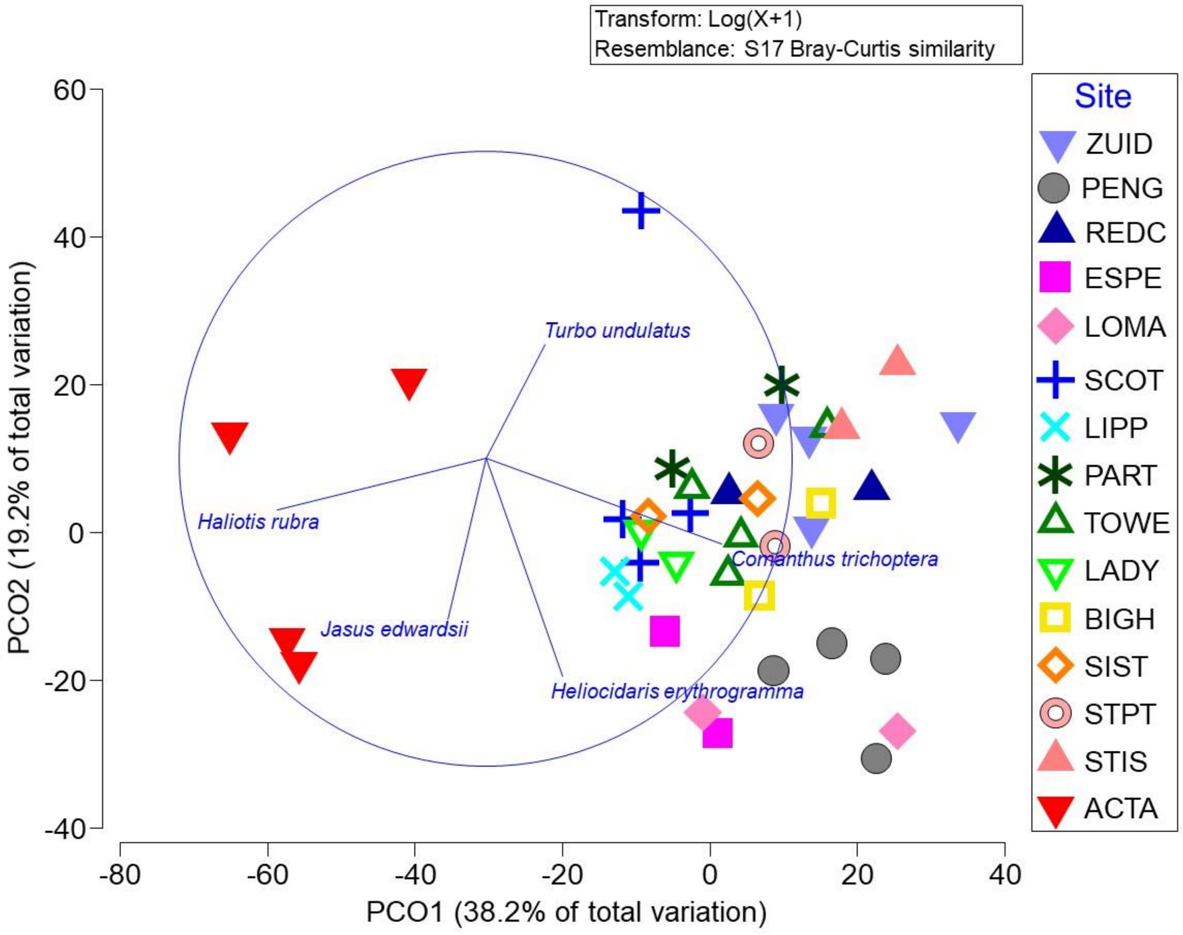

Figure 6.4-14 Principal Coordinate Analysis (PCO) on invertebrate abundance at 5 m from 15 sites in the SE Channel in February 2017. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates with a base variable comparison of > 0.4 included.

PENG, ESPE and LOMA were characterised by comparatively higher numbers of *H. erythrogramma*, with abundances of 64.5, 27.0 and 72.0 individuals/50 m transect respectively (Figure 6.4-14, Table 6.4-6). While *C. trichoptera*, *H. rubra* and *H. erythrogramma* had high abundances at several sites, the numbers of all other invertebrate species were generally low (<5 individuals/50 m transect; Table 6.4-6).

Total abundance of invertebrates/50 m transect was low throughout the region and reflective of the number of *C. trichoptera* found at each site (Table 6.4-6). For example, at ACTA, where the abundance of *C. trichoptera* was only 0.5 individuals/50 m transect, a total of only 14.5 individuals were observed per 50 m transect. Likewise, at BIGH, where the abundance of *C. trichoptera* was high (220.0 individuals/50 m transect), a total of 227.0 individuals/50 m transect was recorded (Table 6.4-6). Species richness was also low, with < 10 invertebrate species observed at all sites except TOWE and SCOT. REDC was particularly species depauperate, with an average of only three species (*C. trichoptera*, *H rubra* and *Jasus edwardsii* (southern rock lobster)) recorded here (Table 6.4-6).

Table 6.4-6 Abundance of invertebrate species (no. individuals/50 m transect) from sites in the SE Channel surveyed in February 2017.

	Site														
Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
Amblypneustes ovum (Short-spined urchin)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0
Australostichopus mollis (Sea cucumber)	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	1.0	0.0
Cabestana spengleri (Triton shell)	0.0	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cabestana tabulate (Fringed triton)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ceratosoma amoenun (Sweet ceratosoma)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ceratosoma brevicaudatum (Short-tailed nudibranch)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Comanthus tasmaniae (Feather star)	0.8	0.0	0.0	0.0	0.0	1.0	0.0	5.0	0.0	26.5	0.5	0.0	0.0	0.0	0.0
Comanthus trichoptera (Feather star)	138.8	110.0	36.5	191.0	102.5	41.3	111.5	123.3	48.5	150.0	34.5	220.0	50.0	62.5	0.5
Coscinasterias muricate (Eleven-armed star)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0
Dicathais orbita (Dog whelk)	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Diogenid spp. (Hermit crab)	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fromia polypore (Many-spotted seastar)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
Goniocidaris tubaria (Pencil urchin)	0.0	0.0	0.0	1.0	0.5	0.8	1.0	0.5	0.0	0.0	0.0	0.5	1.5	0.0	0.0
Haliotis rubra (Black-lip abalone)	1.5	1.0	1.0	7.5	7.0	5.5	24.5	1.8	18.5	3.5	2.0	1.5	0.5	0.5	11.3
Heliocidaris erythrogramma (Purple urchin)	0.3	64.5	0.0	27.0	72.0	0.3	0.0	0.3	2.0	0.0	0.0	0.0	0.0	0.0	0.0
Jasus edwardsii (Southern rock lobster)	0.0	1.5	1.0	1.0	1.5	0.8	5.5	2.5	0.0	1.0	1.0	1.0	1.0	0.0	1.3
Meridionale ambigua (Yellow sea spider)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0
Nectocarcinus tuberculosus (Velvet crab)	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nectria ocellate (Ocellate seastar)	0.0	0.3	0.0	4.0	0.0	0.5	2.5	2.5	1.5	1.0	0.0	0.5	1.5	0.0	0.3
Ostrea angasi (Native oyster)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Penion maximus (Giant whelk)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.5	0.0	0.0	0.0	0.0	0.0	0.0
Penion spp. (Whelk)	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Petricia vernicina (Velvet star)	0.0	0.5	0.0	2.0	1.0	1.8	2.0	1.8	1.0	1.0	1.0	2.5	2.5	0.0	0.3
Plagusia chabrus (Red bait crab)	0.0	0.0	0.0	1.5	0.0	2.0	4.0	1.8	0.0	0.5	0.5	0.0	1.5	0.5	1.0
Pleuroploca australasia (Tulip shell)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ranella australasia (Australian rock whelk)	0.3	0.0	0.0	0.0	0.0	0.3	1.0	0.0	0.5	0.5	0.0	0.0	0.0	0.0	0.0
Turbo undulatas (Turban shell)	2.5	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total Abundance	144.8	**178.0**	**38.5**	**236.5**	**184.5**	**55.0**	**152.0**	**140.0**	**74.5**	**184.0**	**39.5**	**227.0**	**60.0**	**64.5**	**14.5**
Species Richness	7	**7**	**3**	**9**	**6**	**12**	**8**	**12**	**7**	**8**	**6**	**8**	**9**	**4**	**6**

NB: Total abundance refers to the total number of individuals on each transect (transect mean per site) and species richness refers to the sum of species recorded on each transect.

Whilst the data might suggest a direct relationship between the exposure level and invertebrate community structure, this is in large part due to the clear separation of ACTA from all other sites. ACTA had the largest values for wave power (Figure 6.4-15); however, ACTA is a unique site for many reasons. Consequently, this may be an indirect relationship, as the composition of algal and fish assemblages also differed markedly at this site (Table 6.4-5, Table 6.4-7). Unfortunately, establishing quantitative links between exposure and the other environmental factors was beyond scope of this study. Reef rugosity appeared to exert very little influence on site-level invertebrate biodiversity in the SE Channel. There was little differentiation at a site level that could be aligned with either habitat complexity or cryptic habitat availability (Appendix 2).

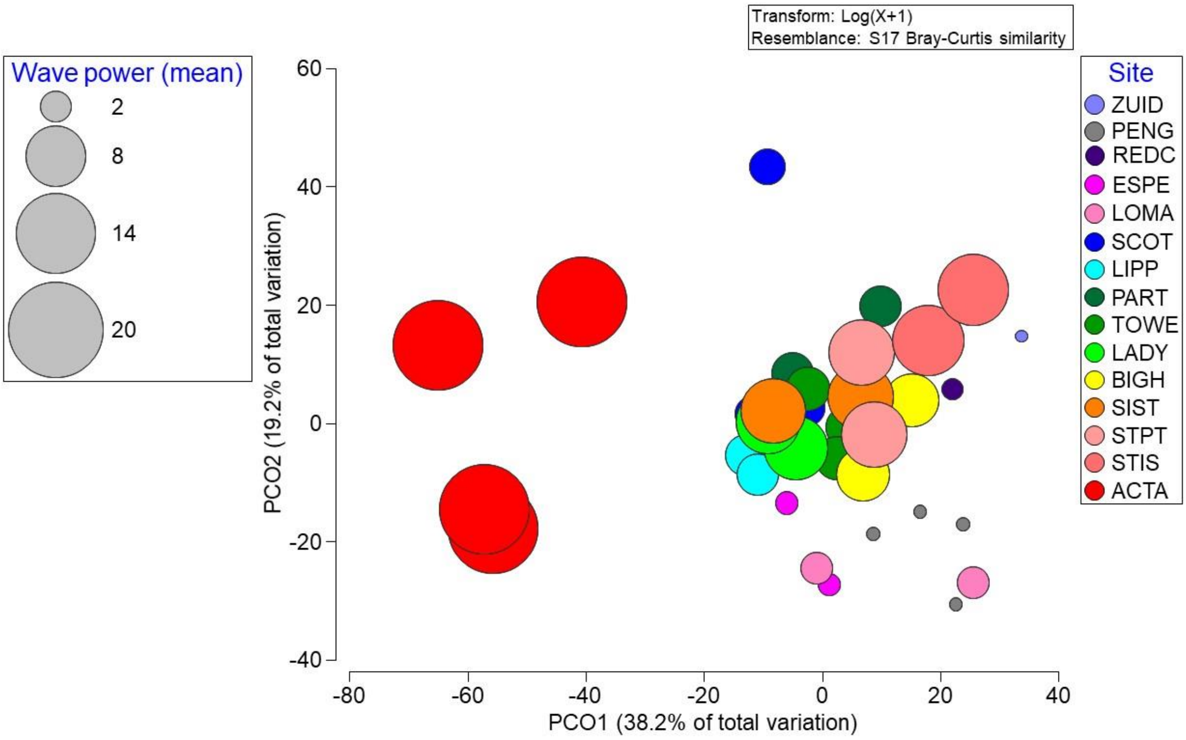

Figure 6.4-15 Principal Coordinate Analysis (PCO) on invertebrate community data (5 m) with mean wave power (5-year average) overlain at 15 sites in the SE Channel.

6.4.3.2.3 Fish

The latitudinal spatial gradient observed in the algae communities was not evident in the fish data from the SE Channel region, but fish communities did vary significantly between sites ($F_{(14, 37)} = 3.6513$, $p = < 0.0001$ (*MC*)). Changes in abundances of the southern hulafish (*Trachinpos caudimaculatus*) tended to characterise fish communities, with high numbers recorded at ZUID (330.2 individuals/50 m transect), PENG (580.0 individuals/50 m transect) and BIGH (480.0 individuals/50 m transect; Figure 6.4-16, Table 6.4-7). In contrast, southern hulafish were not recorded at either ACTA and LADY and both these sites tended to cluster together with low values for PCO1 (Figure 6.4-16).

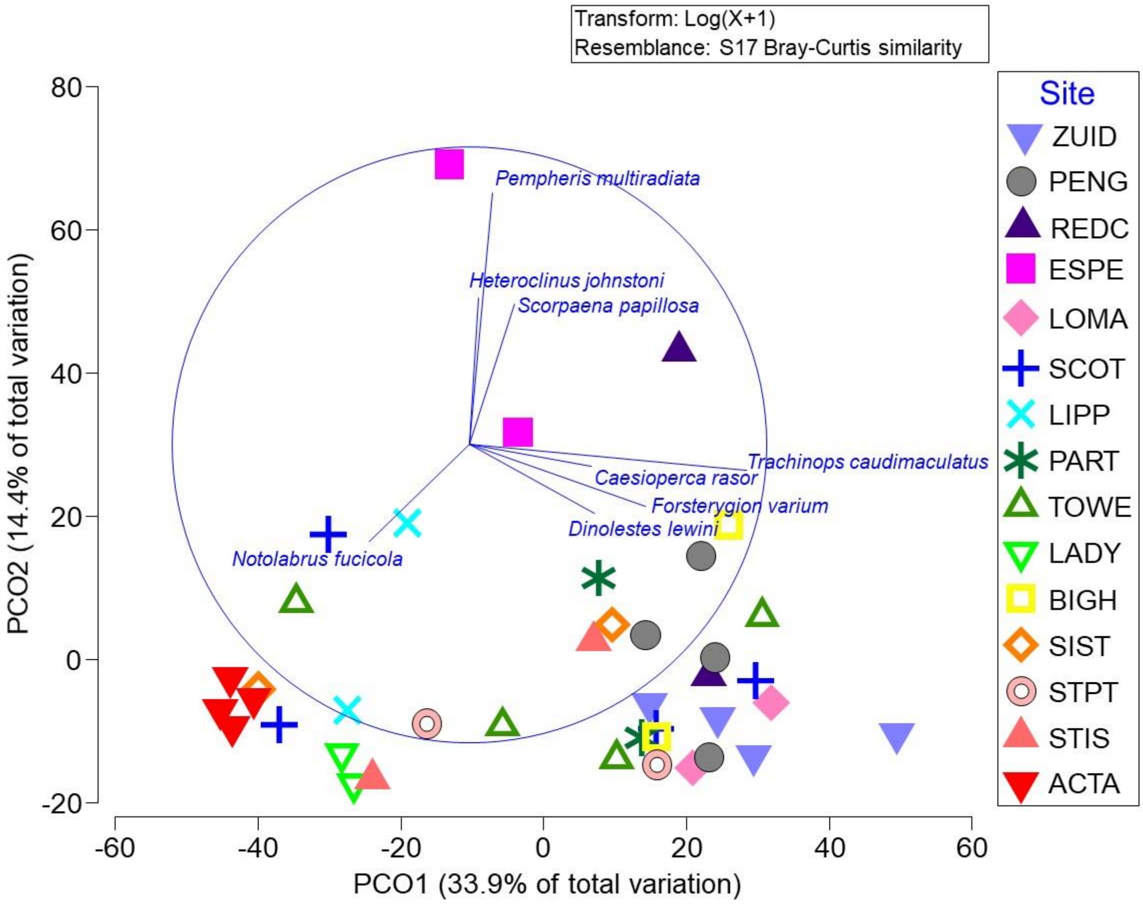

Figure 6.4-16 Principal Coordinate Analysis (PCO) on fish abundance at 5 m from 15 sites in the SE Channel in February 2017. Sites are ordered in the legend from north to south. Vector overlays show the species that contribute most to the sample separation along principal coordinates with a base variable comparison of > 0.4 included.

The composition of fish assemblages across the region was typical of southeastern Tasmania with all sites having high numbers of wrasse and leatherjacket species (Table 6.4-7). Schools of individual species were recorded at some sites, including long-finned pike (*Dinolestes lewini*) (ZUID, STIS, STPT) and common bullseye (*Pempheris multiradiata*) (REDC, ESPE; Table 6.4-7).

Total abundance of fish at any site was generally reflective of the abundance of southern hulafish, with sites where this species was absent, such as ACTA and LADY recording low total fish abundance (16.0 and 17.0 individuals/50 m transect, respectively; Table 6.4-7). Overall species richness was low, with less than ten species per 50 m transect recorded at eight out of the 15 sites surveyed (Table 6.4-7). Species richness was generally higher in the northern part of the SE Channel region compared to the south (Table 6.4-7).

Table 6.4-7 Abundance of fish species (no individuals/50 m transect) from sites in the SE Channel surveyed in February 2017.

	Site														
Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**	**LIPP**	**TOWE**	**LADY**	**SIST**	**PART**	**BIGH**	**STPT**	**STIS**	**ACTA**
Acanthaluteres vittiger (Toothbrush leatherjacket)	0.5	0.0	0.5	1.0	20.0	3.0	1.5	1.0	1.0	30.5	0.0	0.5	0.5	27.5	0.0
Aplodactylus arctidens (Marblefish)	0.0	0.0	0.5	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Aracana aurita (Shaw’s cowfish)	0.0	0.0	0.0	0.0	0.5	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Bovichtus angustifrons (Dragonet)	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Caesioperca razor (Barber perch)	4.8	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cephaloscyllium laticeps (Draughtboard shark)	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cheilodactylus nigripes (Magpie perch)	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dinolestes lewini (Long-fin pike)	32.5	20.0	1.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	9.5	81.5	32.0	0.0
Diodon nicthemerus (Globefish)	0.5	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Forsterygion varium (Variable three-fin)	0.8	1.8	0.0	0.0	4.5	0.5	0.0	0.3	0.0	0.0	0.5	0.5	3.0	0.0	0.0
Genypterus tigerinus (Rock ling)	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heteroclinus johnstoni (Johnston’s Weedfish)	0.0	0.5	0.0	0.5	0.0	0.3	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0
Heteroclinus perspicillatus (Common Weedfish)	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Heteroclinus tristis (Longnose Weedfish)	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Latridopsis forsteri (Bastard trumpeter)	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lotella rhacina (Beardie)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Meuschenia australis (Brownstriped leatherjacket)	1.0	0.0	0.0	0.0	0.5	0.3	0.0	1.0	1.5	1.5	0.5	0.0	0.0	2.0	0.3
Meuschenia freycineti (Sixspine leatherjacket)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Nemadactylus macropterus (Jackass morwong)	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Neoodax balteatus (Little rock whiting)	9.0	0.5	0.5	0.0	14.0	0.0	0.0	0.5	3.0	0.5	0.0	0.0	0.0	0.0	0.0
Neosebastes scorpaenoides (Gurnard perch)	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.0
Norfolkia clarkei (Clark’s triplefin)	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Notolabrus fucicola (Purple wrasse)	0.3	6.0	0.0	0.5	1.0	1.3	4.5	2.5	2.5	1.0	14.5	4.0	5.5	4.0	6.8
Notolabrus tetricus (Blue-throat wrasse)	10.3	6.5	8.0	4.0	4.5	5.5	8.5	7.5	7.0	4.0	8.5	2.0	2.0	1.5	7.0
Odax cyanomelas (Rainbow cale)	0.0	0.0	8.5	0.5	0.0	0.3	1.5	0.0	0.0	0.5	0.0	0.0	0.0	0.5	0.8
Parascyllium ferrugineum (Rusty catshark)	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Parma microlepis (White-ear)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
Pempheris multiradiata (Common bullseye)	0.0	0.3	75.0	38.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	7.5	0.0	0.5	0.0

Pentaceropsis recurvirostris (Long-snouted boarfish)	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pictilabrus laticlavius (Senator wrasse)	3.0	0.5	1.0	0.0	0.5	1.0	0.5	0.5	1.0	0.0	0.5	0.0	0.0	0.5	0.0
Pseudocaranx georgianus (Silver trevally)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	104.5	0.0	0.0
Pseudolabrus rubicundus (Rosy wrasse)	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.5	0.0	0.5
Pseudophycis bachus (Red cod)	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pseudophycis barbata (Bearded Rock Cod)	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Scorpaena papillosa (Southern Red Scorpionfish)	0.0	0.8	0.0	0.5	0.0	0.0	0.0	0.5	0.0	0.5	0.5	0.5	0.0	0.0	0.0
Scorpis aequipinnis (Sea sweep)	0.0	0.5	1.0	0.0	0.0	0.0	5.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8
Scorpis lineolata (Silver sweep)	0.0	0.0	0.0	0.0	0.0	0.0	3.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Siphonognathus beddomei (Pencil weed whiting)	0.0	0.3	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trachinops caudimaculatus (Hulafish)	330.5	580.0	142.5	5.0	251.5	75.0	0.5	82.0	0.0	35.0	179.5	480.0	15.0	12.5	0.0
Trinorfolkia clarkei (Common triplefin)	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Urolophus cruciatus (Banded stingaree)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total abundance	**396.3**	**621.0**	**239.0**	**50.5**	**299.0**	**87.8**	**28.0**	**101.0**	**17.0**	**73.5**	**205.0**	**504.5**	**213.0**	**81.0**	**16.0**
Species Richness	**16**	**16**	**11**	**9**	**11**	**12**	**13**	**14**	**7**	**8**	**8**	**8**	**9**	**9**	**6**

NB: Total abundance refers to the average total number of individuals on each 50 m transect (i.e. transect mean per site) and species richness refers to the average sum of species recorded on each 50 m transect.

Unlike the Bruny region, exposure based on mean wave power (kW/m) did not appear to influence site-level composition of fish communities (Figure 6.4-17), with higher within-site variability observed in fish communities compared to algae.

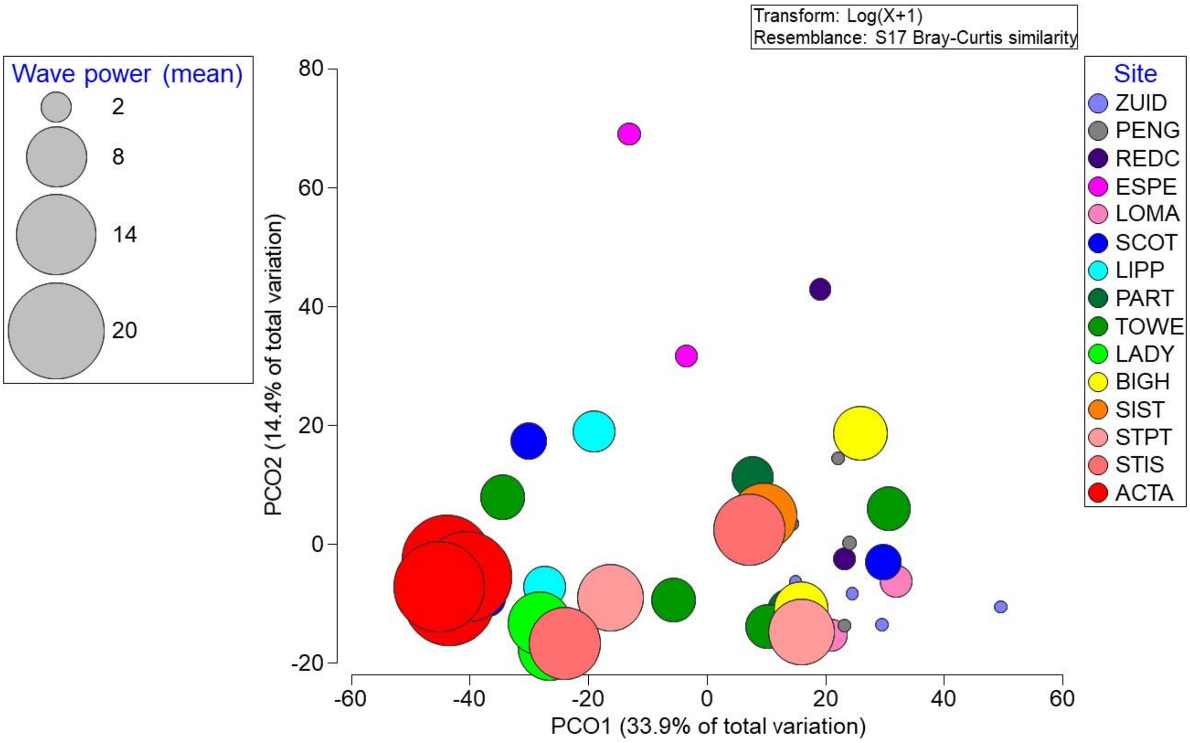

Figure 6.4-17 Principal Coordinate Analysis (PCO) on fish community data (5 m) with mean wave power (5-year average) overlain at 15 sites in the SE Channel.

6.4.4 Discussion

Overall, the communities found at the sites in both North Bruny and SE Channel regions were typical of the marine flora and fauna previously observed in semi-exposed and exposed reef habitats of southeastern Tasmania (Stuart-Smith et al. 2008; Valentine et al. 2016). While there were differences between regions and sites, there were some spatially consistent features. For example, fish assemblages were generally characterised by large numbers of wrasse (family Labridae) and invertebrate assemblages by relatively consistent numbers of featherstars (*Comanthus trichoptera*) and black-lip abalone (*Heliotis rubra*). The algal communities tended to be dominated by high percentage cover of the canopy-forming species *Phyllospora comosa* and *Ecklonia radiata*, with diverse red algae assemblages in the understory and crustose coralline substrate.

Whilst it is important to be mindful that the surveys of North Bruny and SE Channel were undertaken in consecutive years, if the assumption is made that the species assemblages would be broadly similar between 2016 and 2017, then the data would tend to suggest that the SE Channel region has a higher diversity of algal species than North Bruny. The influence of the tannin-rich Huon River in modifying the light environment and driving unique algal assemblages around Ninepin Point, particularly in red species, is well established (Barrett et al. 2007). It is possible this influence extends further down the Channel than previously thought, creating an environmental gradient in light conditions that is reflected in higher species abundances throughout the system. Higher environmental variability, particularly in a factor such as light, could drive the observed increases in algal biodiversity.

Giant kelp (*Macrocystis pyrifera*) was also observed at more sites and in higher densities in the SE Channel than in the North Bruny region. Populations of giant kelp have declined in recent years throughout Tasmania, and this is believed to be largely due to the increasing influence of the warm, nutrient-poor Eastern Australian Current (EAC) on the east coast (Johnson et al. 2011; Mabin et al. 2019). Declines in kelp on the east coast have been particularly pronounced, even as far south as Bruny Island (Johnson et al. 2011). Previous studies have indicated the far-south of Tasmania (e.g., Southport) where cooler, nutrient rich water is more likely to be prevalent, has been a refuge for this species (Johnson et al. 2011). While the high abundance of giant kelp observed at the Actaeons is likely to be associated with these far south refugia, observations of giant kelp at sites within the Channel, such as Redcliffs or Penguin Point that experience higher summer temperatures are interesting. At these sites, the detrimental effects of higher temperature may be mitigated in part by nutrient inputs such as those from the Huon River, salmon aquaculture and other industries that stimulate growth of this species (Johnson et al. 2011; Mabin et al. 2019).

While there was always a small number of endemic algae, invertebrate or fish taxa at each site, the Actaeons had a particularly distinctive species assemblage. It was the only site dominated by a giant kelp (*Macrocystis pyrifera*) canopy, with an especially high cover of red understory algae (*Plocamium dilatatum* and *Plocamium angustatum*) recording a combined average cover of over 40% per quadrat. Previous algal surveys also noted the particularly high cover of red understory algae at this site (56% per quadrat), although the dominant species were not specified (Valentine et al. 2016). The Actaeons also stood out as having high population abundances of black-lip abalone and smaller populations of southern hulafish (*Trachinops caudimaculatus*) and featherstars (*Comanthus trichoptera*) which were common at all other sites. The dominance of both giant kelp and black-lip abalone at this site make it especially significant, both in terms of economic and conservation value.

Many of the survey sites in the North Bruny region were selected based on availability of historical information from those sites. The current data showed that the dominant fish and invertebrate species were similar to those observed in previous biodiversity surveys in 1994 and 2006 (Stuart-Smith et al. 2008). All surveys (1994, 2006 and 2016) showed that fish communities were largely dominated by species of wrasse (family Labridae), in particular blue-throat wrasse (*Notolabrus tetricus*) and purple wrasse (*Notolabrus fucicola*), with large numbers of southern hulafish (*Trachinops caudimaculatus*) also observed (Stuart-Smith et al. 2008). The algal communities also appeared to be similar to the historical data, with Stuart-Smith et al. (2008) observing dominance of canopy-forming macroalgal species *Phyllospora comosa* and *Ecklonia radiata*, along with similar assemblages of understory brown, green and red taxa. Unfortunately, only two sites in the SE Channel region had robust biodiversity data from prior surveys, with IMAS having undertaken biodiversity surveys at Zuidpool Rock and the Actaeons in 1994 and 2006. The data again suggested relatively similar dominant species at those sites over time.

The macroalgae communities clearly responded to different exposure levels in both regions; however, this result was far less pronounced for fish and invertebrate taxa. The influence of exposure on algal communities has been well documented, with the data in this study largely conforming to both the descriptive model outlined by Edgar (1984) and the quantitative model described by Hill et al. (2010). In this study, the canopy was dominated by *Phyllospora comosa* at sites with higher exposures, whilst more sheltered sites were largely dominated by *Ecklonia radiata*. Cape Queen Elizabeth North, which is located in a small gullet surrounded by cliffs and subject to high water movement even in very calm weather, recorded *Durvillaea potatorum* in the canopy, which both Edgar (1984) and Hill et al. (2010)

suggest is typical of maximally exposed sites. In the North Bruny region, the transition from *P. comosa* dominated systems to *E. radiata* dominated systems occurred around One Tree Point. In the SE Channel, a similar transition was observed but in this case the change point was around Redcliffs. Sites south of these points were subject to greater oceanic influence and generally *P. comosa* dominated, whereas sites to the north were less exposed and dominated by *E. radiata*. This suggests that these sites represent the point where the physical environment and exposure levels shift from sub-maximal to a more moderate level (Edgar 1984).

The Actaeons proved to be an exception to the patterns observed at all other sites. The Actaeons had the highest wave power and maximum wave height across both regions yet was dominated by giant kelp *M. pyrifera*. The relationship between exposure and *M. pyrifera* was not explored by Hill et al. (2010), but Edgar (1984) suggested that giant kelp could be common, if not dominant, in sub-maximally or moderately exposed regions, provided the substrate was appropriate i.e. horizontal. Physiological studies suggest *M. pyrifera* would prefer more exposed conditions as it requires a certain level of water movement to ensure adequate supply of nitrogen into tissues (Hepburn et al. 2007). However, given the current data suggests that the community assemblages and biodiversity at the Actaeons was relatively unique, it is possible that other environmental factors may also be influencing the biology and ecology at this site. Numerous environmental factors are likely to covary with exposure, including sea surface temperature (SST), nutrient levels, proximity to the Southern Ocean and light availability, and these may all influence algal species assemblage at any given site. SST and nutrient levels are known to be contributing factors for *M. pyrifera* proliferation (Johnson et al. 2011; Hadley et al. 2018; Mabin et al. 2019). Clearly these results suggest a need for further work to characterise the effect of these key environmental parameters on biodiversity of reef ecosystems, and further studies should aim to incorporate a broad range of potential covariates in that analysis.

It was not possible to clearly identify the influence of rugosity on invertebrate assemblages. Rugosity influences the species diversity on a reef, largely due to increased resource availability in highly complex habitats (Alexander et al. 2009). Alexander et al. (2009) calculated a rugosity index based on contouring a rope of known length to a reef surface profile. In this study, visual metrics were used to ascertain a categorical estimate of reef rugosity that aimed to capture both habitat complexity and availability of cryptic habitat. However, the results showed surprisingly little correlation between this index and biodiversity. This result may be real, but it is possible that the scoring method used in this study did not provide the level of detail necessary to accurately characterise the relationship between invertebrates and rugosity. Conversely, the attempt to characterise rugosity may have been at too great a spatial scale. Rugosity estimates were averaged over a 50 m transect because that was the unit of measure for the assessment of invertebrate communities. However, Alexander et al. (2009) found rugosity measurements at 5 m spatial scales were the best predictor for species richness. Consequently, further refinement of the rugosity assessment may be necessary to accurately relate reef assemblages to habitat complexity or availability of cryptic habitat.

While the primary aim of this chapter was to characterise algal, fish, and invertebrate assemblages as part of a baseline assessment of the rocky reef ecosystems in two regions intended for expansion of salmon farming, the objectives of the broader FRDC project were to develop fit-for-purpose monitoring tools for detecting effects of salmon farming on a range of habitats. In a previous FRDC project, Valentine et al. (2016) used data obtained through “Edgar-Barrett” biodiversity surveys to investigate the interaction between rocky reef communities and salmon farming in the D’Entrecasteaux Channel. While they found no relationship between salmon farming and reef ecosystems, they also acknowledged several limitations to the technique, some of which were confirmed by this study. For instance, accurately characterising macroalgae, fish and invertebrate communities requires many sites to be assessed. However, the assessment is a labour-intensive exercise that requires a high level of expertise, which perversely limits the number of sites that can be covered within a given region (Valentine et al. 2016). The “Edgar-Barrett” approach is also sub-optimal for capturing opportunistic indicator species within macroalgal communities, particularly epiphytic or filamentous algae, which are likely to be first responders within temperate reef ecosystems to nutrient enrichment. The was believed to be largely due to the high spatial and temporal variability in the distribution of these species (Valentine et al. 2016). Valentine et al. (2016) went on to recommend the development of a more targeted rapid approach for assessing organic enrichment on temperate rocky reefs, a concept which we have explored further through subsequent chapters in this FRDC project report.

Whilst a more targeted approach does have benefits in terms of rapid detection of organic enrichment, several recent studies have highlighted metrics obtained through biodiversity surveys that can indicate effects of organic enrichment on temperate rocky reefs. In a comprehensive study of metropolitan embayments, Stuart-Smith et al. (2015) found that reefs impacted by urbanisation were characterised by smaller, faster growing species, reduced fish biomass and richness, and reduced mobile invertebrate abundance and richness. Similarly, Ling et al. (2018) observed declining functional and species richness in reef communities in relation to heavy metal concentrations and nutrient enrichment in inshore coastal systems. These studies provide an insight into change in community structure and function in relation to nutrient enrichment, along with potential loss of ecosystem integrity that is otherwise difficult to detect. Both studies highlight the insight that can be gained from biodiversity assessments that may not be obtainable through targeted surveys. It should also be noted the nutrient enrichment levels that occur in urban estuaries are often quite high relative to those observed in open coastal ecosystems and as such may be easier to distinguish. Having said that, these analyses are relatively complex in nature and require specialist skill sets to analyse and interpret. Also, the field efforts required to capture this level of data is substantial and may not be feasible on a regular basis. While not necessarily fit-for-purpose for immediate adoption into routine monitoring programs, these analyses provide critical information highlighting the insight that can be gained from biodiversity assessments that may not be obtainable through targeted surveys.

The “Edgar-Barrett” method for characterising temperate reef assemblage has some clear advantages and should form part of the baseline for future reef monitoring programs. With marine ecosystems increasingly subjected to a range of stressors, it is important to be able to distinguish the nature (anthropogenic versus natural) and strength (local versus global) of impacts, to ensure the correct long-term management measures are put into place (Halpern et al. 2007; Crain et al. 2009; Stuart-Smith et al. 2010; Strain et al. 2014). Whilst there are many ways to approach monitoring, and it is important to still seek more cost-effective and rapid monitoring approaches, these need to be consistent with the “Edgar-Barrett” approach. The “Edgar-Barrett” surveys are an established and consistent methodology and have been applied in local, regional and state-wide monitoring for many years. They provide an invaluable benchmark for any local studies and a means to place monitoring data in a broader spatial and temporal context.

Understanding variability across spatial and temporal scales is critical as stressors can have antagonistic or synergistic effects on community assemblages affecting species relationships, and food-web complexity, implementing regime shifts, and shaping functional diversity (Crain et al. 2009; Strain et al. 2014). Whilst a rapid assessment technique is an important tool to enable early warning of change and to implement adaptive management, regional and historical biodiversity assessments provide the capacity to contextualise that change, providing an understanding of potential impacts relative to background or natural variation. Although a targeted technique may indicate a change in a reef ecosystem due to organic enrichment, it is only through associated biodiversity surveys that a full understanding of the extent and nature of that change can occur. Thus, we would recommend that although baseline biodiversity studies may be expensive, they are necessary to accurately evaluate the risk of adverse impacts of anthropogenic influences such as salmonid aquaculture on temperate rocky reef ecosystems.

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Appendix 1 – 10 m data

Table 6.4-8 North Bruny 10 m algal data

		Site				
Guild	**Species**	**BBN**	**TBN**	**TBS**	**VBS**	**CQE**
Canopy-forming algae	*Durvillaea potatorum*	0.0	0.0	0.0	0.0	0.0
	Ecklonia radiata	30.8	19.4	26.8	18.0	23.6
	Lessonia corrugata	6.8	0.0	0.0	0.0	0.0
	Macrocystis pyrifera	0.6	0.0	0.0	0.0	0.0
	Phyllospora comosa	0.0	54.2	52.2	58.6	51.4
Canopy forming FG%		**38.2**	**73.6**	**79.0**	**76.6**	**75.0**
Understory brown algae	*Acrocarpia paniculata*	8.8	0.8	0.0	0.4	0.4
	Carpoglossum confluens	27.0	8.0	8.2	7.0	5.0
	Carpomitra costata	0.0	0.2	0.0	0.0	2.2
	Caulocystis uvifera	0.0	0.0	0.0	0.0	0.0
	Colpomenia sinuosa	0.0	0.0	0.0	0.0	0.0
	Cystophora moniliformis	0.0	0.0	0.0	0.0	0.0
	Cystophora platylobium	0.0	0.0	0.0	0.0	0.0
	Cystophora xiphocarpa	0.0	0.0	0.0	0.0	0.0
	Dictyopteris muelleri	0.0	0.0	0.0	0.0	0.0
	Dictyota spp.	0.0	0.0	0.0	0.0	0.0
	Dilophus marginatus	0.0	0.0	0.0	0.0	0.0
	Halopteris paniculata	0.2	0.0	1.6	0.2	0.0
	Lobophora variegata	0.0	1.2	4.6	0.8	0.2
	Perithalia caudata	0.0	0.0	0.0	0.0	0.0
	Sargassum fallax	0.0	0.0	0.0	0.0	0.0
	Sargassum lacerifolium	0.0	0.0	11.2	0.0	0.0
	Sargassum spp.	0.0	0.6	0.6	0.0	0.0
	Sargassum verruculosum	0.0	0.0	0.0	1.0	0.0
	Sargassum vestitum	0.6	0.0	0.0	0.0	0.0
	Unidentified algae (filamentous brown)	0.0	0.0	0.0	0.0	0.0
	Xiphophora gladiata	3.8	0.0	0.0	0.0	0.0
	Zonaria turneriana/angustata	0.0	1.4	5.2	1.0	3.6
Understory brown algae FG%		40.4	12.2	31.4	10.4	11.4
Understory green algae	*Bryopsis* spp.	0.0	0.0	0.0	0.0	0.0
	Caulerpa flexilis	0.0	0.0	0.0	0.0	0.0
	Caulerpa hodgkinsoniae	0.0	0.0	0.0	1.6	2.6
	Caulerpa longifolia	0.0	0.0	0.0	0.0	0.0
	Caulerpa scalpelliformis	0.0	0.0	0.0	0.0	0.0
	Caulerpa simpliciuscula	0.0	0.0	0.0	0.0	0.0
	Caulerpa spp. (rhizomes)	0.0	0.0	4.4	3.2	0.0
	Caulerpa trifaria	0.0	0.4	3.0	1.4	0.2
	Chaetomorpha coliformis	0.2	0.0	0.0	0.0	0.0
	Chaetomorpha spp.	0.0	0.0	0.0	0.0	0.0
	Cladophora spp.	0.0	0.0	0.0	0.0	0.0
	Codium australicum	0.0	0.0	0.0	0.0	3.4
	Codium dimorphum	0.0	0.2	0.0	0.0	0.0
	Codium spp.	0.0	0.0	0.0	1.2	0.6
	Ulva spp.	0.0	0.0	0.0	0.0	0.0
Understory green algae FG%		0.2	0.6	7.4	7.4	6.8
Understory red algae	*Areschougia* spp.	0.2	0.2	0.2	0.2	0.8
	Ballia callitricha	5.8	10.0	2.6	1.4	1.2
	Ballia scoparia	1.4	4.2	0.6	2.2	3.6

	Site	**BBN**	**TBN**	**TBS**	**VBS**	**CQE**
Understory red algae	*Callophyllis rangiferina*	5.4	0.0	0.0	0.6	0.0
	Callophyllis spp.	1.4	0.6	0.2	0.0	0.0
	Champia spp.	0.0	0.0	0.0	0.0	0.0
	Chondria incrassata	0.0	0.0	0.0	0.0	0.0
	Corallina officinalis	0.0	0.0	0.0	0.0	0.0
	Craspedocarpus ramentaceus	1.6	0.0	0.0	0.0	0.0
	Delisea plumosa	2.6	0.0	0.0	0.0	0.0
	Delisea pulchra	0.0	0.0	0.0	0.0	0.0
	Dictyomenia harveyana	0.0	0.0	0.0	0.0	0.0
	Echinothamnion hystrix	0.0	0.0	0.0	0.0	0.0
	Echinothamnion spp.	0.0	0.0	0.0	0.0	0.0
	Erythroclonium sonderi	0.0	0.0	0.0	0.0	0.0
	Euptilota articulata	0.2	0.0	0.0	0.0	0.6
	Gelidium asperum	0.0	0.0	0.0	0.0	0.0
	Gigartina spp.	0.0	0.0	0.0	0.0	0.0
	Gracilaria spp.	0.0	0.0	0.0	0.0	0.0
	Haliptilon roseum	0.2	0.0	0.2	0.0	3.2
	Hemineura frondosa	0.0	0.0	0.0	0.2	0.4
	Hypnea ramentacea	0.0	0.0	0.0	0.0	0.0
	Hypnea ramentacea epiphytic	0.0	0.0	0.0	0.0	0.0
	Laurencia spp.	0.0	0.0	0.4	0.0	0.0
	Lenormandia marginata	4.8	3.4	5.0	0.8	2.4
	Mastophoropsis canaliculata	0.0	0.0	0.2	0.0	0.0
	Nizymenia conferta	0.0	0.0	0.0	0.0	0.0
	Peyssonnelia novaehollandiae	0.0	6.8	3.0	8.0	0.0
	Phacelocarpus apodus	0.6	0.0	0.0	0.0	0.2
	Phacelocarpus peperocarpos	4.8	2.2	1.4	0.0	0.0
	Plocamium angustum	4.8	8.6	4.6	0.4	1.6
	Plocamium cartilagineum	0.0	0.0	0.0	0.0	0.0
	Plocamium costatum	0.0	0.0	0.0	0.0	0.0
	Plocamium dilatatum	3.2	0.2	1.0	5.6	0.8
	Plocamium mertensii	0.6	0.0	0.0	0.0	0.0
	Plocamium patagiatum	0.0	0.0	0.0	0.0	0.0
	Plocamium spp.	0.0	0.6	0.0	0.0	0.0
	Polluxfenia lobata	1.0	0.0	0.0	0.0	0.0
	Polyopes constrictus	2.4	0.4	0.2	0.6	0.8
	Polysiphonia spp.	0.4	0.0	0.0	0.4	0.0
	Ptilonia australasica	1.8	1.4	0.0	0.8	0.6
	Rhodymenia prolificans	0.0	0.0	0.0	0.0	0.0
	Rhodymenia spp.	5.4	9.6	14.2	3.8	10.8
	Sonderopelta coriacea	0.0	0.0	0.0	0.0	0.0
	Sonderopelta/Peyssonnelia	0.0	0.0	0.0	0.0	0.4
	Stenogramme interrupta	0.0	0.0	0.0	0.0	0.8
	Synarthrophyton patena	0.0	0.0	0.0	0.0	0.0
	Thamnoclonium dichotomum	0.0	0.4	0.0	0.2	0.0
	Unidentified algae (filamentous red)	0.0	4.2	0.0	1.4	0.0
	Unidentified algae (foliose red)	0.8	0.0	0.0	0.2	0.0
	Unidentified algae (red)	0.0	0.0	0.0	0.0	0.0
Understory red algae FG%		**49.4**	**52.8**	**33.8**	**26.8**	**28.2**
Encrusting algae	*Peyssonnelia* spp. (encrusting)	7.4	15.8	18.6	10.4	7.0
	Unidentified algae (crustose coralline)	19.2	27.4	13.0	22.6	5.0

	Encrusting algae FG%	**26.6**	**43.2**	**31.6**	**33.0**	**12.0**
Sessile invertebrates	**Site**	**BBN**	**TBN**	**TBS**	**VBS**	**CQE**
	Amathia wilsoni	0.0	0.0	0.0	0.2	0.0
	Corynactis australis	0.0	0.0	0.0	0.0	0.0
	Culicia spp.	0.0	0.0	0.0	0.0	0.0
	Erythropodium hicksoni	0.0	0.0	0.0	0.0	0.0
	Galeolaria spp.	0.2	0.0	0.2	0.0	0.0
	Maoricolpus roseus	0.0	0.0	0.0	0.0	0.0
	Mytilus galloprovincialis	0.0	0.0	0.0	0.0	0.0
	Ostrea angasi	0.0	0.0	0.0	0.0	0.0
	Pyura australis	0.2	0.0	0.0	0.2	0.0
	Pyura gibbosa	0.0	0.0	0.0	0.0	0.0
	Pyura stolonifera	0.2	0.0	0.0	0.0	0.4
	Unidentified ascidians	2.0	0.8	0.0	0.0	0.0
	Unidentified ascidians (encrusting)	0.0	0.0	0.0	0.0	1.4
	Unidentified bryozoans	0.8	1.8	4.0	2.4	0.0
	Unidentified bryozoans (hard)	0.0	0.0	0.0	0.0	0.0
	Unidentified bryozoans (soft)	3.8	7.0	7.0	7.8	3.2
	Unidentified epiphytic bryozoans	0.0	0.0	0.0	0.0	0.0
	Unidentified hydroid	5.6	0.0	0.0	0.0	0.0
	Unidentified invertebrates (encrusting)	6.0	0.0	0.0	0.0	0.0
	Unidentified sponge (encrusting)	23.4	3.2	9.0	7.0	16.6
	Unidentified sponges	1.8	0.6	1.4	3.4	2.8
	Sessile invertebrates FG%	**44.0**	**13.4**	**21.6**	**21.0**	**24.4**
Substrate	Bare rock (non - barrens)	0.8	0.0	0.0	0.6	1.4
	Heterozostera nigricaulis	0.0	0.0	0.0	0.0	0.0
	Sand	6.4	2.4	8.2	23.6	11.6
	Turf/sand/sediment matrix	0.0	0.0	0.0	0.0	14.8
	Unidentified algae (brown turf)	0.0	0.0	0.6	7.8	0.0
	Unidentified algae (green turf)	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (red turf)	0.4	0.0	0.4	0.6	6.6
	Unidentified algae (turf)	0.0	0.0	0.0	0.0	0.0
	Substrate FG%	**7.6**	**2.4**	**9.2**	**32.6**	**34.4**
	Algal species richness	**19**	**21**	**28**	**21**	**21**

Table 6.4-9 North Bruny Invertebrates 10 m

	Site				
Species	**BBN**	**TBN**	**TBS**	**VBS**	**CQE**
Agnewia tritoniformis (Murex shell)	0.0	1.0	0.0	0.0	0.0
Amblypneustes ovum (Short-spined urchin)	0.0	0.0	0.0	0.0	0.0
Amoria undulata (Wavy volute)	0.0	0.0	0.0	0.0	1.0
Australostichopus mollis (Sea cucumber)	1.0	0.0	0.0	1.0	1.0
Cabestana spengleri (Triton shell)	0.0	0.0	0.0	0.0	0.0
Centrostephanus rodgersii (Long-spined urchin)	0.0	0.0	0.0	0.0	0.0
Comanthus tasmaniae (Feather star)	0.0	0.0	0.0	0.0	0.0
Comanthus trichoptera (Feather star)	30.5	106.5	112.5	215.0	31.0
Coscinasterias muricate (Eleven-arm star)	0.0	0.0	0.0	0.0	0.0
Dicathais orbita (Dog whelk)	0.0	0.0	0.0	0.0	0.0
Fromia polypore (Many-spotted seastar)	0.0	0.0	0.0	0.0	0.0
Goniocidaris tubaria (Pencil urchin)	0.5	0.0	0.0	0.0	0.0
Haliotis rubra (Black-lip abalone)	2.5	2.5	1.0	3.5	2.0
Heliocidaris erythrogramma (Purple urchin)	0.0	0.0	16.5	1.5	1.0
Jasus edwardsii (Southern rock lobster)	1.0	5.5	2.0	0.5	0.0
Maoricolpus roseus (New Zealand screw shell)	120.0	0.0	5.0	0.0	0.0
Nectocarcinus integrifrons (Red swimmer crab)	0.0	0.0	0.0	0.0	0.0
Nectocarcinus tuberculosus (Velvet crab)	0.0	0.0	0.0	0.0	0.0
Nectria ocellate (Ocellate seastar)	0.0	2.5	1.0	0.0	0.0
Octopus maorum (Maori octopus)	0.0	0.0	0.0	0.0	1.0
Ostrea angasi (Native oyster)	0.0	0.0	0.0	0.0	0.0
Pagurid spp. (Hermit crab)	0.0	0.0	0.0	0.0	0.0
Paragrapsus laevis (Shore crab)	2.5	0.0	0.0	0.0	0.0
Patiriella regularis (Regular seastar)	0.0	0.0	0.0	0.0	0.0
Penion maximus (Giant whelk)	1.0	0.0	0.0	0.0	0.0
Penion spp. (Whelk)	0.0	0.0	0.0	0.0	0.0
Petricia vernicina (Velvet seastar)	0.5	2.0	0.0	0.0	0.0
Phasianotrochus eximius (Giant kelp shell)	0.0	0.0	0.0	0.0	0.0
Plagusia chabrus (Red bait crab)	0.0	2.5	0.0	0.0	0.0
Pleuroploca australasia (Tulip shell)	0.5	0.0	0.0	0.0	0.0
Ranella australasia (Australian Rock Whelk)	0.0	0.0	0.0	0.0	0.0
Sassia parkinsonia (Trumpet shell)	0.0	0.0	0.0	0.0	0.0
Strigopagurus strigimanus (Rasping hermit crab)	0.0	0.0	1.5	0.0	0.0
Tosia australis (Southern biscuit star)	0.5	0.0	0.0	0.0	0.0
Tosia magnifica (Magnificent biscuit star)	0.0	1.0	0.0	0.0	0.0
Turbo undulatas (Turban shell)	0.0	0.0	0.0	0.0	0.0
Unidentified mollusc	0.0	0.0	0.5	0.0	0.0
Uniophora granifera (Granular seastar)	0.0	0.0	0.0	0.0	0.0
Total abundance	**160.5**	**123.5**	**140**	**221.5**	**37**
Species Richness	**11**	**8**	**7**	**5**	**6**

Table 6.4-10 North Bruny Fish 10 m

	Site				
Species	**BBN**	**TBN**	**TBS**	**VBS**	**CQE**
Acanthaluteres vittiger (Toothbrush leatherjacket)	0.0	91.5	60.5	100.0	0.0
Aplodactylus arctidens (Marblefish)	0.0	0.0	0.0	0.0	0.0
Aracana aurita (Shaw's cowfish)	0.0	0.0	0.0	0.0	0.0
Atypichthys strigatus (Mado sweep)	6.5	20.0	0.0	1.0	0.0
Caesioperca lepidoptera (Butterfly perch)	0.0	0.0	3.5	0.0	0.0
Caesioperca razor (Barber perch)	2.0	0.0	0.0	0.0	0.0
Cephaloscyllium laticeps (Draughboard shark)	0.0	0.0	0.0	0.5	0.0
Cheilodactylus spectabilis (Banded morwong)	0.0	1.0	0.0	0.0	2.0
Conger verreauxi (Southern conger)	0.0	0.0	0.0	0.0	0.0
Dasyatis brevicaudata (Smooth stingray)	0.0	0.0	0.0	0.0	0.0
Dinolestes lewini (Long-fin pike)	0.0	19.0	0.5	0.0	0.0
Diodon nicthemerus (Globe fish)	0.0	0.0	1.0	0.0	0.0
Dotalabrus aurantiacus (Castelnaus wrasse)	0.0	0.0	0.0	0.0	0.0
Forsterygion varium (Variable threefin)	6.5	8.0	1.5	0.0	6.0
Gnathanacanthus goetzeei (Red velvetfish)	0.0	0.0	0.0	0.0	0.0
Helicolenus percoides (Reef ocean perch)	0.0	0.0	0.0	0.0	0.0
Heteroclinus perspicillatus (Common weedfish)	0.0	0.0	0.0	0.0	0.0
Heteroclinus tristis (Longnose weedfish)	0.0	0.0	0.0	0.0	0.0
Heteroscarus acroptilus (Rainbow cale)	0.5	0.0	0.0	0.0	0.0
Latridopsis forsteri (Bastard trumpeter)	0.0	0.0	0.0	0.0	2.0
Lotella rhacina (Beardie)	0.0	0.0	0.0	0.5	0.0
Meuschenia australis (Brown-striped leatherjacket)	0.0	1.0	0.0	0.5	3.0
Meuschenia freycineti (Six-spine leatherjacket)	0.0	0.0	0.0	0.0	0.0
Neoodax balteatus (Little rock-whiting)	0.0	0.0	0.0	0.0	0.0
Notolabrus fucicola (Purple wrasse)	1.0	1.5	1.0	2.5	7.0
Notolabrus tetricus (Blue-throat wrasse)	11.5	4.0	8.0	29.0	7.0
Olisthops cyanomelas (Herring cale)	0.0	0.5	1.0	1.0	0.0
Parequula melbournensis (Silverbelly)	0.0	0.0	0.0	0.0	0.0
Parma microlepis (White-ear)	0.0	0.0	0.0	0.0	1.0
Parma victoriae (Victorian scalyfin)	0.0	0.0	0.0	0.0	0.0
Pempheris multiradiata (Common bullseye)	25.5	0.0	0.0	0.0	10.0
Pentaceropsis recurvirostris (Long-snouted boarfish)	0.0	0.0	0.0	0.0	0.0
Pictilabrus laticlavius (Senator wrasse)	1.5	1.0	0.5	1.0	0.0
Pseudocaranx dentex (White trevally)	0.0	0.0	0.0	0.5	0.0
Pseudolabrus rubicundus (Rosy wrasse)	0.0	0.0	1.0	1.5	0.0
Scorpaena papillosa (Southern Rock Cod)	1.0	1.0	1.5	0.5	0.0
Scorpis aequipinnis (Sea sweep)	0.0	0.0	0.0	0.0	0.0
Scorpis lineolate (Silver sweep)	0.0	0.0	0.0	0.0	0.0
Siphonognathus beddomei (Pencil weed whiting)	4.0	0.5	1.5	0.5	0.0
Tasmanoglobius gloveri (Glover's Tasman goby)	0.0	0.0	0.0	0.0	0.0
Trachinops caudimaculatus (Hulafish)	75.0	10.0	133.5	150.0	0.0
Trinorfolkia clarkei (Common triplefin)	0.0	0.0	0.0	0.0	0.0
Upeneichthys vlamingii (Southern goatfish)	0.0	0.0	0.0	1.0	0.0
Total Abundance	**135**	**159**	**215**	**290**	**38**
Species Richness	**11**	**13**	**13**	**15**	**8**

Table 6.4-11 Southeast Channel 10 m algae

		Site					
Guild	**Species**	**ZUID**	**SCOT**	**LIPP**	**TOWE**	**STIS**	**ACTA**
Canopy-forming algae	*Durvillaea potatorum*	0.0	0.0	0.0	0.0	0.0	0.0
	Ecklonia radiata	24.0	12.4	11.6	23.0	21.0	11.4
	Lessonia corrugata	0.0	0.0	0.0	0.0	0.0	0.0
	Macrocystis pyrifera	0.0	0.0	0.0	0.0	0.0	1.2
	Phyllospora comosa	0.0	55.2	51.6	35.6	50.2	0.0
	Canopy forming FG%	**24.0**	**67.6**	**63.2**	**58.6**	**71.2**	**12.6**
Understory brown algae	*Acrocarpia paniculata*	0.0	0.0	2.4	0.0	3.4	19.2
	Carpoglossum confluens	0.0	13.2	21.0	4.0	5.6	7.2
	Carpomitra costata	0.0	0.0	0.0	0.0	0.6	1.0
	Cystophora monilifera	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora moniliformis	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora platylobium	0.0	1.4	1.8	2.4	0.0	1.8
	Cystophora retorta	0.0	0.0	0.0	0.0	0.0	0.0
	Cystophora retroflexa	0.0	0.0	0.0	0.0	0.0	0.0
	Dictyomenia tridens	0.0	0.0	0.0	0.2	0.0	0.0
	Dictyopteris muelleri	0.0	0.0	0.0	0.0	0.0	0.0
	Halopteris paniculata	0.0	0.0	2.4	0.0	1.0	0.4
	Halopteris spp.	0.0	0.0	0.0	0.6	0.0	0.0
	Lobophora variegata	0.0	0.8	0.0	0.0	1.6	0.0
	Perithalia caudata	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum fallax	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum paradoxum	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum spp.	0.0	0.4	0.6	0.0	0.4	0.0
	Sargassum spp. (subgenus *Arthrophycus*)	0.0	0.0	0.0	0.0	0.0	0.0
	Sargassum verruculosum	0.0	0.0	0.0	0.6	3.0	0.0
	Sargassum vestitum	0.0	0.0	0.0	0.0	0.0	0.0
	Seirococcus axillaris	0.0	3.0	0.0	0.0	2.2	0.0
	Sporochnus comosus	0.0	0.0	0.0	0.0	0.0	0.0
	Sporochnus radiciformis	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (filamentous brown)	0.0	0.0	0.0	0.0	0.0	0.0
	Xiphophora gladiata	0.0	0.0	0.0	0.8	0.0	0.2
	Zonaria turneriana/angustata	0.0	1.8	0.6	1.0	3.4	0.6
	Understory brown algae FG%	**0.0**	**20.6**	**28.8**	**9.6**	**21.2**	**30.4**
Understory green algae	*Bryopsis* spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa brownii	0.0	0.0	0.4	0.0	0.0	1.0
	Caulerpa flexilis	0.0	0.0	0.0	0.2	0.0	0.0
	Caulerpa geminata	0.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa scalpelliformis	0.0	0.0	0.0	9.6	0.0	0.0
	Caulerpa simpliciuscula	1.0	0.0	0.0	0.0	0.0	0.0
	Caulerpa spp. (rhizomes)	0.0	0.0	0.0	0.0	0.0	1.8
	Caulerpa trifaria	34.6	1.6	0.0	0.8	0.0	0.0
	Chaetomorpha spp.	0.0	0.0	0.2	0.0	0.0	0.0
	Cladophora spp.	0.6	0.8	0.0	0.0	0.0	0.0
	Codium fragile	0.0	0.0	0.0	0.0	0.0	0.0
	Codium pomoides	0.0	0.4	0.0	0.0	0.0	0.0
	Codium spp.	0.0	0.0	0.0	0.0	0.0	0.2
	Ulva spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Understory green algae FG%	**36.2**	**2.8**	**0.6**	**10.6**	**0.0**	**3.0**

	Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**
Understory red algae	*Areschougia* spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Asparagopsis armata	1.4	0.0	0.0	0.0	1.0	0.0
	Ballia callitricha	0.0	1.0	6.4	1.4	1.0	0.2
	Ballia scoparia	0.0	0.6	2.6	0.0	0.0	0.4
	Callophyllis lambertii	0.8	10.6	3.0	9.0	1.0	6.0
	Callophyllis rangiferina	4.0	2.6	3.2	6.4	0.2	10.8
	Ceramium spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Champia spp.	0.0	0.6	0.2	1.4	0.2	0.4
	Chondria spp.	0.0	0.0	0.0	0.0	0.2	0.0
	Corallina officinalis	0.0	3.8	3.0	0.2	1.8	2.2
	Craspedocarpus ramentaceus	0.2	0.2	0.0	0.6	1.0	0.0
	Delisea plumosa	1.2	0.8	0.0	0.0	0.0	0.0
	Delisea pulchra	0.0	0.0	2.4	0.0	0.0	0.0
	Echinothamnion hystrix	0.0	1.2	3.0	0.0	0.0	0.2
	Echinothamnion spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Erythrymenia minuta	0.0	1.2	0.0	0.0	0.0	0.0
	Euptilota articulata	0.8	0.4	1.8	0.0	1.0	1.2
	Euptilota sp.2	0.0	0.0	0.0	0.0	0.0	0.0
	Gelidium australe	0.0	0.0	0.0	0.0	0.0	0.0
	Haliptilon roseum	0.0	1.6	0.8	0.6	0.2	0.2
	Haloplegma preissii	0.0	0.0	0.0	0.0	0.0	0.0
	Hemineura frondosa	0.8	0.8	1.8	0.8	0.8	1.4
	Hypnea ramentacea	0.0	0.0	0.0	0.0	0.0	1.0
	Kallymenia cribrosa	1.0	0.0	0.0	0.0	0.0	0.0
	Kallymenia spp.	2.6	0.0	0.0	0.0	0.0	0.0
	Kallymenia tasmanica	0.0	0.0	0.0	0.0	0.0	0.0
	Laurencia elata	0.0	0.0	0.0	0.0	0.0	0.6
	Laurencia spp.	0.6	0.0	0.0	0.0	0.0	0.0
	Lenormandia marginata	2.6	1.8	2.4	1.2	4.8	0.0
	Mastophoropsis canaliculata	0.0	0.0	1.0	0.0	1.2	0.0
	Melanthalia obtusata	0.0	0.0	0.0	0.8	0.0	0.0
	Metagoniolithon radiatum	0.0	0.0	0.0	0.0	0.0	0.0
	Mychodea acanthymennia	0.0	0.4	0.4	0.2	0.0	0.4
	Mychodea acicularis	0.0	0.0	0.0	0.4	0.0	0.0
	Myriogramme gunniana	3.0	0.0	0.0	0.0	0.0	0.0
	Nitospinosa tasmanica	0.8	1.8	0.8	0.6	0.4	0.0
	Nizymenia australis	0.0	0.0	0.8	0.8	0.0	0.0
	Peyssonnelia novaehollandiae	4.2	5.6	0.0	3.4	2.8	0.6
	Phacelocarpus apodus	1.0	0.0	6.2	1.0	2.2	1.6
	Phacelocarpus peperocarpos	1.4	0.0	4.6	4.0	0.0	0.0
	Phacelocarpus spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Plocamium angustum	1.2	1.4	2.4	2.8	2.0	10.8
	Plocamium costatum	0.6	0.0	0.6	0.0	0.0	0.0
	Plocamium dilatatum	0.0	6.2	5.2	11.2	5.8	26.4
	Plocamium mertensii	0.0	3.0	0.0	0.4	1.0	4.0
	Pollexfenia lobata	5.6	0.4	0.0	0.0	0.0	0.0
	Polyopes constrictus	0.0	0.2	0.4	0.8	0.6	0.0
	Ptilonia australasica	2.4	1.2	0.0	3.2	0.0	0.0
	Rhodymenia sonderi	2.4	1.0	0.0	0.8	0.0	0.0
	Rhodymenia spp.	0.6	1.6	3.0	4.8	8.6	4.6
	Sarcodia marginata	0.6	0.4	0.0	0.0	0.0	0.0

	Species	ZUID	PENG	REDC	ESPE	LOMA	SCOT
Understory red algae	*Sonderopelta coriacea*	0.0	0.0	4.4	3.0	2.8	0.0
	Sonderopelta/Peyssonnelia	0.0	0.0	0.0	0.0	0.0	0.0
	Stenogramme interrupta	1.2	0.4	1.0	0.0	0.0	0.0
	Synarthrophyton patena	0.0	0.0	0.0	0.2	0.0	0.0
	Thamnoclonium dichotomum	4.0	0.0	0.0	0.4	2.2	0.0
	Unidentified algae (filamentous red)	0.4	1.0	0.0	0.0	0.0	0.6
	Unidentified algae (foliose red)	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (red)	0.0	0.0	0.0	0.0	0.0	0.0
	Understory red algae FG%	**45.4**	**51.8**	**61.4**	**60.4**	**42.8**	**73.6**
Encrusting algae	*Peyssonnelia* spp. (encrusting)	**17.4**	**14.0**	**36.8**	**19.8**	**16.6**	**6.4**
	Unidentified algae (crustose coralline)	17.2	29.8	20.6	29.8	10.4	14.0
	Unidentified algae (encrusting brown)	0.0	0.0	0.0	0.0	0.0	0.0
	Encrusting algae FG%	**34.6**	**43.8**	**57.4**	**49.6**	**27.0**	**20.4**
Sessile invertebrates	Amathia wilsoni	0.0	0.0	0.0	0.0	0.0	0.0
	Erythropodium hicksoni	0.0	0.0	0.0	0.0	0.0	0.0
	Galeolaria spp.	1.0	0.0	0.0	0.0	0.0	0.0
	Ostrea angasi	0.2	0.0	0.0	0.0	0.0	0.0
	Parazoanthus spp.	0.0	0.0	0.0	0.0	0.0	0.0
	Pyura australis	0.0	0.0	0.0	0.0	0.0	0.0
	Pyura stolonifera	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified ascidians	0.2	0.0	0.0	0.0	0.0	0.0
	Unidentified ascidians (encrusting)	1.8	0.0	0.0	0.4	0.0	0.0
	Unidentified bryozoans	2.4	1.2	7.4	7.2	9.4	2.0
	Unidentified bryozoans (hard)	0.0	0.0	2.2	0.8	2.0	2.6
	Unidentified bryozoans (soft)	1.6	1.6	0.0	2.4	0.0	0.0
	Unidentified hydroid	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified invertebrates (encrusting)	0.0	0.0	0.0	0.4	0.0	0.0
	Unidentified sponge (encrusting)	9.6	5.0	6.4	3.6	12.6	5.0
	Unidentified sponges	5.2	0.0	1.0	1.2	3.6	1.4
	Sessile invertebrates FG%	**22.0**	**7.8**	**17.0**	**16.0**	**27.6**	**11.0**
Substrate	Bare rock (non - barrens)	0.0	1.6	0.0	1.6	0.0	0.0
	Cobble	0.0	2.4	0.0	0.0	0.0	0.0
	Gravel	0.0	0.0	0.0	0.0	0.0	0.0
	Sand	18.2	19.4	7.8	7.8	14.4	10.8
	Shell	1.0	0.0	0.0	0.0	0.0	0.0
	Turf/sand/sediment matrix	1.8	7.2	2.2	4.0	10.0	9.4
	Unidentified algae (brown turf)	0.0	0.0	0.0	0.0	0.0	0.0
	Unidentified algae (red turf)	1.6	1.6	4.4	4.0	2.6	4.6
	Substrate FG%	**22.6**	**32.2**	**14.4**	**17.4**	**27.0**	**24.8**
	Algal species richness	**28**	**34**	**31**	**36**	**30**	**27**

Table 6.4-12 Southeast Channel 10 m Invertebrates

	Site				
Species	**ZUID**	**SCOT**	**LIPP**	**TOWE**	**ACTA**
Amblypneustes ovum (Short-spined urchin)	0.0	0.0	0.0	0.0	0.0
Australostichopus mollis (Sea cucumber)	0.0	0.0	0.0	0.0	1.0
Cabestana spengleri (Triton shell)	0.0	0.0	0.0	0.0	0.0
Cabestana tabulate (Fringed triton)	0.0	0.0	0.0	0.0	0.0
Ceratosoma amoenun (Sweet ceratosoma)	0.0	0.0	0.0	0.0	0.0
Ceratosoma brevicaudatum (Short-tailed nudibranch)	0.0	0.0	0.0	0.0	0.0
Comanthus tasmaniae (Feather star)	0.0	0.0	0.0	0.0	0.0
Comanthus trichoptera (Feather star)	0.0	0.0	0.0	73.0	80.5
Coscinasterias muricate (Eleven-armed star)	0.0	0.0	0.0	0.0	0.0
Dicathais orbita (Dog whelk)	0.0	0.0	0.0	0.0	0.0
Diogenid spp. (Hermit crab)	0.0	0.0	0.0	0.0	0.0
Fromia polypore (Many-spotted seastar)	0.0	0.0	0.0	0.0	0.0
Goniocidaris tubaria (Pencil urchin)	0.0	0.0	0.0	0.0	0.0
Haliotis rubra (Black-lip abalone)	0.0	0.0	0.0	3.5	0.5
Heliocidaris erythrogramma (Purple urchin)	0.0	0.0	0.0	0.0	0.0
Jasus edwardsii (Southern rock lobster)	0.0	0.0	0.0	0.0	0.5
Meridionale ambigua (Yellow sea spider)	0.0	0.0	0.0	0.0	0.0
Nectocarcinus tuberculosus (Velvet crab)	0.0	0.0	0.0	0.0	0.0
Nectria ocellate (Ocellate seastar)	0.0	0.0	0.0	0.0	0.0
Ostrea angasi (Native oyster)	0.0	0.0	0.0	0.0	0.0
Penion maximus (Giant whelk)	0.0	0.0	0.0	0.0	0.0
Penion spp. (Whelk)	0.0	0.0	0.0	0.0	0.0
Petricia vernicina (Velvet star)	0.0	0.0	0.0	2.5	0.0
Plagusia chabrus (Red bait crab)	0.0	0.0	0.0	0.0	0.0
Pleuroploca australasia (Tulip shell)	0.0	0.5	0.0	0.0	0.0
Ranella australasia (Australian rock whelk)	0.5	0.0	0.0	0.0	0.0
Turbo undulatas (Turban shell)	1.5	0.0	0.0	0.0	0.0
Total Abundance	**222.0**	**59.0**	**99.0**	**79.0**	**82.5**
Species Richness	**6**	**7**	**6**	**3**	**4**

Table 6.4-13 Southeast Channel 10 m Fish

	Site					
Species	**ZUID**	**PENG**	**REDC**	**ESPE**	**LOMA**	**SCOT**
Acanthaluteres vittiger (Toothbrush leatherjacket)	0.0	0.0	0.0	0.0	5.5	0.0
Aplodactylus arctidens (Marblefish)	0.0	0.0	0.0	0.0	0.0	0.0
Aracana aurita (Shaw's cowfish)	0.0	0.0	0.0	0.0	0.0	0.0
Bovichtus angustifrons (Dragonet)	0.0	0.0	0.0	0.0	0.0	0.0
Caesioperca razor (Barber perch)	14.5	0.0	0.0	0.0	0.0	0.0
Cephaloscyllium laticeps (Draughtboard shark)	2.5	0.5	0.0	0.0	0.0	0.0
Cheilodactylus nigripes (Magpie perch)	0.0	0.0	0.5	0.0	0.0	0.0
Dinolestes lewini (Long-fin pike)	21.0	0.0	0.0	3.5	2.5	0.0
Diodon nicthemerus (Globefish)	0.0	0.0	0.0	0.0	1.0	0.0
Forsterygion varium (Variable three-fin)	3.0	0.5	1.5	0.0	0.0	0.0
Genypterus tigerinus (Rock ling)	0.0	0.0	0.0	0.0	0.0	0.0
Heteroclinus johnstoni (Johnston's Weedfish)	0.0	0.0	0.0	0.0	0.0	0.0
Heteroclinus perspicillatus (Common Weedfish)	0.0	0.0	0.0	0.5	0.0	0.0
Heteroclinus tristis (Longnose Weedfish)	0.0	0.0	0.0	0.0	0.0	0.0
Latridopsis forsteri (Bastard trumpeter)	0.0	0.0	0.0	0.0	0.0	0.0
Lotella rhacina (Beardie)	0.0	0.0	0.0	0.0	1.0	0.0
Meuschenia australis (Brownstriped leatherjacket)	1.0	0.0	0.0	0.5	2.5	0.0
Meuschenia freycineti (Sixspine leatherjacket)	0.0	0.0	0.0	0.0	0.0	0.0
Nemadactylus macropterus (Jackass morwong)	0.5	0.0	0.0	0.0	0.0	0.0
Neoodax balteatus (Little rock whiting)	0.5	0.5	0.0	0.0	0.0	0.0
Neosebastes scorpaenoides (Gurnard perch)	0.5	0.0	0.0	0.0	0.0	0.0
Notolabrus fucicola (Purple wrasse)	0.0	2.5	1.0	0.0	0.0	0.5
Notolabrus tetricus (Blue-throat wrasse)	16.0	16.0	5.0	10.5	2.0	9.5
Odax cyanomelas (Rainbow cale)	0.0	0.0	0.0	0.0	0.0	0.0
Parascyllium ferrugineum (Rusty catshark)	0.0	0.0	0.0	0.0	0.0	0.0
Parma microlepis (White-ear)	0.0	0.0	0.0	0.0	0.0	0.0
Pempheris multiradiata (Common bullseye)	0.0	0.0	0.5	0.0	1.5	0.0
Pentaceropsis recurvirostris (Long-snouted boarfish)	0.0	0.0	0.0	0.0	0.0	0.0
Pictilabrus laticlavius (Senator wrasse)	2.0	2.0	2.0	3.0	3.0	0.5
Pseudocaranx georgianus (Silver trevally)	0.0	0.0	0.0	0.0	0.0	0.0
Pseudolabrus rubicundus (Rosy wrasse)	2.0	0.0	0.0	0.0	0.0	2.0
Pseudophycis bachus (Red cod)	0.0	0.0	0.0	0.0	0.0	0.0
Pseudophycis barbata (Bearded Rock Cod)	0.0	0.0	0.0	0.0	0.0	0.0
Scorpaena papillosa (Southern Red Scorpionfish)	0.0	0.0	0.0	0.5	0.5	0.0
Scorpis aequipinnis (Sea sweep)	0.0	0.0	5.0	0.0	0.0	0.0
Scorpis lineolata (Silver sweep)	0.0	0.0	0.0	0.0	0.0	0.0
Siphonognathus beddomei (Pencil weed whiting)	0.0	0.0	0.0	0.0	1.0	0.0
Trachinops caudimaculatus (Hulafish)	503.5	0.0	0.0	88.0	108.0	0.0
Trinorfolkia clarkei (Common triplefin)	0.0	0.0	0.0	0.0	0.0	0.0
Urolophus cruciatus (Banded stingaree)	0.5	0.0	0.0	0.0	0.0	0.0
Total abundance	**567.5**	**22.0**	**15.5**	**106.5**	**128.5**	**12.5**
Species Richness	**13**	**6**	**7**	**7**	**11**	**4**

Appendix 2 – Rugosity and cryptic habitat PCOs

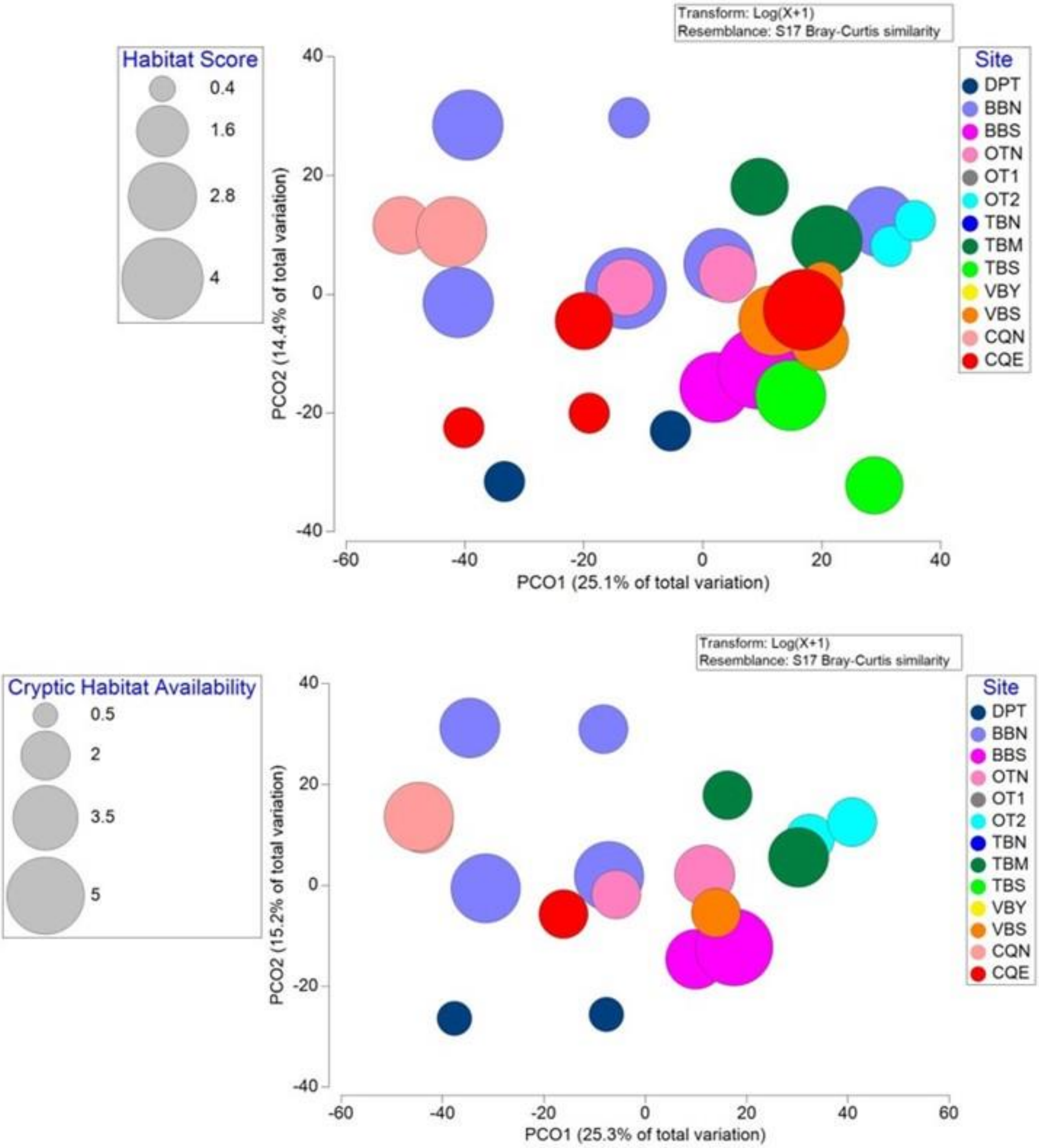

Figure 6.4-18 Principal Coordinate Analysis (PCO) on invertebrate community data (5 m) with a) reef habitat complexity and b) the availability of cryptic habitat overlain at 13 sites on North Bruny Island.

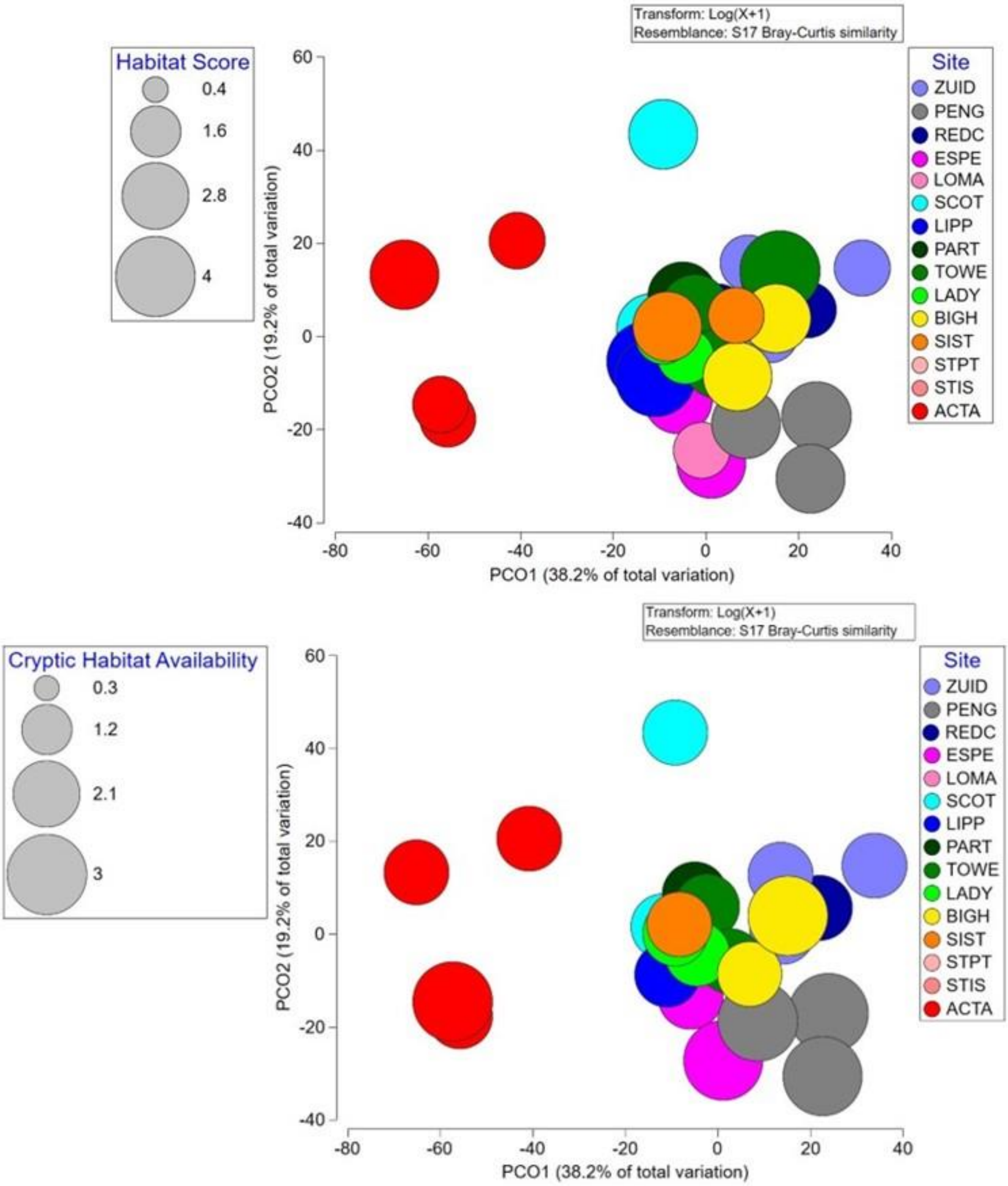

Figure 6.4-19 Principal Coordinate Analysis (PCO) on the categorical scores of (a) reef habitat complexity and (b) the availability of cryptic habitat for invertebrates from 15 sites in the SE Channel region during the 2017 surveys.

6.5 Functional evaluation of organic enrichment in temperate rocky reefs using a novel rapid assessment technique

6.5.1 Introduction

While terrestrially-derived pollution from coastal industry and urbanisation has long been recognised as a major source of organic enrichment to the marine environment (Airoldi & Beck 2007; Connell et al. 2008), feed-additive aquaculture is increasingly becoming another source of anthropogenic enrichment (Holmer et al. 2005; Hargrave et al. 2008; Wang et al. 2012). In many instances, coastal aquaculture is now occurring at densities with the potential to drive regional level change (Trujillo et al. 2012; Taranger et al. 2014). The process of organic enrichment is similar regardless of source, with effects occurring either directly from nutrient enrichment and increased sedimentation, or indirectly through trophic interactions and changes (Worm et al. 1999; Benedetti-Cecchi et al. 2001). While the process is well understood, the response of temperate reef ecosystems to organic enrichment is often highly variable and dependent upon local environmental factors. Temperate reef ecosystems are inherently complex in nature, as they are multi-dimensional habitats with multi-trophic linkages, often combined with a high degree of regional endemism, all of which can affect the response to organic enrichment (Connell & Irving 2008; Gorman et al. 2009). Sustained organic enrichment has been linked to wide-scale loss of macroalgae and the potential for phase shifts in ecosystem function (Graham 2004; Connell et al. 2008; Teagle et al. 2017). Identifying when temperate reef ecosystems are under stress and at risk of such devastating change represents a considerable challenge for management.

Whilst determining the trigger points for phase shifts is difficult, there are several common responses of temperate reef ecosystems to organic enrichment. The most extreme responses are loss of canopy forming kelp and proliferation of turfing algae (Eriksson et al. 2002; Connell et al. 2008). Turfing algae has the capacity to form its own biogenic matrix, trapping sand and sediment and effectively prohibiting the growth of larger perennial kelp species (Airoldi 2003; Gorman & Connell 2009). Other opportunistic algal types with fast growth rates, rapid reproduction and high demand for nitrogen also respond positively to organic enrichment (Oh et al. 2015). These include opportunistic green algae species from the genera *Ulva*, *Cladophora* and *Chaetomorpha* (Lavery & Mccomb 1991; Nelson et al. 2008), red algae such as *Asparagopsis armata* (Paul et al. 2006; Mata et al. 2010), and several filamentous and epiphytic algal species (Oh et al. 2015). While rapid growing algae can initially act as a nutrient sink buffering the ecosystem from the effects of organic enrichment, under eutrophic conditions, these algae can form dense blooms, significantly altering ecosystem structure and function (Nelson et al. 2008).

Other functional groups, including filter feeders such as sponges, ascidians or bryozoans, may initially benefit from low levels of organic enrichment, either directly through increases in particulate organic carbon, or through nutrients stimulating primary productivity in the water column (Hughes et al. 2005; Cook et al. 2006). However, most of these species will have threshold limits, above which levels of eutrophication will have a negative effect on populations. Increases in primary productivity and thus food resources at the base of the food chain may also stimulate populations of grazers (Miller & Hay 1996; Burkepile & Hay 2006). These increases in opportunistic species with high growth and reproductive capacity suggest temperate reef ecosystems follow an ecological response to organic enrichment similar to that established by Pearson and Rosenberg (1978) for soft sediments. However, the nature of the multi-dimensional habitats in temperate reefs makes their response inherently more complex as trophic interactions will occur over multiple levels. Therefore, understanding both the inherent resilience of these ecosystems and potential impact pathways is far more difficult.

In south-eastern Tasmania, aquaculture of Atlantic salmon (*Salmo salar*) is well established and currently in a phase of expansion (DPIPWE 2017). Historically, environmental monitoring of salmon farming has focused on localised point source impacts on soft sediments under or near the cages, or on broader scale nutrient monitoring in the nearby water column (Ross & Macleod 2013). However, as the industry continues to expand there is mounting concern regarding the potential for impacts on nearby reef ecosystems. Temperate reef ecosystems in south-eastern Tasmania have intrinsic biodiversity and conservation value, but also support high value commercial fisheries, such as abalone (*Haliotis rubra*) and rock lobster (*Jasus edwardsii*) and are important to the tourism and recreational sectors. For salmon farming to be sustainable in the long term, it must be able to operate in these areas without adversely impacting ecosystem services. Consequently, there is a clear need i) to better understand the level of exposure to organic enrichment and the potential thresholds for change, ii) to develop an assessment technique that has the power to detect adverse impacts of organic enrichment on temperate reefs in advance, and iii) to identify how this understanding can be incorporated into routine monitoring programs to support management.

Existing standardised approaches for quantifying temperate reef environments focus on biodiversity (e.g. Edgar et al. 1999; Edgar and Stuart-Smith 2009), as this data is invaluable in the assessment of community response to global stressors and conservation efforts such as the establishment of marine protected areas (Edgar et al. 2009; Edgar et al. 2017). However, when assessing resilience or vulnerability of a reef ecosystem to organic enrichment, there is a need for a more targeted approach that can support management and the capacity to respond to a decline in ecosystem function (Valentine et al. 2016). Management responses often occur after widespread loss of kelp or the phase-shift in ecosystem functionality has occurred (e.g. Connell et al. 2008; Benedetti-Cecchi et al. 2001). The challenge is to develop a technique sensitive enough to detect early declines and functionality to enable rapid management response, yet broad enough that can be applied unaltered across geographic regions. The method must be sensitive enough to detect change, but rapid and simple enough for widespread uptake.

To address this need, a novel method to assess the functional response of temperate reef ecosystems to organic enrichment was developed and tested. This technique was based on, and validated against, the principals established in the more detailed biodiversity and conservation assessments. The suitability of a suite of relatively easy to observe functional parameters and organic enrichment indicators for monitoring was evaluated, along with the robustness of the overall technique. The overall aim was to develop a method that could be used to scale the increasing effects of organic enrichment both spatially and temporally, taking into account natural variability, thus providing a measure of vulnerability of temperate reef ecosystems to impact from organic enrichment.

6.5.2 Methods

6.5.2.1 Site selection

Salmon farming started in south-east of Tasmania in the Huon estuary in 1985 before expanding into the D’Entrecasteaux Channel. In 2016 a new farming site was established which is now the most southern of the leases in the region (Figure 6.5-1). While aquaculture of Atlantic salmon is a key industry in this region, the south-east D’Entrecasteaux Channel (SE Channel) is a multi-use area, subject to commercial and recreational fishing, forestry and other industry inputs from the catchment, as well as localised urbanisation from several small townships. The Huon River is the largest river in the region, with smaller rivers such as the Esperance and the Lune also providing freshwater inputs.

Eleven sites were selected within the SE Channel region; these sites represented a gradient of broadscale organic enrichment (Figure 6.5-1). Four sites were determined to be relatively pristine based on proximity to sources of anthropogenic organic enrichment (Actaeons, Southport, Sisters Bay and Lady Bay). Four sites were chosen in closer proximity (1.5 - 5 km) to salmon farming that were considered to have a low level of exposure to enrichment (Tower Bay, Lippies Point, Scott Point and Esperance Point), and the three remaining sites (Lomas Point, Zuidpool and Penguin Point) were considered to have a higher likelihood of being influenced by either natural and/or anthropogenic organic enrichment, although of note, the source of enrichment was not examined. Lomas Point is less than 500 m from an active aquaculture lease as well as being influenced by the Esperance River, and both Zuidpool and Penguin Point are in areas directly influenced by the Huon River as well as being subject to the dispersive influence of organic enrichment from multiple farms. On this basis the eleven sites were classified into one of three organic enrichment categories: control (Actaeons, Southport, Sisters and Lady), diffuse (Tower, Lippies, Scott and Esperance) and direct (Lomas, Zuidpool and Penguin; Figure 6.5-1). As the aim of this study was to develop a method to detect a shift in ecosystem function over a broad spatial scale, reefs subject to acute organic enrichment and gradient effects from point-source enrichment were not examined. This study focused on testing how the proposed technique detects functional change due to organic enrichment as opposed to change caused by light, salinity or temperature, and thus the design reflects a broadscale enrichment gradient. Through this study we did not attempt to test attribution of any observed change to a specific source of organic enrichment.

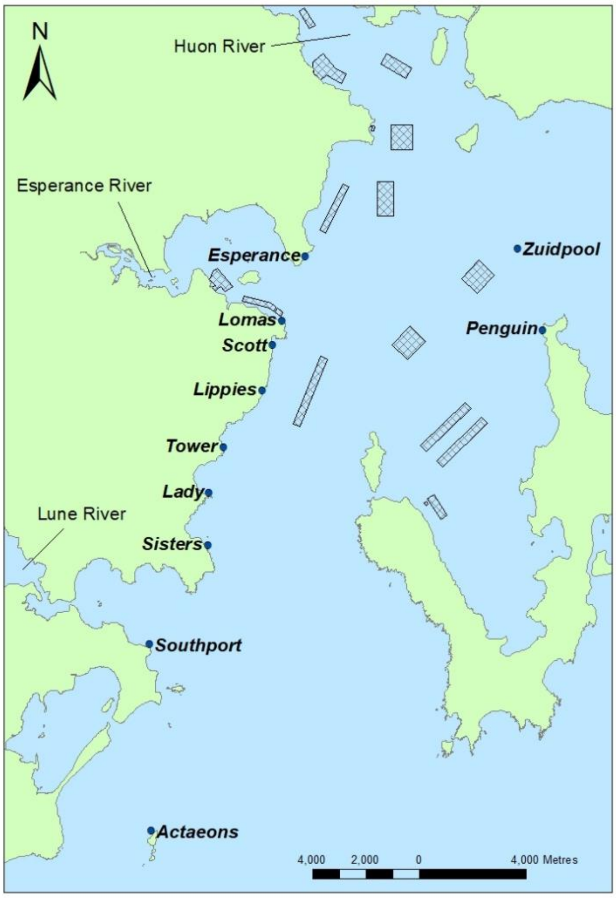

Figure 6.5-1 Southern D'Entrecasteaux Channel region with rapid visual assessment sites, along with finfish aquaculture leases and major freshwater inputs. Sites were categorised according to enrichment exposure: control (Actaeons, Southport, Sisters and Lady), diffuse (Tower, Lippies, Scott and Esperance) and direct (Lomas, Zuidpool and Penguin).

6.5.2.2 Method development & sampling design

Fifteen functional parameters were selected for evaluation (Table 6.5-1) based on indicators of organic enrichment and biodiversity previously identified from the SE Channel. Of these fifteen parameters, ten related to broad structural conditions associated with ecosystem function (i.e. four provided an evaluation of the macroalgal canopy conditions, four assessed substrate condition and two reflected trophic interactions), and the five remaining parameters were related specifically to enrichment response. The fifteen parameters were incorporated into a scorecard (Table 6.5-1, Supplementary Content S1).

Broad functional parameters included percentage total canopy cover, sub-canopy red, brown and green algal cover, turfing algal cover, pink and red encrusting algal cover, sponge cover, levels of encrusting fauna, and numbers of the dominant major mobile invertebrates (Supplementary content S1). Canopy cover was broken down into species, with the dominant species of algae and invertebrates recorded where possible for future reference. Enrichment parameters included percentage cover of epiphytic and filamentous algae, cover of opportunistic green (characterised by *Ulva*, *Cladophera* and *Chaetomorpha* in our sampling region) and opportunistic red species (characterised by *Asparagopsis armata* in our sampling region), along with the level of "dust" (sedimentation from the water column) covering the algae.

Table 6.5-1 Functional parameters for rapid visual assessment of temperate reef ecosystems in south-east Tasmania.

Functional parameter	***Expected response to increased organic enrichment***	***Reference***
Total canopy cover (including breakdown of species)	Decline	Connell Connell et al. 2008; Eriksson et al. 2002; Benedetti-Cecchi et al. 2001
Sub-canopy brown cover	Likely to decline as per canopy	
Sub-canopy green cover	Potential increase due to increased nutrient availability	Oh et al. 2015; Nelson et al. 2008
Sub-canopy red cover	Potential increase due to higher sedimentation in water column. Overall increase in red+green:brown algae ratio expected in enhanced nutrient conditions	Stuart-Smith et al. 2008
Turfing algal cover	Increase	Connell et al. 2008; Eriksson et al. 2002; Benedetti-Cecchi et al. 2001
Pink encrusting algal cover	Potential decline and replacement by turfing or opportunistic algae if canopy is lost	Burkepile and Hay, 2006
Red encrusting algal cover	Could decline as per pink encrusting, or increase due to changes in predation pressure or light conditions	Burkepile and Hay, 2006
Sponge cover (including breakdown of encrusting vs branching)	Likely to increase under mild organic enrichment.	
Encrusting & epibiotic fauna	Potential increases with increases in opportunistic algal cover likely	Russell and Connell, 2005; Burkepile and Hay, 2006
Species and number of dominant mobile invertebrate	Unknown	

Epiphytic algal cover	Increase	Oh et al. 2015
Filamentous algal cover	Increase	Oh et al. 2015; Lavery and Mccomb, 1991
Opportunistic green algal cover	Increase	Oh et al. 2015; Nelson et al. 2008
Opportunistic red algal cover	Increase	Anecdotal
“Dust” on algae	Increase (a reflection of sedimentation)	Anecdotal

A fixed design was used, with the aim of limiting the natural variability between sampling events. At each site twelve fixed quadrats were established at 5 m depth using eye-bolts drilled into the substrate. Rapid visual assessment (RVA) was undertaken by two divers on SCUBA. Diver 1 assessed and scored parameters “in situ” at all 12 quadrats using a 1 m^2 quadrat that had been sub-divided into four smaller 0.5 m^2 subsections to increase scoring accuracy. Diver 2 photographed each quadrat for archive. All parameters were assessed in the full 1 m^2 quadrat, except for substrate parameters, which were sub-sampled using a 0.5 m^2 subsection of the quadrat. Quadrats 1-4 were assessed by both divers at each site to test inter-diver variability. Quadrats 5-12 were assessed by only one diver only. All eleven sites in the SE Channel were surveyed in September 2018 and repeated in February 2019.

6.5.2.3 Data analysis

Patterns in functional parameters were investigated using the multivariate software package PRIMER v7 (Plymouth Routines in Multivariate Research; Clarke and Warwick, 2001) and its complementary software package PERMANOVA+(v7) (Anderson et al. 2008). A Bray-Curtis dissimilarity index was used to examine differences between samples and principal coordinates analysis (PCO) and to visualise patterns in data. Vector overlays using a Pearson correlation along with SIMPER analysis were used to identify key parameters driving trends in data. A 3-way PERMANOVA was undertaken to test the effect of organic enrichment, site (nested within organic enrichment) and timing of sampling on the data. The data were also analysed to explore spatial and temporal patterns, with site means calculated and differences across site and sampling event examined through vector analysis.

Power analysis was used to test the robustness of the key rapid assessment parameters using a paired t-test in Piface Version 1.76. The level of change that could be reliably detected in each parameter using 12 quadrats was determined at a power level of 0.8. Spatial and temporal differences in these parameters were examined in RStudio Version 1.2.1335 using 2-way ANOVA with site and season as fixed factors within the model. Data were transformed where necessary to meet the assumptions of the model. Lastly, the robustness of the method to transfer between divers was investigated by examining differences between divers in both multivariate analysis and through 2-way ANOVA with diver as a nested factor within site on the key assessment parameters identified above.

6.5.3 Results

6.5.3.1 Responses to organic enrichment on temperate reefs using rapid visual assessment

Enrichment at the category level (control, diffuse and direct) had a significant effect on the functional parameters of the reef ($F_{2,263}$ = 1.94, P(perm) = 0.03), with sites directly exposed to organic enrichment differing from control and diffuse sites. The timing of sampling also influenced parameters ($F_{1,263}$ = 6.31, P(perm) = 0.004), while site location was highly significant ($F_{8,263}$ = 13.3, P(perm) = 0.0001), indicating ecosystem function may vary on a site-by-site basis. There was also a strong interaction between site location and timing of sampling ($F_{8,263}$ = 4.52, P(perm) = 0.0001), indicating time of sampling may also influence reef function.

Further investigation of the interaction between site and time suggested functional differences between sites were primarily a consequence of variation in structural and enrichment associated parameters. Differences in structural parameters such as the percentage cover of canopy, encrusting red and pink algae, as well as understorey brown, red and green algae could be used to define functional differences between sites, regardless of time of sampling (Figure 6.5-2), with patterns observed in the PCO supported by SIMPER analysis (Table S2). However, the influence of enrichment parameters did show a large degree of temporal difference, with vectors for the cover of epiphytic and filamentous algae, and *Asparagopsis* corresponding to sites exposed to higher levels of organic enrichment (Zuidpool, Penguin Point and Lomas Point) in the February sampling only (Figure 6.5-2, Table S2).

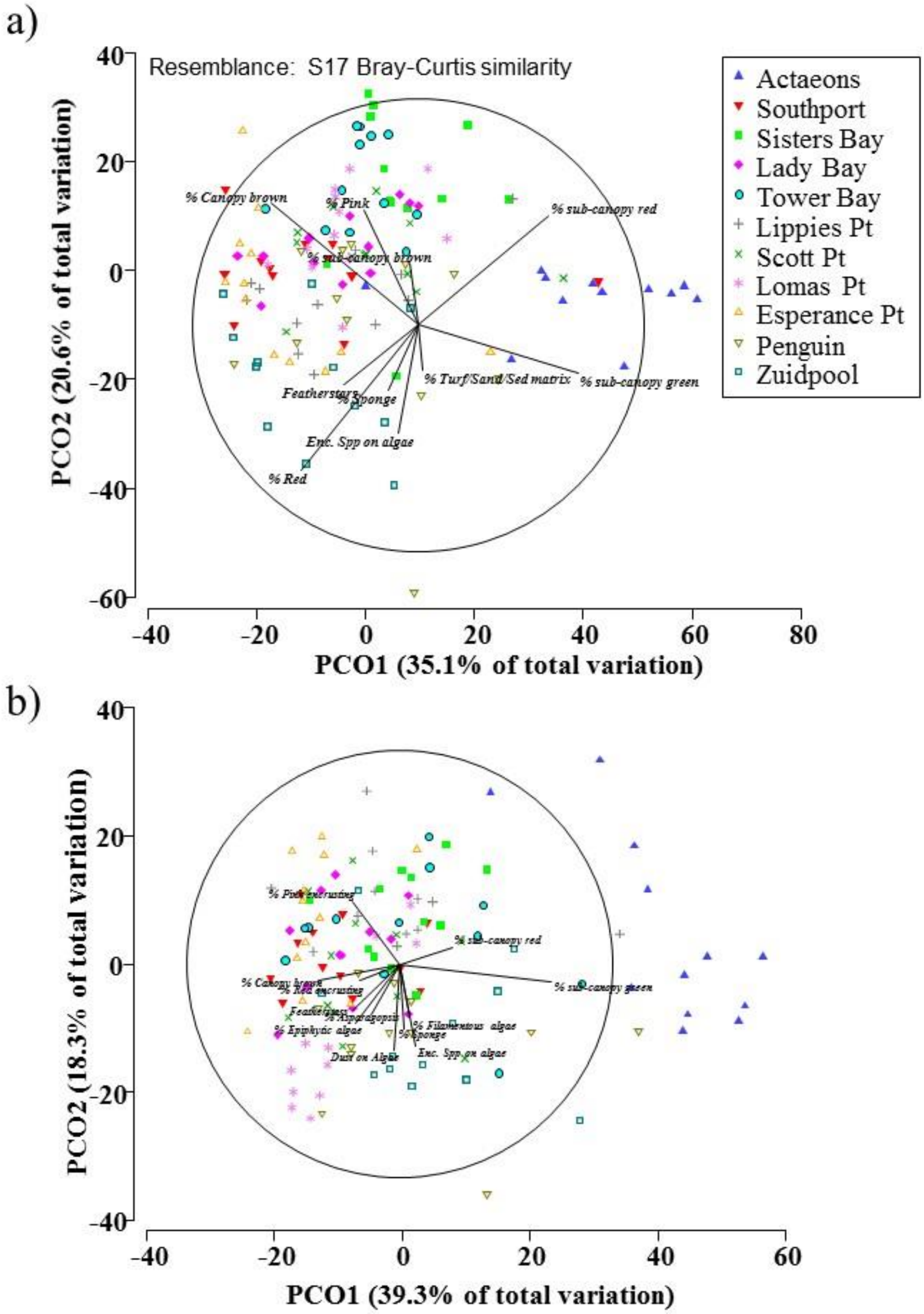

Figure 6.5-2 Principal coordinates analysis (PCO) on rapid visual assessment (RVA) parameter values across site during a) September 2018 and b) February 2019 sampling events. Vector overlays contributing to principal coordinates with a base variable comparison of > 0.25 are included.

6.5.3.2 Patterns in reef function using rapid visual assessment

A site-averaged comparison of site across time demonstrated a clear difference in functional parameters between February and September (Figure 6.5-3). Sites in February were generally characterised by much higher values for canopy and sub-canopy brown algal cover (Figure 6.5-3), with mean percentage canopy values for February higher at all sites except for Tower Bay (Figure 6.5-4, Table 6.5-2 and Table 6.5-3). The spread of data was much higher in September than February, indicating there was more variation in functional parameters between sites in September (Figure 6.5-3). One of the control sites, the Actaeons, was a clear outlier in this analysis, with functional differences between sampling times largely due to changes in sub-canopy red and sub-canopy green algae. This was reflected in the mean values, with changes in the cover of sub-canopy red (September: 35.4 ± 3.5%, February: 62.1 ± 4.5%) and sub-canopy green algae (September: 42.5 ± 5.2%, February: 7.7 ± 1.9%) evident (Figure 6.5-4, Table 6.5-2 and Table 6.5-3).

Vector analysis indicated only a low correlation between canopy cover and the Actaeons (Figure 6.5-3), with canopy values of only 2.1 ± 1.3% and 6.7 ± 2.5% recorded in September and February respectively (Figure 6.5-4, Table 6.5-3). However, this is likely an artefact of the method, as the canopy cover at the Actaeons was dominated by the giant kelp, *Macrocystis pyrifera*. As *M. pyrifera* forms canopies on the surface of the water, it is impossible to accurately assess cover using benthic quadrats and the resultant analysis may be biased accordingly. If the Actaeons is removed from the dataset, a site average comparison demonstrated an even clearer separation between the September and February sampling (Figure 6.5-5). Sites aligned much more closely with the vectors for canopy and sub-canopy brown algal cover during February. However, vector analysis indicated high covariation of enrichment parameters, including cover of *Asparagopsis*, epiphytic algae and filamentous algae (Figure 6.5-5). Sites with high values for the organic enrichment indicator parameters tended to have low values for pink encrusting coralline cover and sub-canopy red (Figure 6.5-5, Table 6.5-3). Sites in the direct enrichment category, including Zuidpool and Penguin Point, tended to align closely with these vectors in February, although not in September, and recorded comparatively high mean values for these parameters (Figure 6.5-5, Table 6.5-3). For example, in February, Penguin Point had epiphytic algal cover and filamentous algal cover of 22.5 ± 7.2% and 12.1 ± 5.5% respectively (Figure 6.5-5, Table 6.5-3). Comparatively high values for sub-canopy green (10.0% ±1.4) were also recorded at Penguin Point in February (Figure 6.5-5, Table 6.5-3).

Also in the direct enrichment category, Lomas Point appeared to be the exception. This site showed *Asparagopsis* and epiphytic algal cover were equally important in both September and February (Figure 6.5-4), and these metrics were high in both cases compared with other sites (Figure 6.5-3, Table 6.5-3). While the enrichment parameters were present consistently at Lomas Point in both September and February, other parameters relating to structural function (i.e., % canopy cover, % sub canopy brown, green, red algae, encrusting pink and red) did not differ significantly between Lomas Point and other sites (Figure 6.5-3, S1). Lomas Point had significantly higher epiphyte cover than other sites in September, and elevated *Asparagopsis* across both sampling events (Table 6.5-3, S1). However, in both cases the absolute mean values were reasonably low, with epiphytic algal cover being 27.5 ± 3.6% in September and 8.8 ± 1.1% in February, and *Asparagopsis* 8.8 ± 2.6% in September and 4.3 ± 1.4% in February (Figure 6.5-3, Table 6.5-3). Interestingly, Lomas Point also recorded relatively high mean values for percentage canopy cover overall (Figure 6.5-3, Table 6.5-3).

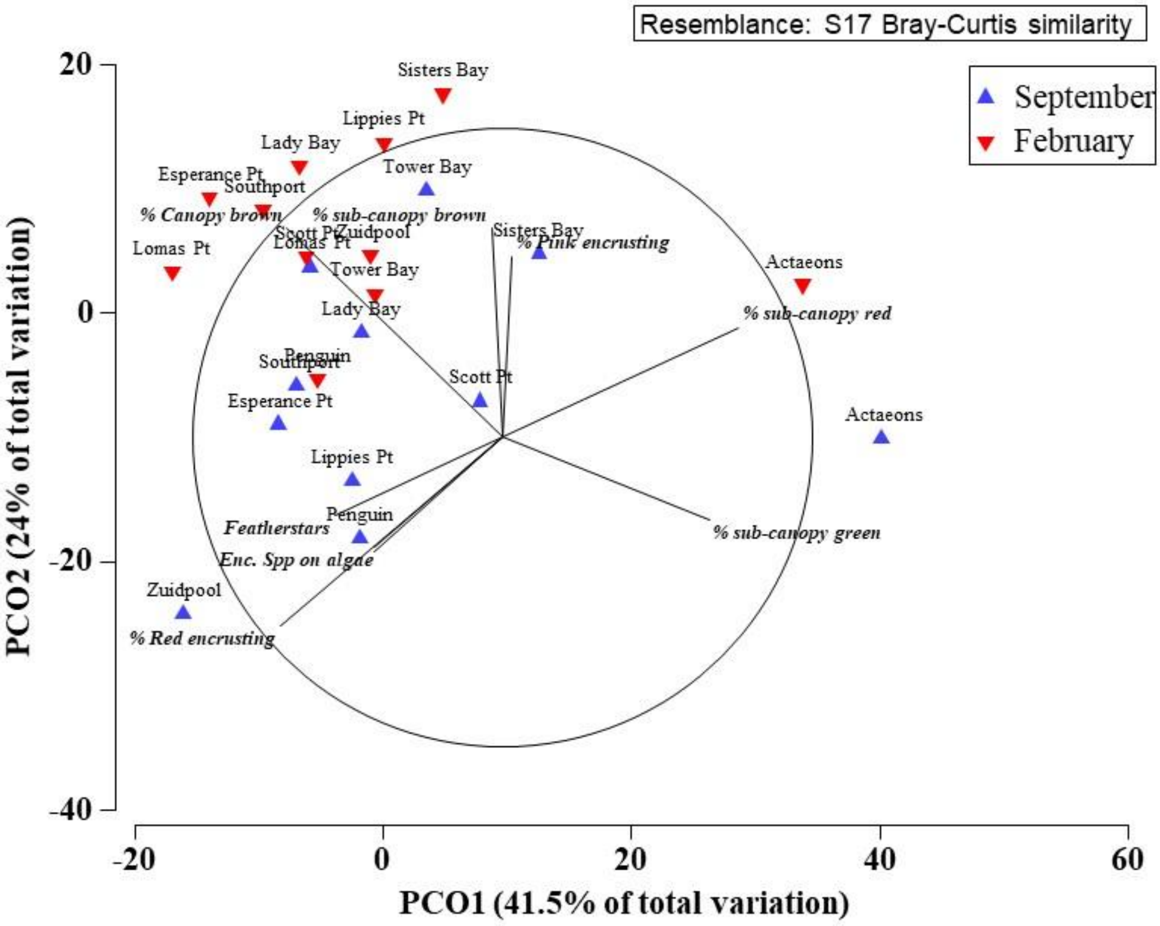

Figure 6.5-3 Principal coordinates analysis (PCO) on RVA parameter values based on means for each site shown for both September 2018 (blue) and February 2019 (red). Vector overlays contributing to principal coordinates with a base variable comparison of > 0.5 are included.

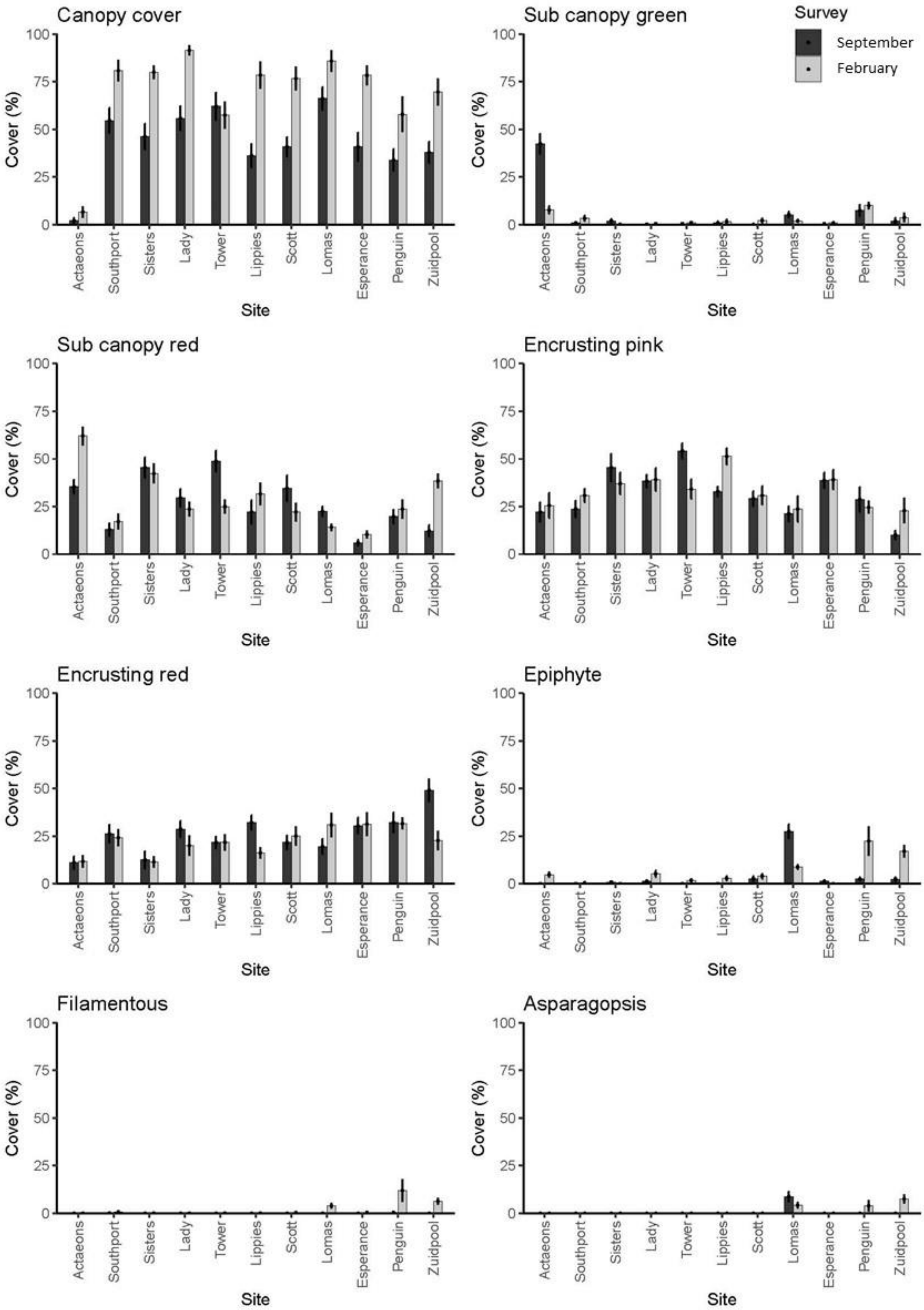

Figure 6.5-4 Percentage cover of the eight functional parameters identified as key through multivariate analysis across both September and February sampling events.

Table 6.5-2 The effect of sampling event and site on the eight key functional parameters through 2-way ANOVA.

Parameter	Site *F*	Sampling event *F*	Sampling * Site *F*
Canopy brown	22.17***	111.5***	3.17***
Sub canopy green	39.76***	14.03***	21.94***
Sub canopy red	17.06***	1.22	6.80***
Encrusting pink	7.51***	0.42	2.22*
Encrusting red	6.31***	3.80 ·	2.90**
Epiphytic algae	18.24***	9.63**	12.11***
Filamentous algae	5.30***	16.66***	4.77***
Asparagopsis armata	10.84***	2.53	5.09***

NB: Significance codes as follows 0 '***' 0.001 '**' 0.01 '*' 0.05 '·'

Table 6.5-3 Mean values (± SE) for rapid visual parameters surveyed in both September and February across eleven sites.

	Actaeons	**Southport**	**Sisters**	**Lady**	**Tower**	**Lippies**	**Scott**	**Lomas**	**Esperance**	**Penguin**	**Zuidpool**
September											
% Canopy brown	2.1 ± 1.3	54.6 ± 6.6	46.3 ± 6.5	55.8 ± 6.3	62.1 ± 7.2	36.3 ± 6.1	40.8 ± 5	66.3 ± 5.8	40.8 ± 7.5	33.9 ± 5.7	37.9 ± 5.4
% sub-canopy brown	11.3 ± 2.9	10.8 ± 2.4	13.3 ± 4.4	9.7 ± 1.5	19.2 ± 4.0	8.3 ± 1.3	6.5 ± 1.4	21.7 ± 5.1	11.7 ± 3.1	5.8 ± 0.8	10.8 ± 2.7
% sub-canopy green	42.5 ± 5.2	0.8 ± 0.4	1.9 ± 0.9	0.2 ± 0.1	0.4 ± 0.4	0.8 ± 0.8	0.0 ± 0.0	5.3 ± 1.4	0.4 ± 0.4	7.4 ± 2.9	1.8 ± 1.7
% sub-canopy red	35.4 ± 3.5	12.9 ± 3.2	45.4 ± 5.3	29.6 ± 4.6	48.8 ± 5.4	22.1 ± 6.1	34.6 ± 6.6	22.5 ± 2.5	6.0 ± 1.4	19.8 ± 3.6	12.3 ± 2.9
% Sponge	3.8 ± 0.7	7.3 ± 1.4	2.7 ± 0.6	5.5 ± 1.5	2.7 ± 1.7	5.3 ± 1.1	4.6 ± 1.2	5.3 ± 1.5	2.7 ± 0.9	9.1 ± 2.1	17.7 ± 4.5
% Pink encrusting	22.1 ± 5.1	23.8 ± 4.3	45.4 ± 7.1	38.3 ± 3.3	54.2 ± 3.9	32.9 ± 2.5	29.2 ± 3.7	21.3 ± 3.9	38.8 ± 3.9	28.8 ± 6.3	10.2 ± 2.2
% Red encrusting	11.1 ± 3.2	26.3 ± 4.7	12.5 ± 4.5	28.8 ± 4.1	21.7 ± 3.0	32.1 ± 3.8	21.7 ± 3.6	19.6 ± 4.0	30.4 ± 4.1	32.1 ± 5.2	49.2 ± 5.8
% Turf/Sand/ Sed matrix	4.5 ± 0.8	4.6 ± 0.7	6.2 ± 1.8	5.2 ± 1.3	7.3 ± 3.3	7.7 ± 3.0	5.5 ± 1.1	20.4 ± 6.2	3.5 ± 1.4	15.0 ± 5.0	20 ± 5.7
% Epiphytic algae	0.0 ± 0.0	0.0 ± 0.0	0.8 ± 0.6	1.3 ± 0.6	0.0 ± 0.0	0.0 ± 0.0	2.7 ± 1.2	27.5 ± 3.6	1.3 ± 0.6	2.5 ± 0.8	2.4 ± 0.9
% Filamentous algae	0.0 ± 0.0	0.1 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.4 ± 0.4	0.0 ± 0.0
% Ulva/ Chaetomorpha	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.2 ± 0.2	0.0 ± 0.0	1.3 ± 0.9	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
% Asparagopsis	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0 0.0 ± 0.00	8.8 ± 2.6	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0
Featherstars	0.0 ± 0.0	0.2 ± 0.0	0.3 ± 0.1	0.8 ± 0.2	0.4 ± 0.1	0.3 ± 0.1	1.4 ± 0.4	0.7 ± 0.1	1.5 ± 0.2	1.0 ± 0.1	1.8 ± 0.3
Dust on Algae	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	1.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	1.3 ± 0.1	0.0 ± 0.0	1.5 ± 0.2	1.0 ± 0.1
Enc. Spp on algae	0.9 ± 0.2	1.1 ± 0.4	0.0 ± 0.0	0.8 ± 0.1	0.0 ± 0.0	0.3 ± 0.2	0.4 ± 0.1	0.8 ± 0.1	0.8 ± 0.2	1.9 ± 0.1	2.3 ± 0.1
February											
% Canopy brown	6.7 ± 2.5	80.8 ± 5.1	80 ± 3.1	91.5 ± 2.2	57.5 ± 6.7	78.5 ± 6.8	76.7 ± 5.8	85.8 ± 5.2	78.3 ± 4.7	57.9 ± 8.8	69.6 ± 6.7
% sub-canopy brown	18.8 ± 2.8	17.9 ± 4.4	14.2 ± 4.3	17.1 ± 3.6	13.8 ± 3.0	15.4 ± 1.4	11.7 ± 0.9	16.7 ± 1.5	20 ± 3.4	5.9 ± 1.5	12.5 ± 2.0
% sub-canopy green	7.7 ± 1.9	3.3 ± 1.4	0.1 ± 0.1	0.5 ± 0.4	1.1 ± 0.4	1.5 ± 1.2	2.1 ± 1.3	1.9 ± 0.6	0.8 ± 0.6	10.0 ± 1.4	3.8 ± 2.1

% sub-canopy red	62.1 ± 4.5	17.1 ± 3.7	42.3 ± 4.8	23.8 ± 3.4	25 ± 3.4	31.7 ± 5.4	22.1 ± 4.5	14.2 ± 1.5	10.4 ± 1.6	23.8 ± 4.6	38.3 ± 3.4
% Sponge	6.1 ± 1.5	18.8 ± 5.7	8 ± 4.8	10.4 ± 2.9	3.7 ± 1.1	6.4 ± 1.2	6.7 ± 1.9	11 ± 2.1	3.6 ± 1	6.8 ± 1.5	14.1 ± 3.4
% Pink encrusting	25.6 ± 6.5	30.8 ± 3.4	37.1 ± 5.5	39.2 ± 5.8	34.2 ± 5	51.3 ± 4.2	30.8 ± 4.7	23.8 ± 6.6	39.2 ± 5	24.6 ± 3	22.9 ± 6.2
% Red encrusting	12.7 ± 2.7	24.2 ± 4.1	12.5 ± 2.5	21.8 ± 4.7	21.7 ± 3.9	16.3 ± 2.5	25 ± 4.6	30.8 ± 6	31.3 ± 5.7	31.7 ± 2.7	22.7 ± 4.6
% Turf/Sand/ Sed matrix	1.0 ± 0.6	1.4 ± 0.6	0.4 ± 0.4	5.2 ± 1.3	7.3 ± 3.3	7.7 ± 3.0	6.9 ± 3.3	24.6 ± 6.6	2.1 ± 0.6	7.9 ± 1.4	11.4 ± 3.0
% Epiphytic algae	4.8 ± 1.3	0.4 ± 0.4	0.0 ± 0.0	5.3 ± 1.6	1.8 ± 0.7	2.8 ± 1.1	4.0 ± 1.1	8.8 ± 1.1	0.0 ± 0.0	22.5 ± 7.2	17.1 ± 2.9
% Filamentous algae	0.0 ± 0.0	0.8 ± 0.3	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.3 ± 0.2	4.0 ± 1.1	0.4 ± 0.2	12.1 ± 5.5	6.4 ± 1.3
% Ulva/ Chaetomorpha	0.5 ± 0.4	0.0 ± 0.0	0.1 ± 0.1	0.1 ± 0.1	1.1 ± 0.4	1.1 ± 0.8	2.0 ± 1.3	0.6 ± 0.2	0.8 ± 0.4	1.8 ± 0.9	0.0 ± 0.0
% Asparagopsis	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	4.3 ± 1.4	0.0 ± 0.0	3.9 ± 2.5	7.5 ± 2.0
Featherstars	0.0 ± 0.0	0.0 ± 0.0	0.2 ± 0.1	0.6 ± 0.1	0.4 ± 0.1	0.8 ± 0.2	0.8 ± 0.2	1.8 ± 0.2	1.5 ± 0.2	0.6 ± 0.1	1.6 ± 0.4
Dust on Algae	0.0 ± 0.0	0.2 ± 0.2	0.0 ± 0.0	0.0 ± 0.0	0.2 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	1.2 ± 0.2	0.0 ± 0.0	1.7 ± 0.2	2.1 ± 0.2
Enc. Spp on algae	0.6 ± 0.1	0.8 ± 0.2	0.6 ± 0.1	1.4 ± 0.1	0.9 ± 0.1	0.9 ± 0.1	0.9 ± 0.1	1.7 ± 0.1	1.2 ± 0.1	1.2 ± 0.1	2.1 ± 0.1

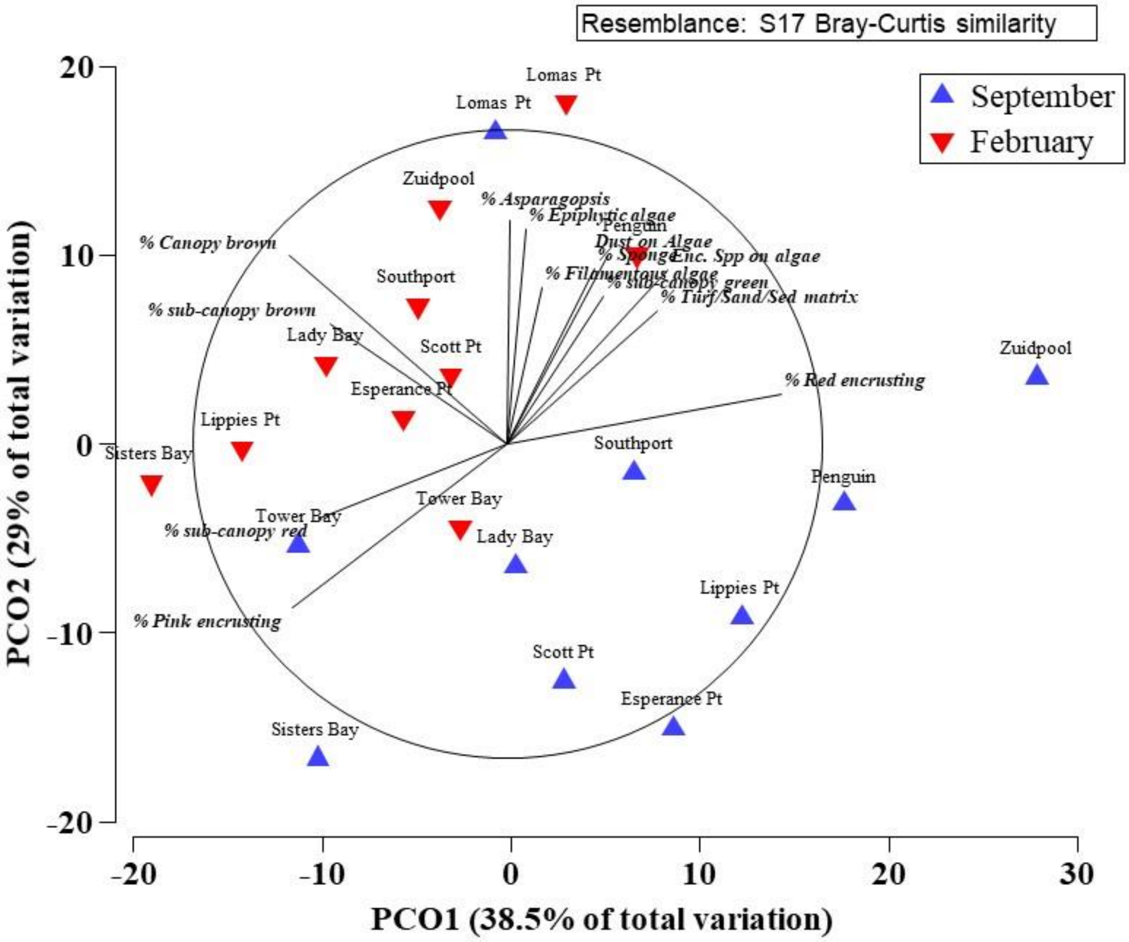

Figure 6.5-5 Principal coordinates analysis (PCO) on RVA parameter values based on means for each site (excluding the Actaeons), shown for both September 2018 (blue) and February 2019 (red).

Vector overlays contributing to principal coordinates with a base variable comparison of > 0.5 are included.

6.5.3.3 Robustness of key assessment parameters using rapid visual assessment

The multivariate analysis identified eight parameters that appear to be key to characterising response across each of the factors examined through our analysis: site, time and organic enrichment level. These included functional parameters of canopy cover, sub-canopy green, sub-canopy red, encrusting pink algae, encrusting red algae, and enrichment parameters epiphytic algal cover, filamentous algal cover and *Asparagopsis* cover. When examined individually with a two-way ANOVA with site and time of sampling as factors, site was a significant factor for each of the parameters above, as was the interaction between site and time of sampling (Figure 6.5-3, S1). However, time of sampling by itself was a significant factor for canopy cover, sub-canopy green, epiphyte and filamentous algae (Figure 6.5-3, S1).

Power analysis indicated that rapid visual assessment using 12 quadrats could detect a 35-56% change to mean canopy cover in September and a 7-45% change to mean canopy cover in February, across all sites except the Actaeons (Table 6.5-4). In general, the ability to detect change was higher for canopy cover in February, where percentage cover was itself generally higher, and therefore cover was less spatially heterogenous. Once cover dropped below 10-15%, the reliability of the RVA method to accurately detect change was reduced regardless of parameter. For example, where canopy cover at the Actaeons was 2.1 ± 1.3% in September and 6.7 ± 2.5% in February, the level of change that could be accurately detected was 192% and 115% respectively (Table 6.5-3, Table 6.5-4). Where organic enrichment parameters (epiphyte cover, filamentous cover & *Asparagopsis* cover) occurred at values lower than 10% cover as a baseline, the percentage cover of these parameters needed to more than double before significant change could be detected (Table 6.5-4).

Table 6.5-4 Percentage change detectable using the rapid visual assessment (RVA) method for eight key parameters (based on results of a paired t-test; power = 0.08).

	Actaeons	**Southport**	**Sisters**	**Lady**	**Tower**	**Lippies**	**Scott**	**Lomas**	**Esperance**	**Penguin**	**Zuidpool**
September											
% Canopy brown	192	37	44	35	36	52	38	27	56	51	44
% sub-canopy green	38	163	148	210	307	308	-	84	307	121	292
% sub-canopy red	30	77	36	48	34	85	58	34	74	56	73
% Pink encrusting	71	55	48	27	22	23	39	57	31	68	68
% Red encrusting	88	56	112	44	43	36	51	62	42	50	36
% Epiphytic algae	-	-	208	128	-	-	139	40	128	93	118
% Filamentous algae	-	312	-	-	-	-	-	-	-	307	-

% Asparagopsis	-	-	-	-	-	-	-	92	-	-	-
February											
% Canopy brown	115	20	12	7	36	27	23	19	18	47	30
% sub-canopy green	78	133	312	256	118	253	185	94	208	42	176
% sub-canopy red	22	66	35	45	42	53	63	32	21	59	27
% Pink encrusting	78	34	46	46	45	25	47	86	40	37	84
% Red encrusting	73	52	69	74	56	47	57	60	56	26	63
% Epiphytic algae	81	307	-	92	123	120	89	38	-	99	51
% Filamentous algae	-	115	-	-	-	-	174	89	144	139	64
% Asparagopsis	-	-	-	-	-	-	-	104	-	193	82

NB: Dashed lines indicate this parameter was absent during that sampling event.

6.5.3.4 Variability between divers

There was no significant difference between divers for the quadrats scored side-by-side at each site ($F_{11,175}$ = 0.21, P(perm) = 1.0), indicating the outcome of the RVA would be the same regardless of which diver is collecting the data. This was also true for each of the key assessment parameters individually (Figure 6.5-6, S4). When averaged across all sites, the discrepancy between divers was relatively small and ranged from 0.25% for *Asparagopsis* assessments to 4.07% in sub-canopy red algae (Table 6.5-5). Standard deviation of the mean discrepancy was also relatively small, and less than 5% for all parameters except encrusting red algae (5.44%). This suggests that for most parameters, the inherent level of variation associated with any measurement using the RVA scoring method is less than 10% when using well-trained divers.

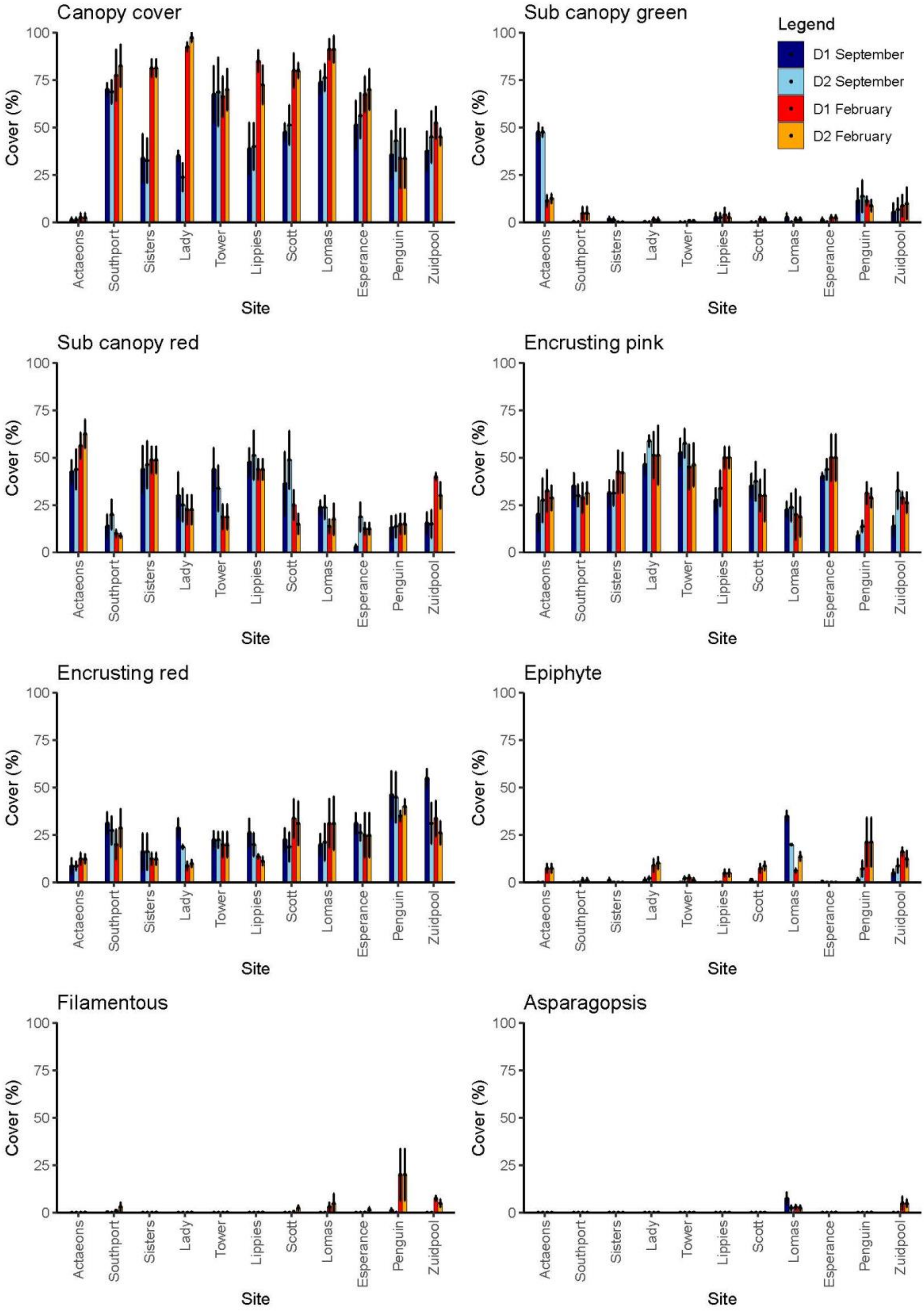

Figure 6.5-6 Mean% cover value recorded for Diver 1 and Diver 2 for each of the key assessment parameters across all sites surveyed.

Table 6.5-5 Discrepancy between Diver 1 and Diver 2 for each of the key assessment parameters across all sites surveyed.

Parameter	Mean discrepancy between divers	Standard deviation
Canopy	3.58	3.73
Sub-canopy green	0.73	0.90
Sub-canopy red	4.07	4.79
Encrusting pink	3.72	4.53
Encrusting red	3.75	5.44
Epiphytic algae	2.10	3.54
Filamentous algae	0.53	0.89
Asparagopsis armata	0.25	1.07

6.5.4 Discussion

This study shows that multiple functional parameters can be used to detect differences in broadscale enrichment on temperate reefs. Sites categorised as being subject to direct organic enrichment (natural and/or anthropogenic) tended to have higher values for functional parameters commonly associated with nutrient enrichment, such as epiphytic and filamentous algal growth. These parameters are known to respond to increased ambient nutrient concentrations associated with both organic pollution and seasonal nutrient pulses (Burkepile & Hay 2006; Oh et al. 2015). However, it was not possible to detect the source of organic enrichment; with the broadscale sampling design, the functional parameter response was the same for sites exposed to natural and finfish nutrient sources, and we could not detect any change where sites were more than 1.5 km from fish farms. The findings suggest that functional assessment is sensitive enough to characterise the condition at individual sites, and to capture change over time, both at the individual site level and across regions.

This method focused on the functional parameters associated with algal communities on temperate reefs as a key initial response to nutrient enrichment stimulus. However, the functional condition of algal communities may also vary between sites and over time in response to a number of environmental factors, including exposure, light availability and water temperature (Edgar 1984; Hill et al. 2010; Leaper et al. 2011). In addition, the dominant canopy forming species will affect trophic interactions, with the functional status of the temperate reef at any given site or time reflecting the influence of all the above environmental parameters (Edgar 1984; Bishop et al. 2010). This method is necessarily focused on the detection of organic enrichment; however, the technique can also be used to examine shifts in the broader functional ecology of the algal communities. While the temporal replication of the study was limited, having only sampled once in September and February, the results suggest there would be capacity to use the approach to assess temporal variation in functional ecology over a longer timeframe.

6.5.4.1 Functional changes in temperate reef ecosystems

While enrichment parameters such as epiphytic and filamentous algal cover clearly increased at sites with higher exposure to organic enrichment, most of the broad functional parameters such as canopy cover, sub-canopy brown and sub-canopy red algal cover were quite stable. In a temperate reef ecosystem, canopy cover is by far the most important parameter in terms of maintaining structure and function (Schiel 2011; Smale et al. 2013; Wernberg et al. 2013). The condition of the canopy will drive trophic interactions and underpin the overall productivity of the ecosystem (Miller et al. 2009). Temperate reefs with dense, healthy canopy-forming macroalgae are stable and resilient and unlikely to be detrimentally affected by pulses of epiphytic, filamentous or other rapidly growing nuisance algal species (Layton et al. 2019; Shelamoff et al. 2019). There was no relationship between canopy cover and the category of organic enrichment of a particular site, indicating that despite increases in other enrichment parameters in February, the overall ecosystem function was largely unaffected, and the canopy remained intact.

Coastal ecosystems are adapted to seasonal or pulse nutrient inputs; it is not uncommon for taxa with capacity for fast growth that can rapidly absorb nutrients from the system to be present, and at times dominant in coastal systems (Duarte 1995; Pedersen & Borum 1997). In this study, epiphytic, filamentous and opportunistic algal species were more abundant at all sites subject to increased organic enrichment especially during February (summer), where greater daylight hours and light intensity were likely to exacerbate the effects of high levels of nutrient. Zuidpool and Penguin were in the “direct” nutrient zone and subject to elevation of nutrients from the Huon Estuary. At both sites cover of epiphyte and filamentous algae was shown to be elevated in February compared to September. While proliferation might be expected in response to increased light over summer, a pulse of nutrients following a storm event or increased freshwater flows from an estuary may also trigger a short-term bloom of epiphytic or opportunistic algal species. Die-back often occurs after the pulse of nutrient has abated (Birch et al. 1981; Raffaelli et al. 1998; Smith et al. 2005), but such effects are often regional or broadscale.

In contrast, anthropogenic nutrient inputs are often point-source and tend to be sustained low level additions to the system (Gillanders & Kingsford 2002). This may result in a different response from nutrient sensitive taxa, with constant nutrient elevation leading to a prolonged growth period. In this study, Lomas Point sited < 500 m from a finfish farm, was the only site where enrichment indicators (epiphytic algae and the opportunistic red *Asparagopsis armata*) were present over both September and February. While canopy was unaffected, the response of these more specific indicators suggested Lomas Point might be exhibiting the effects of sustained organic enrichment from the farm (Oh et al. 2015), with the functional indicators reflecting the biological mitigation response. Given only two sampling events within a year, it is difficult to comment further on either the potential longevity or ongoing effects of this level of increase in opportunistic algae taxa at sites exposed to organic enrichment. However, sustained increases of these functional groups across multiple surveys would suggest they are likely to be good first indicators of chronic organic enrichment exposure. Given prolonged enrichment and proliferation of opportunistic species on temperate reefs can lead to a loss of functional richness (Airoldi et al. 2008; Ling et al. 2018), a better understanding of temporal responses is required to assess the significance of this observed response.

While the RVA method was able to detect a functional response to organic enrichment, the approach also managed to clearly identify a temporal change in reef function between September and February. It is not surprising that rapidly growing filamentous algae were more prolific in February, when temperature and light conditions favour fast growth (Krause-Jensen et al. 2007; Oh et al. 2015). More interesting was the temporal difference in canopy cover, with the total canopy cover being on average 30% greater in February compared to

September across the study area. Shifts in the canopy of this magnitude represent a significant increase in the overall productivity and reflect the system's inherent ability to adjust to changing nutrient and environmental conditions. This finding also highlights that the carrying capacity or resilience of the system is not static but changes over time and with prevailing environmental conditions, highlighting a further difficulty in identifying standardised monitoring approaches and thresholds.

Neither *Phyllospora comosa* nor *Ecklonia radiata*, the two dominant canopy-forming species in SE Tasmania, generally experience large-scale seasonal dieback or necrosis associated with change in water temperatures (Sanderson 1992; Bearham et al. 2013; Flukes et al. 2015; Coleman & Wernberg 2017). While both species have elevated growth rates as light and temperature increase, canopy dieback and recovery of the magnitude observed are more likely to be due to mechanistic rather than physiological processes (Wernberg & Goldberg 2008). Pruning via abrasion or wave disturbance can account for up to 50% biomass loss in *E. radiata*, with increased swells and lower light levels over winter exacerbating the noticeable effect of these processes (Kirkman, 1984; Wernberg & Goldberg, 2008). The strong swell associated with winter storm events in the SE Channel has the capacity to remove canopy entirely, both at specific sites and more broadly throughout the system, with both *P. comosa* and *E. radiata* susceptible to damage from exposure (Wernberg & Goldberg 2008; Bearham et al. 2013; Coleman & Wernberg 2017). A more long-term dataset would make it easier to assess whether the loss and recovery of the canopy are annual or seasonal events.

Other parameters that varied between sampling events included sub-canopy green, sub-canopy red, and epiphytic and opportunistic algal cover. Variation in these parameters was also highly site specific, as indicated by the significant interaction term in the analysis. For example, at the Actaeons and Lomas sub-canopy green algal cover was greater in September than February, whereas it was the reverse at Penguin and Zuidpool with green algal cover elevated in February. This likely reflects the influence of differences in environmental conditions at the individual sites and the relative sensitivities of the different sub-canopy groups (i.e., brown, green, red) to changes in localised light and nutrient conditions (Miller et al. 2004; Toohey et al. 2004; Alestra & Schiel 2015). Again, understanding these processes and the natural variation within the system will be improved with longer term data for each site. More frequent sampling would clarify the short-term variability (noise) in the data which would make it easier to distinguish with certainty the difference between natural process variation (e.g., filamentous algae blooms during peak ambient growing conditions) and variations that are likely to be driven by anthropogenic inputs (e.g. sustained growth of epiphytic algae as a result of salmon farm derived nutrient impacts).

Many studies have demonstrated the capacity of the macroalgal canopy to shape biodiversity and influence ecosystem function on temperate reefs (e.g. Toohey et al. 2004; Wernberg and Goldberg 2008; Miller et al. 2011; Coleman and Wernberg 2017). There is generally a negative relationship between canopy cover and understorey abundance and diversity, with fluctuations in canopy cover driving change in subsequent algal layers (Wernberg et al. 2005; Wernberg & Goldberg 2008). In their study of short-term temporal dynamics in *E. radiata* dominated kelp beds, Wernberg and Goldberg (2008) found species losses and gains through time were associated with recovery and decline of algal canopy at highly localised spatial scales. Likewise, the vector analysis indicated a strong correlation between canopy cover and encrusting pink substrate, particularly at sites dominated by *P. comosa*. In this instance, mechanical abrasion by the canopy from swell, along with low space and light availability is likely a key limitation for sub-canopy growth and recruitment, and consequently encrusting pink algae is dominant (Kim et al. 1998; Wernberg & Goldberg 2008). In contrast, sub-canopy green and red algae have generally greater cover at the Actaeons, where the giant kelp canopy is metres above the substrate, providing space subsidies to the understory with no mechanical abrasion to remove sporophytes (Breda & Foster 1985; Wernberg & Goldberg 2008). Whilst a longer time-series would improve assessment of the robustness of this method for capturing fluctuations in ecosystem function, these examples highlight the potential for the RVA method to capture trophic linkages and flow-on functional effects due to canopy change.

6.5.4.2 Functional assessment and management responses

Another strength of the RVA method is the capacity to detect concurrent shifts across multiple parameters. This is particularly useful in assessing organic enrichment, where several functional parameters are expected to correlate strongly. For instance, in extreme cases of organic enrichment, large losses of canopy might be expected along with increases in epiphytic, filamentous and opportunistic algae, as well as turfing algae (Russell et al. 2005; Connell et al. 2008; Oh et al. 2015). Assessing each parameter singularly would make it harder to relate changes to organic enrichment. For example, loss of canopy could be due to environmental factors including storm events (e.g. Wernberg 2006), or overgrazing by pest species (e.g. Ling et al. 2015), whilst opportunistic species may respond to seasonal increases in light or nutrient availability (Smith et al. 2005; Krause-Jensen et al. 2007). By examining shifts in multiple parameters, multiple lines of evidence are used to link cause and effect to organic enrichment. Subsequent examination of individual parameters can provide an understanding of the magnitude of any effects. The multivariate approach provides greater capacity to both detect and relate a system level shift in response to organic enrichment.

While the RVA method indicated that increases in abundance of epiphytic and filamentous algae can be a useful indicator of organic enrichment, other prospective measures were shown to be less valuable through the analysis. Parameters such as increased turfing algae, increased red to pink encrusting algae ratios or increased green and red sub-canopy to canopy, previously found to be reliable indicators of enrichment (Connell et al. 2008; Stuart-Smith et al. 2008) were found not to be useful here. However, this may be because these parameters tend to be associated with loss of canopy forming macroalgae (Connell et al. 2008; Gorman et al. 2009), which did not occur at any of the sites. It is possible that the effects of organic enrichment were not extreme enough in our study for the value of these parameters to become evident. For example, sustained moderate cover of epiphytic and filamentous algae, but an intact macroalgal canopy, may indicate low to moderate organic enrichment, while losses of canopy along with increases in turfing algae and opportunistic species may be more indicative of extreme organic enrichment (Connell et al. 2008; Oh et al. 2015). While this might indicate the method needs to be further tested in an environment subject to more extreme organic enrichment, it also highlights the potential of this method to be used to scale the effects of organic enrichment. A long-term reef monitoring program that can produce a scale or index of organic enrichment for each monitoring site would be a valuable tool for management.

Management of other environments, such as soft sediment or pelagic systems, and the response to organic enrichment is dependent on identifying critical ecosystem changes or tipping points and their thresholds. The value at which a particular parameter exceeds a known compliance point (Macleod et al. 2004a; Macleod et al. 2004b; Ross & Macleod, 2013; Keeley et al. 2015) becomes a key management value.

The maximum cover of epiphytic or opportunistic species that a temperate reef can sustain without causing functional loss to an ecosystem is presently unknown. However, due to the inherent complexity in temperate reef ecosystems, it is likely to be dependent on a range of factors. In this study, loadings of epiphytic and filamentous species were negligible at most sites and only exceeded 25% once. With values for canopy cover generally greater than 50%, it is unlikely this level of epiphytic growth is causing stress to the canopy. Oh et al. (2015) noted that where reef systems were 100 m or closer to active salmon farms at sheltered sites, epiphytic and filamentous species cover could be greater than 50%. However, even at these higher opportunistic species loadings, Oh et al. (2015) found that canopy brown algae appeared relatively stable at approximately 50% cover. That said, if a large storm event removed the macroalgae from sites where there was simultaneously high epiphyte loads, it is unknown whether the canopy would re-establish, or if the higher nutrient loadings would have prevented this through proliferation of opportunistic species, including sediment trapping algal turfs (Eriksson et al. 2002; Connell 2005; Connell et al. 2008). While a temperate reef may be resilient to relatively high loadings of epiphytic and opportunistic species, the sustained presence of these species may indicate ongoing vulnerability should disturbance lead to the clearance of the canopy.

To adequately manage temperate reef ecosystems there needs to be a much clearer understanding of the level of change that will result in a significant alteration to ecosystem function. The development of tools such as trigger values that can be used in a management context require long term datasets to capture sufficient natural ecosystem variation. The study indicates natural fluctuations are likely to be quite high, with mean canopy values increasing by 30% across the study area between September and February. While the effect on temperate ecosystems of total canopy loss due to organic enrichment may be relatively well understood (e.g. Connell et al. 2008; Gorman and Connell 2009; Coleman and Wernberg 2017), the more refined sequence of events that would lead to this are less clear and there is still a need for management advice and strategies for these systems that could ideally mitigate this scenario.

The amount of canopy loss an ecosystem can sustain before there is a functionally significant transition is relatively unknown and likely to vary depending on several environmental variables, such as canopy type. The majority of studies have focused on the effects of total canopy loss (Edgar et al. 2004; Valentine & Johnson 2004), with less known regarding canopy thinning, which is potentially more likely to occur as a result of sustained high loadings of opportunistic species. Flukes et al. (2014) suggests a 66% reduction of canopy will cause a significant shift in understory in *E. radiata* dominated assemblages in pristine habitat, although it is unknown if this value would be applicable to the *P. comosa* and *M. pyrifera* dominated habitats that are prevalent in our study area. Effective management of reef systems should aim to detect changes in resilience long before complete canopy loss has occurred. Consequently, this information is critical to management of these systems in the future.

There also needs to be better understanding regarding the amount of epiphytic, filamentous or opportunist algal growth that indicates a reef ecosystem is vulnerable. This is an inherently complex process to understand, but is vital for successful management of these systems, as disturbance is a natural process for canopy regeneration (Wernberg et al. 2005; Flukes et al. 2014). In habitats degraded by nutrient enrichment, canopies are unlikely to reform following disturbance, with turfing or filamentous algae inhibiting recruitment (Gorman & Connell 2009; Moy & Christie 2012; Strain et al. 2014; Layton et al. 2019). Sustained presence of epiphytic and filamentous algae is a sign of an ecosystem under nutrient stress, and as the presence of these species increases, the potential for canopy loss also becomes more likely (Moy & Christie 2012; Norderhaug et al. 2015). In this study, the Lomas Point site was closest to salmon farm derived nutrient inputs, with epiphytic and opportunistic algae was observed at this site on both sampling occasions, although levels were generally less than 20% and canopy cover was over 70%. Thus, for the time period surveyed the site appeared able to cope with the level of organic enrichment sustained. It is worth noting that change at this site outside of this survey period is unknown and thus we are unable to draw conclusions around how the time period of exposure may influence the level of enrichment observed. In contrast, the sites surveyed by Oh et al. (2015) were more likely to be vulnerable, with epiphytic, filamentous and opportunistic algae recorded at approximately 50% at 100 m from salmon farms. It is highly likely that vulnerability to organic enrichment is site specific, with environmental factors, particularly physical factors that determine the exposure to nutrients (e.g., hydrodynamics and wave exposure) rather than distance to the source per se, playing a critical role. If establishing a monitoring program using the RVA technique, initial experimental design and the subsequent interpretation of results would undoubtedly be enhanced through the use of tools such as hydrodynamic modelling, that could aid in identification of areas most susceptible to organic enrichment (Chapter 7.3).

6.5.4.3 Robustness of the functional approach to assessing reef ecosystems

As part of the evaluation of our method, we assessed both the power to detect change and diver bias. For this method to be incorporated into an ongoing monitoring program, it is vital to detect an ecologically meaningful change. For canopy cover we found that rapid visual assessment on 12 quadrats could reliably detect a 42% change in September and a 24% change in February, averaged across sites (excluding the Actaeons). If a 66% reduction in canopy cover on *E. radiata* dominated reefs in Tasmania might be considered biologically significant (Flukes et al. 2014), then the RVA approach has the power to accurately detect meaningful levels of change and provide an early warning of ecosystem degradation.

This study found the power to detect change was influenced by site heterogeneity and as the percentage cover of a parameter increased, sites became less heterogenous and consequently, ability to detect change increased (c.f. Maxwell & Jennings 2005; Blanchard et al. 2008). The power analysis also provided useful information regarding the patchiness of macroalgae at particular sites. For example, canopy cover was similar at Southport and Sisters Bay at approximately 75% in February; however, the ability to detect change differed between sites at 20% and 12% respectively. This would suggest that Southport had a patchier canopy distribution. It was observed that epiphytic, filamentous, and opportunistic algae tend to be considerably patchy in distribution, particularly at low abundances. Consequently, the power to detect change in organic enrichment parameters was markedly less than in parameters such as canopy or sub-canopy cover, which tended to have slightly more even distributions. While power would be expected to increase as percentage cover of enrichment parameters increased, at low cover, a much larger change is required before it will be statistically significant. Further study is required to determine how this may relate to biologically significant change within these enrichment parameters.

Discrepancy between divers across all key parameters was low (less than 10%) and well within the power of the method to detect change. This indicates that when using experienced, well-trained divers, the RVA method is highly transferable between individuals. Solid training is key to reducing diver bias. The divers in this study engaged in several training dives before undertaking the method independently. However, there will always be some level of inherent variation in measurements, even when the same diver is resurveying the same quadrat. The split-quadrat method for estimating cover provides good accuracy between divers. Accuracy could potentially have been increased by switching to point count analysis, but this would also increase dive time. Using the split-quadrat method, two well-trained divers can complete a site in approximately 45 minutes. As a key study objective was to develop a reliable and accurate method that was rapid enough to facilitate the survey of multiple sites within a day, this level of accuracy might be considered acceptable.

One notable limitation of this method was the inability to effectively quantify the canopy cover of *M. pyrifera* using quadrats. At the Actaeons site where *M. pyrifera* dominated, mean canopy cover in the assessment appeared to be less than 10%, even though the canopy was often so dense on the surface as to be impenetrable by boat (C. White pers. obs). Unfortunately, it was impossible to accurately assess surface canopy cover using benthic quadrats. This is an area that would benefit from further method development. One approach might be to undertake the evaluation in situ, with divers counting stipes within the survey area. While this data would not be directly comparable with other canopy forming macroalgae measurements, a time-series of stipe counts would at least allow for within-site comparison over time. Other possible options include sampling using remote or developing technologies and including towed video, aerial photography, satellite, or drones. Attempts by Crawford and Harwin (2018) to characterise *M. pyrifera* beds in the D'Entrecasteaux Channel using drones were largely unsuccessful, as the technology was unable to adequately capture submerged vegetation and prevailing adverse weather conditions often limited/ prohibited image capture. However, technology is improving rapidly which will help to overcome these limitations.

The final key limitation to this method is that being a functional survey, it cannot capture the effects of organic enrichment on temperate reef biodiversity. Macroalgal canopy was the only functional group where a full species characterisation occurred. When functional ecology is significantly altered by organic enrichment, there will be flow-on effects to biodiversity, with algae, invertebrate and fish populations potentially affected (Stuart-Smith et al. 2015). Organic enrichment tends to be associated with a reduction in food web complexity, with high nutrient conditions favouring short-lived faster growing opportunistic species (Stuart-Smith et al. 2015; Ling et al. 2018). This will in turn affect the ecosystem services provided by the temperate reef environment (Smale et al. 2013; Teagle et al. 2018). Long term monitoring should include an approach that integrates both functionality and biodiversity.

6.5.4.4 Conclusion

In conclusion, functional evaluation of temperate reef ecosystems using the rapid visual assessment (RVA) approach can provide a measure of broadscale organic enrichment. Functional surveys are relatively quick and easy and as such can be undertaken multiple times per year, with results providing an ongoing assessment of whether organic enrichment is having an impact at any given site, regardless of the source of that enrichment. Biodiversity surveys can provide information regarding the extent or severity of this impact and could potentially be undertaken either on a much larger timeframe, or when the functional survey indicates a significant change has occurred at a site. Using an integrated approach allows for a more holistic assessment of temperate reef ecosystems and would enable better management of these resources into the future.

It is clear from this study a much longer time series is needed to fully validate this approach, with a greater understanding of the temporal variation inherent in the system required to better understand the nature of organic enrichment at affected sites. A longer time series would support the development of an organic enrichment index, similar to that produced for soft sediment. This would aid in the establishment of trigger values and tipping points at specific sites, providing tangible guidelines for management. While the proposed RVA method can detect broadscale organic enrichment, it could not attribute causality in a multi-use system. Detailed baselines (i.e. both biodiversity and RVA), validation based around robust data and a gradient design would be important to properly infer causality. A combination of approaches would also aid in attribution, such as an understanding of local hydrodynamics or the use of dispersal modelling for identifying areas that are likely to be at greater risk from any point source of pollution (Chapter 7.3). As temperate reef ecosystems are multi-trophic, multi-dimensional habitats, there is an inherent level of complexity that will be challenging to manage. However, the development of tools such as this enables rapid insight into ecosystem health will certainly aid this process into the future.

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Supplementary content

S1: Scorecard developed for assessing functional change on temperate reef ecosystems

Circle Quadrat #	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Total% canopy												
Pcom / Sarg%												
Era / others%												
% Sub canopy brown + major spp.												
% Sub-canopy green + major spp.												
% Sub-canopy red												
% Epiphytic algae on kelp												
% Filamentous algae												
% *Ulva/ Chaetomorpha*												
% *Asparagopsis*												
Substrate characterisation												
% UALC & type Pink vs. Att. Red	P R	P R	P R	P R	P R	P R	P R	P R	P R	P R	P R	P R

% Sponge & type												
% Turfing algae												
# Feather stars												
MMI spp and #												
Dust on algae (H/M/L/N)?												
Enc. spp. on algae? (H/M/L/N)												

Table S2. Parameters contributing to 70% dissimilarity (highest to lowest average dissimilarity) between sites determined through SIMPER analysis for a) September 18 (above grey line) and b) February 19 (below grey line).

	Actaeons	Southport	Sisters Bay	Lady Bay	Tower Bay	Lippies Pt	Scott Pt	Lomas Pt	Esperance	Penguin	Zuidpool
Actaeons		Canopy SC Green SC Red Red enc.	Canopy SC Green Pink enc. SC Red	Canopy SC Green Pink enc. Red enc.	Canopy SC Green Pink enc. SC Red	SC Green Canopy SC Red Red enc.	SC Green Canopy SC Red Pink enc.	Canopy SC Green Epiphyte Pink enc. SC Red	SC Green Canopy SC Red Pink enc.	SC Green Canopy Red enc. Pink enc. SC Red	SC Green Red enc. Canopy SC Red Pink enc.
Southport	Canopy SC Red Pink enc.		SC Red Pink enc. Canopy Red enc.	Canopy SC Red Pink enc. Red enc.	SC Red Pink enc. Canopy Red enc.	Canopy SC Red Red enc. Pink enc.	Canopy SC Red Pink enc Red enc.	Epiphyte Canopy Red enc. Pink enc. TSS	Canopy Pink enc. Red enc. SC Red	Canopy Pink enc. Red enc. SC Red TSS	Red enc. Canopy Pink enc. TSS Sponge
Sisters	Canopy Pink enc SC Red	SC Red Pink enc. Sponge SC Brown Canopy		Canopy Red enc. SC Red Pink enc.	Canopy Pink enc. SC Red Re enc.	SC Red Canopy Red enc. Pink enc.	SC Red Pink enc. Canopy Red enc.	Canopy Pink enc. Epiphyte SC Red SC Brown	SC Red Canopy Red enc. Pink enc.	Pink enc. SC Red Canopy Red enc.	Red enc. Pink enc. SC Red Canopy
Lady	Canopy SC Red Pink enc.	Pink enc. Red enc. Canopy SC Brown SC Red	SC Red Pink enc. Canopy SC Brown Red enc.		Canopy SC Red Pink enc. Red enc.	Canopy SC Red Red enc. Pink enc.	Canopy SC Red Red enc. Pink enc.	Epiphyte Canopy Pink enc. Red enc. SC Red	Canopy SC Red Red enc. Pink enc.	Canopy Pink enc. Red enc. SC Red	Pink enc. Red enc. Canopy SC Red
Tower	Canopy SC Red Pink enc. Red enc.	Canopy SC Red Pink enc. Red enc. Sponge	Canopy SC Red Pink enc. Red enc.	Canopy Pink enc. Red enc. SC Red		Canopy SC Red Pink enc.	Canopy Pink enc. SC Red SC Brown	Pink enc. SC Red Epiphyte Canopy SC Brown	SC Red Canopy Pink enc. Red enc.	Canopy Pink enc. SC Red Red enc.	Pink enc. SC Red Canopy Red enc.
Lippies Pt	Canopy SC Red Pink enc.	Pink enc. SC Red Canopy Red enc. Sponge	Pink enc. SC Red Canopy SC Brown Red enc.	Pink enc. Canopy SC Red Red enc. SC Brown	Canopy Pink enc. SC Red Red enc.		SC Red Canopy Red enc. Pink enc.	Canopy Epiphyte SC Red Red enc. Pink enc.	Canopy SC Red Red enc. Pink enc.	Canopy Pink enc. SC Red Red enc. TSS	Red enc. Pink enc. Canopy SC Red TSS
Scott Pt	Canopy SC Red Pink enc.	Canopy Red enc. Pink enc.	SC Red Pink enc. Canopy	Pink enc. Canopy Red enc.	Canopy Pink enc. Red enc.	Pink enc. Canopy SC Red		Canopy Epiphyte SC Red	SC Red Canopy Pink enc.	SC Red Canopy Pink enc.	Red enc. SC Red Canopy

		Sponge SC Red	Red enc. Sponge	SC Red SC Brown	SC Red TSS	Red enc.		Pink enc. TSS	Red enc.	Red enc. TSS	Pink enc. TSS
Lomas Pt	Canopy SC Red TSS Pink enc.	TSS Pink enc. Red enc. Canopy Sponge	SC Red Pink enc TSS Red enc. Canopy	Pink enc. Red enc. TSS Canopy SC Red	Canopy Pink enc. TSS Red enc. SC Red	Pink enc. Canopy SC Red TSS Red enc.	Pink enc. Canopy TSS Red enc. SC Red		Canopy Epiphyte Pink enc. Red enc. SC Red	Canopy Epiphyte Pink enc. Red enc. TSS	Canopy Red enc. Epiphyte TSS Pink enc.
Esperance	Canopy SC Red Pink enc.	Red enc. Canopy Pink enc. Sponge SC Brown	SC Red Red enc. Pink enc. SC Brown	Pink enc. Red enc. Canopy SC Red SC Brown	Canopy Red enc Pink enc. SC Red	SC Red Canopy Pink enc. Red enc.	Canopy Pink enc. Red enc. SC Red SC Brown	Pink enc. TSS Red enc. Canopy SC Brown		Canopy Pink enc. Red enc. SC Red TSS	Pink etc. Canopy Red enc. TSS Sponge
Penguin	Canopy SC Red Red enc. Pink enc. Epiphyte	Canopy Epiphyte SC Red Red enc. SC Brown Sponge	Canopy SC Red Epiphyte Red enc. Pink enc. Filamentous	Canopy Pink enc. Epiphyte Red enc. SC Red SC Brown	Canopy Epiphyte Red enc. Pink enc. SC Red Filamentous	Canopy Pink SC Red Epiphyte Red Filamentous	Canopy Epiphyte SC Red Red enc. Pink enc. Filamentous	Canopy Pink enc. Red enc. TSS Epiphyte SC Red	Canopy Epiphyte Pink enc. Red enc. SC Red SC Brown		Red enc. Canopy Pink enc. TSS SC Red
Zuidpool	Canopy SC Red Pink enc. Red enc. Epiphyte	Canopy SC Red Pink enc. Epiphyte Red enc. Sponge	Pink enc. Canopy Epiphyte SC Red Red enc. Sponge	Pink enc. Canopy SC Red Red enc. Epiphyte Sponge	Canopy Pink enc. SC Red Red Epiphyte Sponge	Pink enc. Canopy SC Red Epiphyte Red enc. TSS	Canopy SC Red Pink enc. Red enc. Epiphyte Sponge	Canopy SC Red Pink Red TSS	SC Red Pink enc. Canopy Red enc. Epiphyte Sponge	Canopy SC Red Epiphyte Pink enc. Red enc. Filamentous	

NB: SC = Sub canopy, Enc. = Encrusting algae, TSS = Turf/Sand/Sediment matrix

S3: Model outputs

3-Way PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Zone	2	27483	13741	1.9404	0.0314	4735
Time	1	15132	15132	6.305	0.0038	9949
Site(Zone)	8	56652	7081.5	13.326	0.0001	9886
Zo x Time	2	2769	1384.5	0.57686	0.8063	9946
Si(Zo) x Ti	8	19200	2400	4.5164	0.0001	9897
Res	242	1.286E+05	531.4			
Total	263	2.5024E+05				

S4 Diver comparison

2-way PERMANOVA table of results

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Site	10	70526	7052.6	53.679	0.0001	9883
Diver(Site)	11	1445.2	131.39	0.20895	1	9886
Res	154	96831	628.78			
Total	175	1.688E+05				

2-way ANOVA individual parameters

Response: Canopy

```
             Sum Sq    Df        F value    Pr(>F)
Site         76506     9         14.0680    5.617e-16 ***
(Diver)Site 283        10        0.0468      1
Residuals    84595     140
```

Response: SCGreen

```
             Sum Sq  Df F value Pr(>F)
Site         11988.1   9  18.074 <2e-16 ***
Diver(Site) 18.4  10   0.025      1
Residuals    10317.6 140
```

Response: SCRed

	Sum Sq	Df	F value	Pr(>F)
Site	27254	9	10.2214	5.164e-12 ***
Diver(Site)	358	10	0.1207	0.9995
Residuals	41476	140		

Response: PinkEnc

	Sum Sq	Df	F value	Pr(>F)
Site	17669	9	6.8312	4.152e-08 ***
Diver(Site)	532	10	0.1849	0.9971
Residuals	40233	140		

Response: RedEnc

	Sum Sq	Df	F value	Pr(>F)
Site	13715.0	9	6.8710	3.717e-08 ***
Diver(Site)	1209.4	10	0.5453	0.8553
Residuals	31050.0	140		

Response: Epiphyte

	Sum Sq	Df	F value	Pr(>F)
Site	5292.3	9	8.3753	6.161e-10 ***
Diver(Site)	99.9	10	0.1423	0.9991
Residuals	9829.4	140		

Response: Filamentous

	Sum Sq	Df	F value	Pr(>F)
Site	1472.8	9	3.3552	0.0009496 ***
Diver(Site)	18.9	10	0.0387	0.9999978
Residuals	6828.2	140		

Response: Asparagopsis

	Sum Sq	Df	F value	Pr(>F)
Site	275.23	9	8.0249	1.575e-09 ***
Diver(Site)	30.25	10	0.7938	0.6348
Residuals	533.50	140		

6.6 Development of remote techniques for assessing change

6.6.1 Introduction

Primary productivity of temperate reefs is underpinned by canopy forming macroalgae, which are sensitive to both elevated and deficient nutrient concentrations in the water column (Munda 1993; Morand & Briand, 1996; Pinedo et al. 2007; Connell et al. 2008; Juanes et al. 2008; Gorman et al. 2009; Van Rein et al. 2009; Oh et al. 2015). Anthropogenically-derived nutrient inputs can manifest shifts away from perennial species such as canopy forming macroalgae to turfing species. While these turfing species tend to be ephemeral, in areas of elevated nutrients they can persist, forming their own biogenic matrix, trapping sediment and impeding the establishment of macroalgae (Connell et al. 2008; Gorman et al. 2009). Through these mechanisms anthropogenic activities in coastal areas that lead to nutrient enrichment can have a detrimental effect on the adjacent marine environment (Dauer et al. 2000; Benedetti-Cecchi et al. 2001). Activities such as recreational boating, the operation of wastewater treatment plants, housing and land development, agriculture, forestry and finfish aquaculture can all contribute to nutrient enrichment in coastal environments. Whilst there have been a limited number of studies into fish farming impacts to reef communities in Tasmania, Ross and Macleod (2013) emphasized the need for broad-scale monitoring and studies. In a targeted study examining the effect of farming over a gradient, changes in biological structure could be correlated to fish farming activities up to 400 m from farms (Oh et al. 2015). However, further work is required to better understand the potential of these broad-scale impacts from nutrient inputs.

To assess the condition of reef habitats, diver-based surveys remain the most used method. The existing, widely accepted diver-based survey methods in Tasmania provide high-quality biodiversity data for temperate rocky reef habitat by assessing demersal and cryptic fish species and numbers, motile and sessile macroinvertebrate species and numbers, and algae species (Edgar & Barrett 1999). However, only 5 m^2 of algae is assessed within the total 2000m^2 of reef assessed under this method (Valentine et al. 2016). These biodiversity-centric methods are an effective means to establish a baseline of habitat species and structure and examine broader habitat changes in response to large scale events such as climate change, or community shifts relating to the establishment of Marine Protected Areas (Edgar & Barrett 1999; Borja et al. 2013). However, they are unlikely to be sensitive enough to detect a loss of resilience in temperate reef ecosystems due to organic enrichment. Such changes can be in the form of canopy forming macroalgae deterioration and/or the increasing presence of specific enrichment indicators such as epiphytic or turfing algae.

An alternate option to biodiversity surveys is to adopt a targeted monitoring approach and assess changes to habitat functionality. Methods to address this were developed through this project (see Chapter 6.5) and trialled over a broadscale enrichment gradient in the southern D'Entrecasteaux Channel. This rapid visual assessment (RVA) survey method was purposefully developed to be efficient enough to be able to cover a large number of sites in one day yet detailed enough to identify shifts in ecosystem function (Chapter 6.3). A key limitation of both the Edgar-Barrett and RVA survey methods is they are both diver dependent, meaning that survey activity is restricted by factors including allowable safe dive time and diver expertise. As such, the area that can be surveyed in a day is limited. Remote visual based techniques of photo quadrat analysis and/or towed camera allow for greater spatial coverage by maximising dive time or eliminating divers all together (Harriott 1999;

Morrison & Carbines 2006; Leujak & Ormond 2007; Shortis et al. 2007; Assis et al. 2008; Juanes et al. 2008; Van Rein et al. 2009; Mallet & Pelletier 2014; Walsh et al. 2017). Photo quadrats can be taken in a fraction of the time it takes to complete an in-situ assessment. Midwater video systems have been used for 60 years and are accepted means to quantify changes to macroalgae, seagrass and coral (Mallet & Pelletier 2014). Photo quadrats have the potential to make the most efficient use of dive time, whereby it would be expected to take approximately 10-15 minutes to photograph of all 12 quadrats of a rapid visual assessment in this case, compared to 40-45 minutes per site to undertake an in situ RVA. It is therefore expected that a much larger number of sites could potentially be covered over a similar length of time required for diver surveys. Using a towed camera removes the requirement for divers all together, and as it is operated from a boat is expected to allow much greater spatial coverage than in situ or photo-quadrat techniques (Harriott 1999; Morrison & Carbines 2006; Leujak & Ormond 2007).

The primary aim of this research was to develop methods for accurately assessing temperate reef habitats for organic enrichment using the remote techniques of towed camera and photo quadrat analysis. The objectives of this study were to:

- Determine a list of robust metrics for assessment using towed camera and photo-quadrats.
- Compare and validate the data collected from both photo quadrat and towed camera methods to understand reef dynamics in the lower D’Entrecasteaux Channel.
- Consider advantages and limitations of both methods.
- Make recommendations for improvement of the methods into the future.

6.6.2 Methods

6.6.2.1 Study Area

Eleven sites on the rocky reefs of the southern D’Entrecasteaux Channel were selected for this study (Figure 6.6-1). Sites corresponded to those established as part of the FRDC 2015-024 reef biodiversity and ongoing RVA surveys. Six of the sites (Zuidpool, Penguin Bay, Esperance Point, Lomas Point, Scott Point and Lippies Point) are exposed in some degree to sources of organic enrichment including the Huon and Esperance Rivers, industry and several small townships, along with outputs from farming of Atlantic salmon (*Salmo salar*). In contrast, five of the sites to the south (Tower Bay, Lady Bay, Sisters Bay, Southport, Actaeons) are more exposed to the Southern Ocean and relatively unimpacted in terms of obvious anthropogenic sources of organic enrichment (Figure 6.6-1).

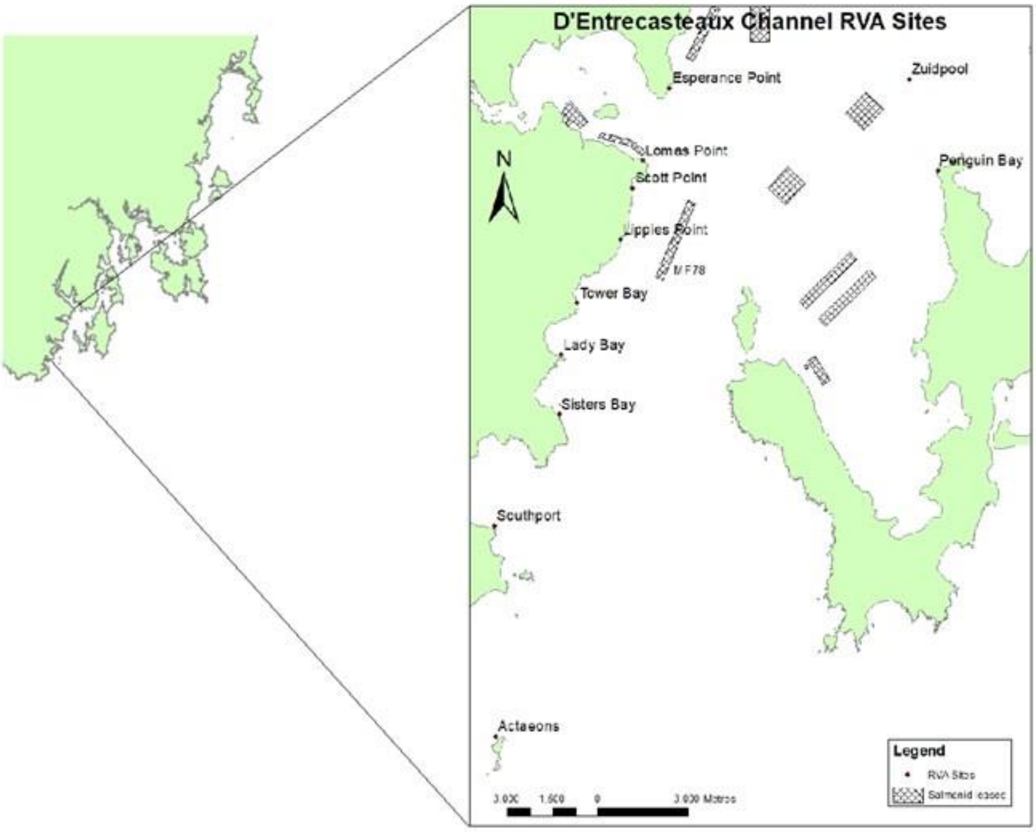

Figure 6.6-1 Southern D'Entrecasteaux Channel with rapid visual assessment (RVA) sites labelled and salmon leases marked. Note that towed camera transects of Scott Point incorporated Lippies Point, and Tower Bay incorporated Lady Bay.

6.6.2.2 Photo Quadrats

6.6.2.2.1 Data collection

In February 2018 at each of the 11 established RVA sites (Figure 6.6-1), 12 x 1 m^2 quadrats were laid by divers using SCUBA at a depth of approximately 5 m. Quadrat locations were fixed, as per the established RVA method (Chapter 6.5). Quadrats were photographed using a Canon G16 12 mp camera coupled with a Fantasea Line underwater housing and BigEye Wide Angle fisheye lens. Photographs were taken from directly above the quadrat to ensure the quadrat occupied as much of the image as possible. No algae were moved or removed from the quadrat prior to photographing.

In the laboratory, photos were digitally cropped and rotated when needed prior to importing to Coral Point Count with Excel Extension (CPCe) for analysis. In CPCe, a grid of 64 points was laid over each image. Algae species, mobile invertebrates, substrate characteristics and enrichment indicators identified below each point were recorded where possible. Categories assessed were user-defined and loaded into CPCe prior to photo assessment. In this case, categories were chosen to align directly with those assessed for the RVA method (Chapter 6.3) and to fall under the broad categories of canopy forming algae species, subcanopy brown, green and red algae, enrichment indicators, and major mobile invertebrates. Photo quadrat data generated through Coral Point were exported directly into Excel for analysis.

6.6.2.2.2 Data analysis

To assess the robustness of each of the key biota categories, a measure of uncertainty was calculated. The first five quadrats for each site were assessed for the ability to definitively identify each of the 64 points against the key biota categories of the RVA method (i.e. ‘scorable or not’). Scores for biota categories were averaged across the five quadrats, giving an average value for each category for each site, which was then converted to a percentage. This metric was taken as a representation of certainty in the method to assess key biota categories, and the less scorable a category was, the greater uncertainty there was in the assessment of this category. It was then inferred that those biota categories with a low value, i.e. less than 5% uncertain, were classified as robust, and therefore assessment of photo quadrat using CPCe was a valid method for this category.

To characterise the reef systems in the southeast D’Entrecasteaux Channel (SE Channel) using photo quadrat analysis, mean canopy cover was determined for each site in the survey area, with the effect of site (fixed factor) determined through a one-way ANOVA and Tukey’s post-hoc testing. Mean percentage cover of the canopy species *Phyllospora comosa*, *Ecklonia radiata* and *Macrocystis pyrifera* was determined for each site.

6.6.2.3 Towed Camera

6.6.2.3.1 Data collection

In February 2018 a towed camera unit was operated from a vessel by tether and winch, utilising an HD Sony Handycam and two LED flood lights. The camera was positions to capture approximately 60-70% substrate to 30-40% horizon in the footage. A pair of lasers spaced at 150 mm enabled field of view calculation and sizing of observed specimens. The towed camera was deployed on reef habitat up to 1km from the associated photo quadrat/RVA sites (Figure 6.6-1) and towed parallel to the shoreline for up to 4.5 km passing over the photo quadrat site. The tether was used to maintain the towed camera approximately 1-2 m above the habitat between the 5 and 8 m depth contours and the vessel was manoeuvred at approximately 1 knot (0.51 ms^{-1}) (Morrison & Carbines 2006; Assis et al. 2008; Nichol et al. 2009; Logan et al. 2017; Carroll et al. 2018). Due to the contiguous reef structure of the coastline south from Scott Point to Lady Bay, filming of the transects commencing at Scott Point included the Lippies Point RVA site, and Tower Bay included the Lady Bay RVA site. This maximised spatial coverage and expediated the process of undertaking the towed camera surveys in these areas.

The vessel’s date stamp and track were recorded using the onboard GPS (Carroll et al. 2018). Comprehensive field notes were taken on the day and included filming start and end times, weather conditions and general observations of reef habitat that were used in companion when post processing and analysing video footage.

In the laboratory, the SeaGIS computer program TransectMeasure was used to assess the towed camera footage. This program allows for habitat assessment to align with the vessel’s GPS tracks. A predetermined series of categories were defined and used to assess the recorded footage. Similar to the photo quadrats, categories were chosen to align as close as possible to those established for the RVA method and included:

- Percent cover and dominant species of canopy,
- *P. comosa*, *E. radiata* and *M. pyrifera* constituent percentage of canopy cover,
- Sub-canopy brown, green and red algae dominant species and percentage cover,

- Presence of the enrichment indicators epiphytic algae growth, filamentous algae, *Ulva* spp., *Chaetomorpha* spp., *Asparogopsis armata*, and percent sponge cover and type (free standing or encrusting),
- Substrate characterisation,
- Encrusting species including white worm, bryozoans and snails, and
- Major mobile invertebrates.

To accurately score against these categories, the footage was played at half speed when analysing, with the first ten seconds of every minute being viewed and analysed. Each TransectMeasure point measurement is a composite of the previous ten second interval, i.e., the above-mentioned categories are estimated over the entire ten seconds of footage, deeming it impossible to accurately record a single figure percentage algal cover for each segment of analysed video. Further to this, limitations associated with the program TransectMeasure mean that a finite number of characteristics could be recorded for each point; N=8. Hence, percentage cover was recorded categorically as unscorable, 0%, 1-20%, 20-40%, 40-60%, 60-80% or 80-100%.

6.6.2.3.2 Data analysis

Towed camera data was exported from TransectMeasure to CSV format. It was transformed using RStudio to a format that allowed assessment and graphical display of the data. RStudio was then used to transform this data, rearranging it so there was one row per frame, with biota categories listed across the columns.

To assess the robustness of each of the key biota categories, a measure of uncertainty was calculated using the unscorable class from the assessment of the towed camera footage. Unscorable points for biota categories were averaged by site and converted to a percentage. This metric was taken as a representation of certainty in the method to assess these categories. The more unscorable a category was, the greater uncertainty there was in the assessment of this category. Those biota categories with a low value, i.e. less than 5% uncertain, were classified as robust; therefore, assessment of towed camera footage using TransectMeasure was a valid method for this category.

To characterise the reefs of the SE Channel using towed camera data, standardised frequency bar graphs were produced using Microsoft Excel. To compare with the photo quadrat method of assessment, percentage canopy algae cover and characteristic species graphs were produced. Epiphytic algae were also investigated, with percent occurrence calculated to produce bar graphs.

All categories found to be robust through the uncertainty screening were then analysed spatially. The start time of each transect was recorded as local time, corrected for daylight savings, and then used in TransectMeasure to provide a reference time, i.e., time zero, from which timing of each assessed frame was subsequently based. This was aligned with the time stamp from the vessel's GPS track, giving each data point a geographical position which could then be mapped in ArcMap. RStudio was used to perform the alignment where the GPS and CSV files were combined based on time and site. From here, robust categories were retained in the spreadsheet and those with high levels of uncertainty were removed. Data points were then bundled together in RStudio to produce a GeoPackage database in a form to input to ArcMap, allowing for each of the key biota categories to be geographically displayed.

Given canopy algae's role as a key foundation species (Teagle et al. 2017), a measure for canopy algae patchiness was developed and applied to the towed camera algal canopy data. Canopy cover data was classified and grouped by percentage cover into four bins:

1. resilient canopy cover (60-100% cover)
2. intermediate canopy cover (20-60% cover)
3. low canopy cover (1-20% cover)
4. 0% canopy cover.

The first patch of canopy algae started with the commencement of the transect. The subsequent patches started at each point where there was a change or break in canopy algae cover. Canopy algae cover patches were counted chronologically by classification bin for each site giving individual patch lengths and total number of patches. A scoring system was then developed to differentiate sites by canopy cover patchiness relative to transect length.

$$Patchiness\ score = \left(\frac{total\ number\ of\ patches\ of\ each\ bin}{length\ of\ transect\ (km)}\right)$$

As the patchiness score did not consider the percentage cover of the canopy throughout any given patch, average patch length was then calculated as a percentage of the total transect length for each of the four bins for each site and graphed in Microsoft Excel.

6.6.3 Results

6.6.3.1 Analysis of metrics - photo quadrat uncertainty

Of all categories surveyed, canopy cover and canopy species were both highly robust for assessment from photo quadrat images, with all sites assessed scoring <5% uncertainty (Table 6.6-1). In levels below the canopy, all other categories had very high values of uncertainty (>90%) at five or more sites and were therefore considered not robust enough to be assessed using this technique.

Table 6.6-1 Comparative percentage uncertainty against each of the key biota metrics recorded from the photo quadrat assessment of the southern D'Entrecasteaux Channel collected in February 2018. Green highlights indicate < 5% uncertain, red highlights indicate > 90% uncertain.

	Zuidpool	Penguin Bay	Esperance Pt	Lomas Pt	Scott Pt	Lippies Pt	Tower Bay	Lady Bay	Sisters Bay	Southport	Actaeons
Canopy Cover	0	0	0	0	0	0	0	0	0	0	1
Canopy Species	0	0	0	0	0	0	0	0	0	0	1
Subcanopy Brown	91	82	91	97	96	79	88	93	91	81	6
Subcanopy Green	96	72	91	97	96	80	90	94	94	89	23
Subcanopy Red	94	85	91	97	97	66	84	94	88	90	13
Substrate	98	95	91	97	98	93	90	94	95	90	93
Sponge	98	94	91	97	98	93	90	94	95	90	93
Epiphytic Algae Growth	98	92	91	96	97	82	89	75	95	90	7
Filamentous Algae	98	94	91	97	97	82	90	94	95	90	9
Ulva/Enteromorpha	98	94	91	97	98	82	90	94	95	90	48
Asparagopsis	98	94	91	97	98	82	90	94	95	90	46
Major Mobile Invertebrates	98	94	93	97	98	82	90	95	95	90	61
Encrusting Fauna	62	92	20	84	96	82	90	71	95	90	2

6.6.3.2 Analysis of metrics - towed camera uncertainty

Algal canopy cover, canopy species, and epiphytic algae were determined with confidence using towed camera (Table 6.6-2). There was a low level of uncertainty when assessing canopy cover and canopy species, with these categories scoring < 5% uncertainty at ten of the 11 sites (Table 6.6-2). The Actaeons was the only site not to have a < 5% uncertainty for both these metrics due to entanglement of the towed camera (Table 6.6-2). The enrichment indicator metrics of epiphytic algal growth (< 5% uncertainty at nine out of 11 sites) and filamentous algae (< 5% uncertainty at seven out of 11 sites) also had low uncertainty values. However, while these metrics generally had low uncertainty values, their assessment using towed camera is not robust given they can all be found on subcanopy and substrate which are generally obscured by the canopy algae. While their presence can be confidently stated, their absence cannot be fully ascertained using these metrics. In levels below the canopy, uncertainty increases due to obscuring by the canopy algae itself.

Table 6.6-2 Comparative percentage uncertainty against each of the key biota metrics recorded from the towed camera assessment of the southern D'Entrecasteaux Channel in February 2018. Green highlights indicate <5% uncertain, red highlights indicate >90% uncertain.

	Zuidpool	Penguin Bay	Esperance Pt	Lomas Pt	Scott Pt	Lippies Pt	Tower Bay	Lady Bay	Sisters Bay	Southport	Actaeons
Canopy Cover	0	0	0	0	0	0	2	0	2	3	12
Canopy Species	0	0	0	0	0	0	0	0	2	0	12
Subcanopy Brown	78	0	50	44	38	27	23	25	22	58	14

Subcanopy Green	11	18	68	63	69	54	66	67	80	91	40
Subcanopy Red	11	21	9	56	18	19	15	36	14	67	45
Substrate	22	29	29	44	13	5	8	17	16	24	31
Sponge	0	68	97	67	91	69	72	83	94	91	71
Epiphytic Algae Growth	0	0	0	4	0	2	0	3	6	0	12
Filamentous Algae	0	6	3	4	0	2	0	17	6	0	12
Ulva/Enteromorpha	100	97	100	78	96	88	100	89	98	94	71
Asparagopsis	100	100	100	78	96	88	100	92	100	94	71
Major Mobile Invertebrates	100	97	100	93	100	100	100	100	98	100	98
Encrusting Fauna	11	53	6	56	20	47	42	28	18	39	12

6.6.3.3 Use of robust metrics to characterise southern D'Entrecasteaux Channel rocky reef

6.5.3.3.1 Photo quadrat analysis of canopy algae cover

Average canopy algae cover was determined from photo quadrat assessment at all sites in the southern D'Entrecasteaux Channel (Figure 6.6-2). Scott Point had the highest canopy cover recorded at 85 ± 3.97%, with all other sites recording relatively high values of canopy cover (> 60%) (Figure 6.6-2). The Actaeons recorded 11.2 ± 6.9% algal canopy cover, which was significantly lower than all other sites ($F_{10,117}$ = 16.96, p=<0.0001), confirmed by Tukey's post-hoc testing.

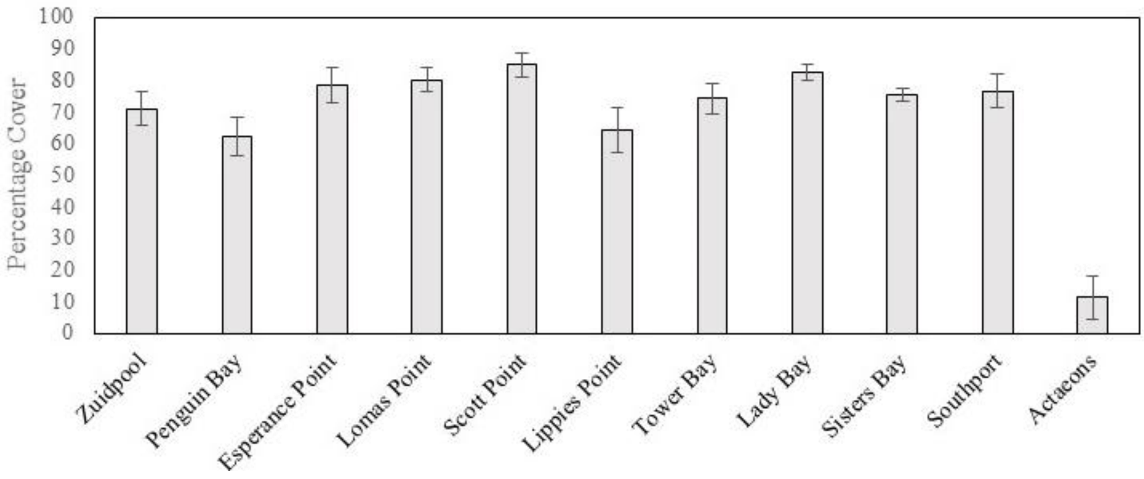

Figure 6.6-2 Total mean canopy algae cover ±SE from photo quadrat analyses of the southern D'Entrecasteaux Channel. Lowercase letters a and b indicate significant differences (P<0.05) in treatment based on Tukey's post-hoc testing.

6.6.3.3.2 Towed camera analysis of canopy algae cover

Results for percentage canopy cover using towed camera were comparable to photo quadrats, with most of the sites recording higher frequencies of relatively dense canopy cover of 80-

100% (Figure 6.6-3). The exceptions to this were Zuidpool, Penguin Bay, Lady Bay and Actaeons, where more varied canopy categories indicated less canopy cover across the length of each video transect.

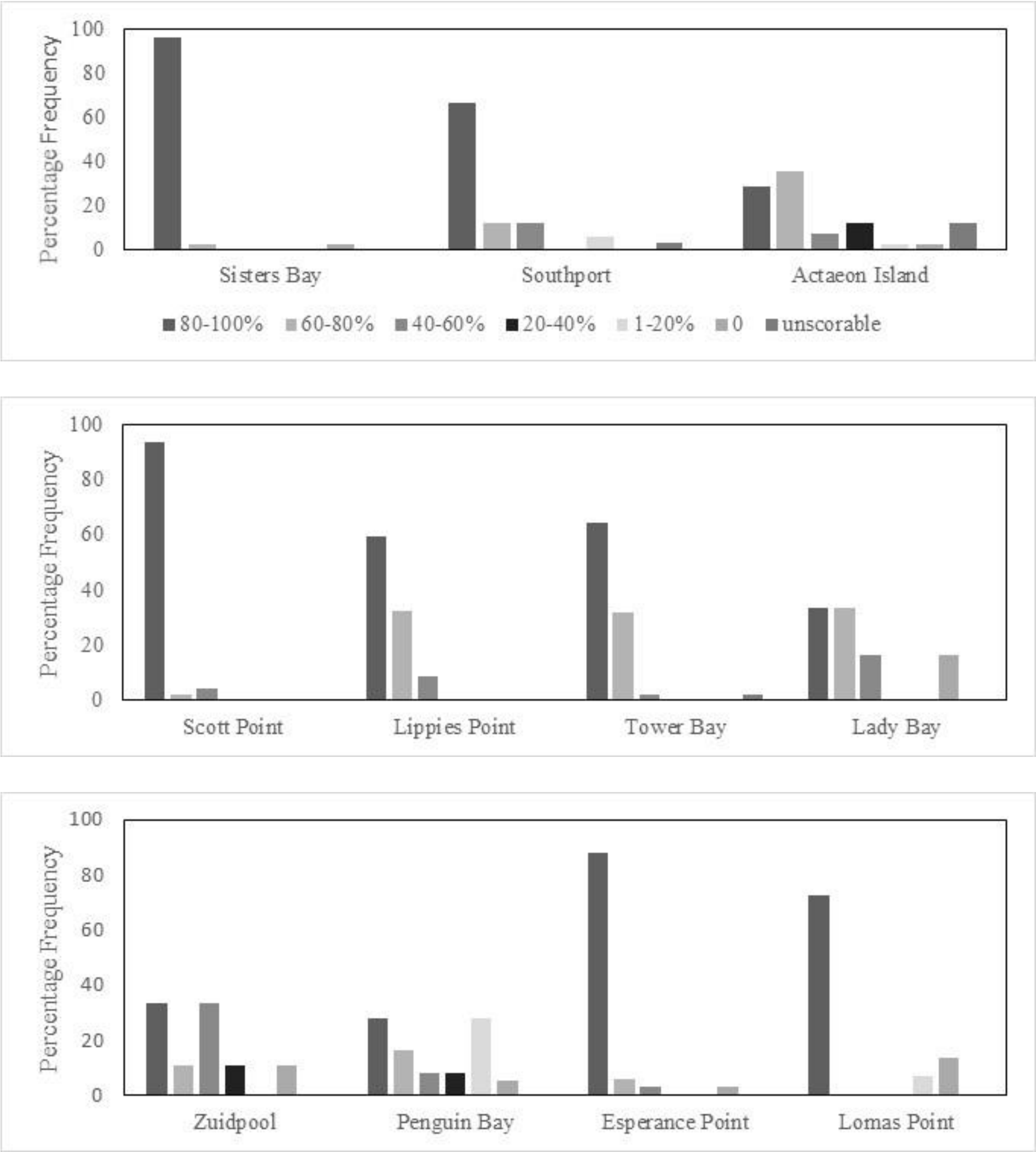

Figure 6.6-3 Percentage frequency of algal canopy cover for the towed camera survey of the southern D'Entrecasteaux Channel.

The spatial analysis of canopy cover showed variations in canopy algae cover along the length of the video transects (Figure 6.6-4). Particularly dense canopy cover occurred around the northern end of the Scott Point transect and the southern end of the Sisters Bay transect (Figure 6.6-4). Patches of low or no canopy could be clearly observed at Tower Bay and Lady Bay where the towed camera passed over areas of sand.

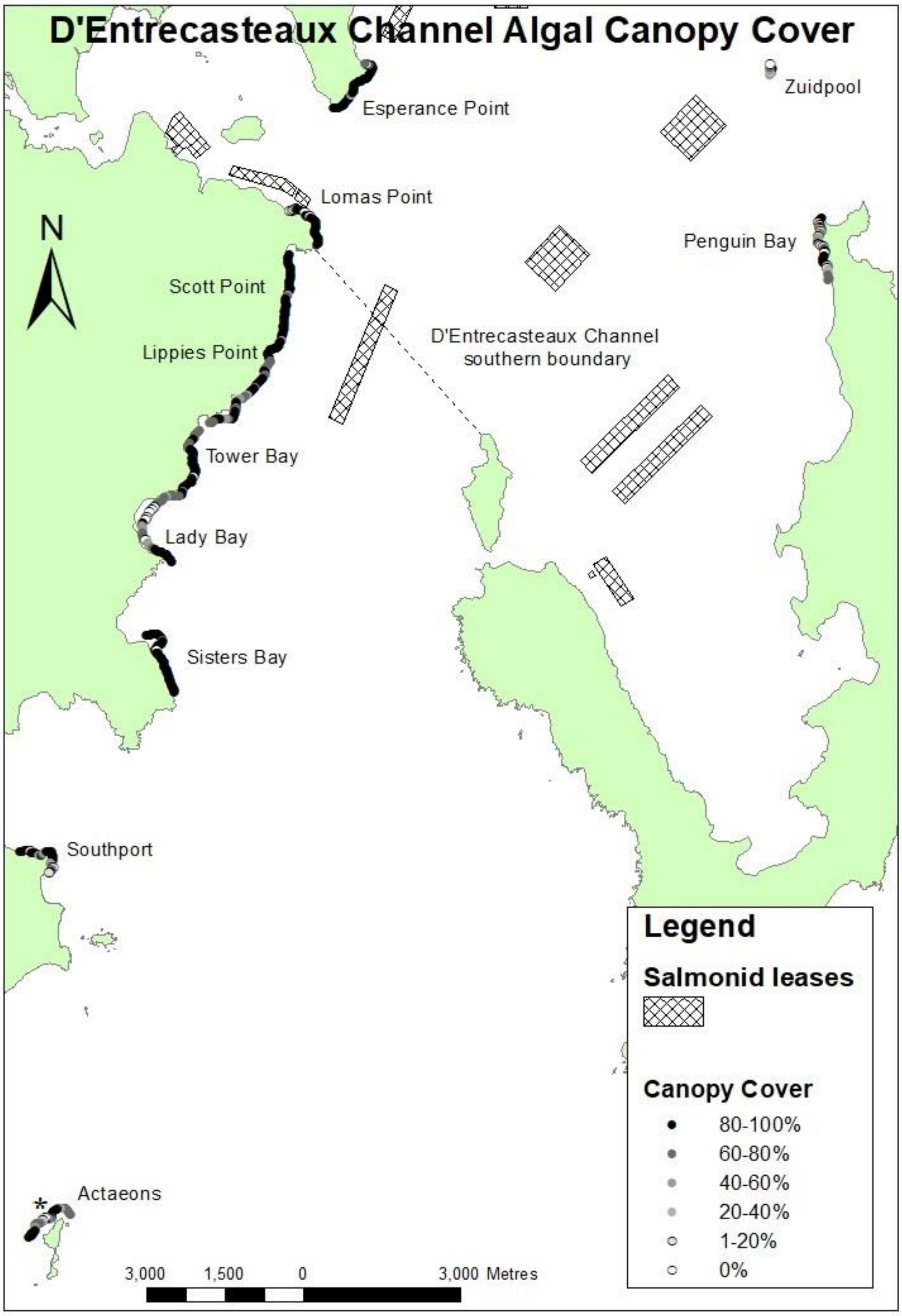

Figure 6.6-4 Canopy cover percentage from towed camera data in the southern D'Entrecasteaux Channel collected in February 2018. Farm leases are depicted as cross-hashed polygons. Note southern D'Entrecasteaux Channel border is marked as a dotted south-easterly line between Scott Point to Partridge Island on Bruny Island. Actaeons RVA site is located to the SE of the *.

Patches of low levels of canopy algae were seen at Zuidpool, Penguin Bay, Esperance Point and Lomas Point and occurred in areas that were in closer proximity to anthropogenic activity and freshwater inputs. In the Actaeons transect canopy cover was low in the area where the photo quadrat analysis took place, which can be seen to the southeast of the (*) mark in Figure 6.6-4.

6.6.3.3.3 Analysis of characteristic canopy algae composition

Photo quadrat analysis showed that *P. comosa* was the most common canopy species, being dominant at eight of the 11 sites, from Esperance Point in the north to Southport in the south (Figure 6.6-5). Zuidpool, Penguin Bay and Actaeons were the only sites where *P. comosa* was not dominant, with Zuidpool and Penguin Bay dominated by *E. radiata* and Actaeons a mixed assemblage of *P. comosa, E. radiata* and *M. pyrifera* (Figure 6.6-5).

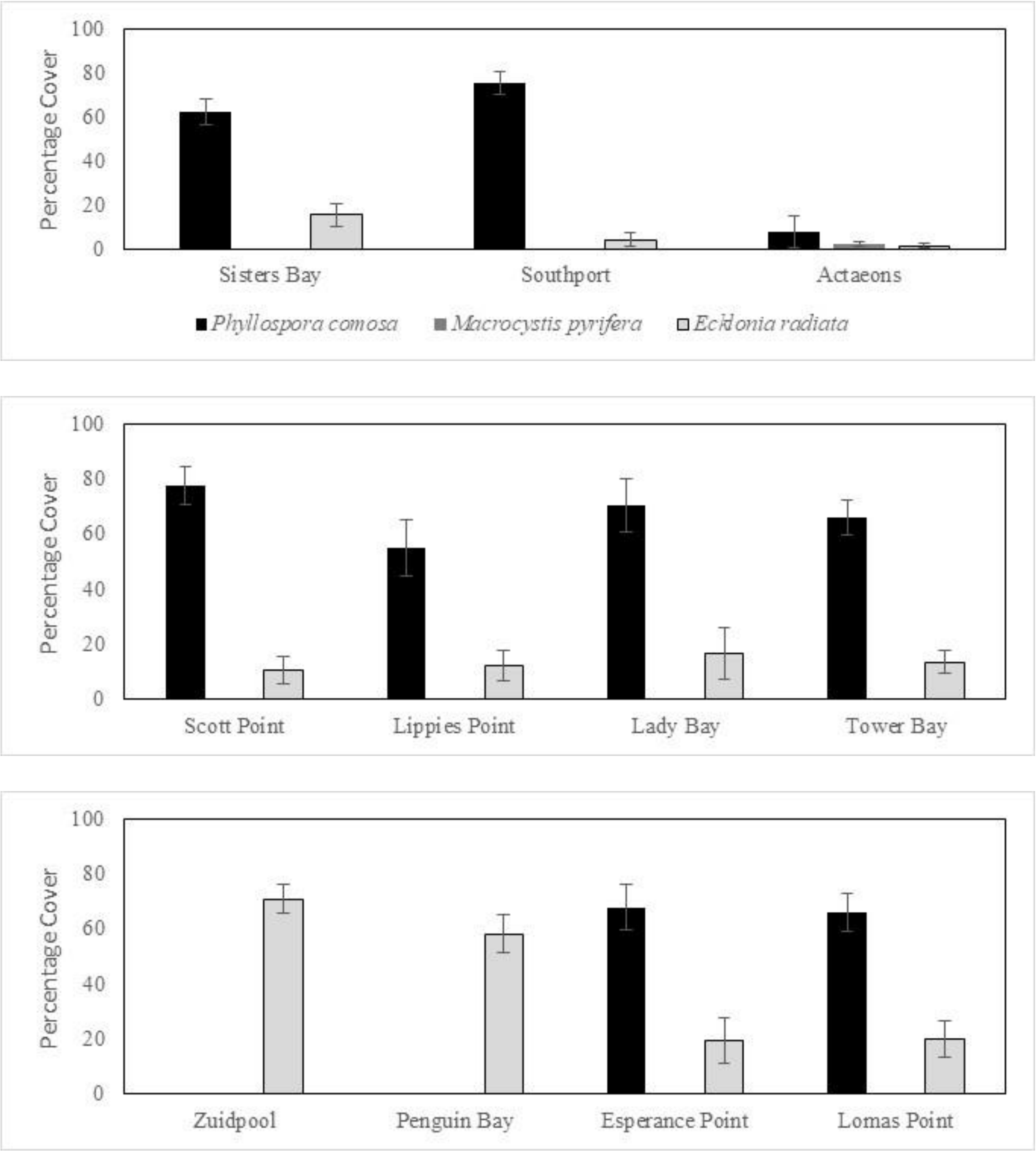

Figure 6.6-5 Average percentage canopy algae species composition ±SE as recorded from the photo quadrat assessment of the southern D'Entrecasteaux Channel.

Canopy algae species assessed from towed camera was comparable to the photo quadrat assessment (Figure 6.6-6). The northernmost sites of Zuidpool and Penguin Bay were dominated by *E. radiata*, while the more exposed areas from Esperance Point south to Southport were dominated by *P. comosa*. Actaeons recorded mixed canopy composition of *M. pyrifera* dominated by *P. comosa* with an absence of *E. radiata*.

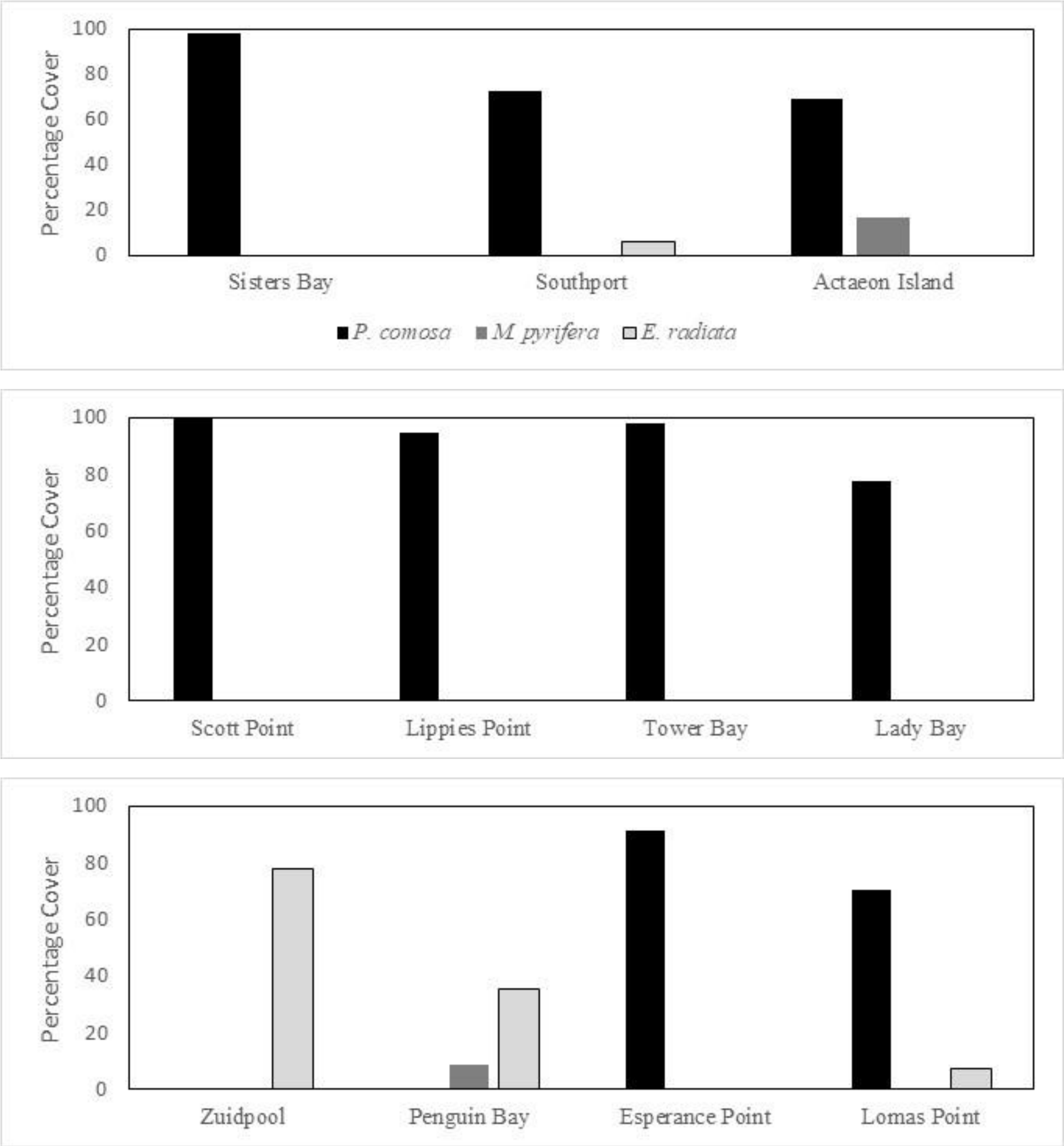

Figure 6.6-6 Percentage frequency of characteristic algal canopy species for the towed camera survey of southern D'Entrecasteaux Channel.

When viewed spatially, *P. comosa* dominated the survey area, except for the sheltered sites of Zuidpool, Penguin Bay and a small section of the Lomas Point transect, where *E. radiata* was dominant (Figure 6.6-7). One comparatively large contiguous area of *M. pyrifera* was observed in the central section of the transect at the Actaeons site which aligns with the photo quadrat site that is shown to the southeast of the (*) in Figure 6.6-7.

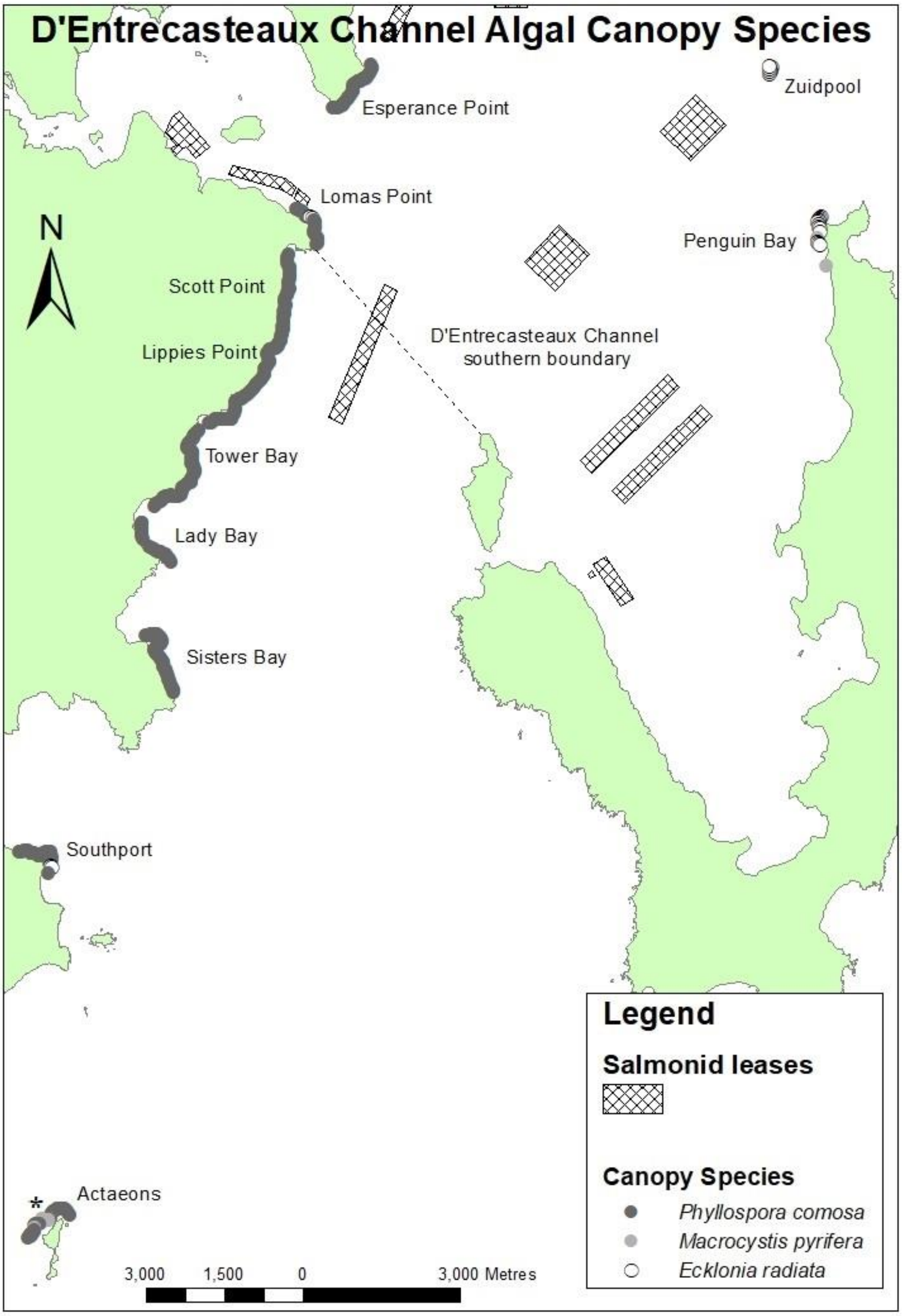

Figure 6.6-7 Characteristic algae canopy species from towed camera data in the southern D'Entrecasteaux Channel area collected in February 2018. Farm leases are depicted as cross-hashed polygons. Note southern D'Entrecasteaux Channel border is marked as a dotted line between Scott Point (just south of Lomas Point) south-easterly towards Partridge Island on Bruny Island. Actaeons RVA site is located to the SE of the *.

6.6.3.3.4 Towed camera analysis of epiphytic algae

Data from towed camera for epiphytic algae cover had lower uncertainty values than most enrichment categories and thus spatial patterns were examined further. All sites recorded the presence of epiphytic algae to some extent. The sites of Lomas Point, Lady Bay and Southport all recorded the highest frequencies of epiphytic algae occurrence (Figure 6.6-8).

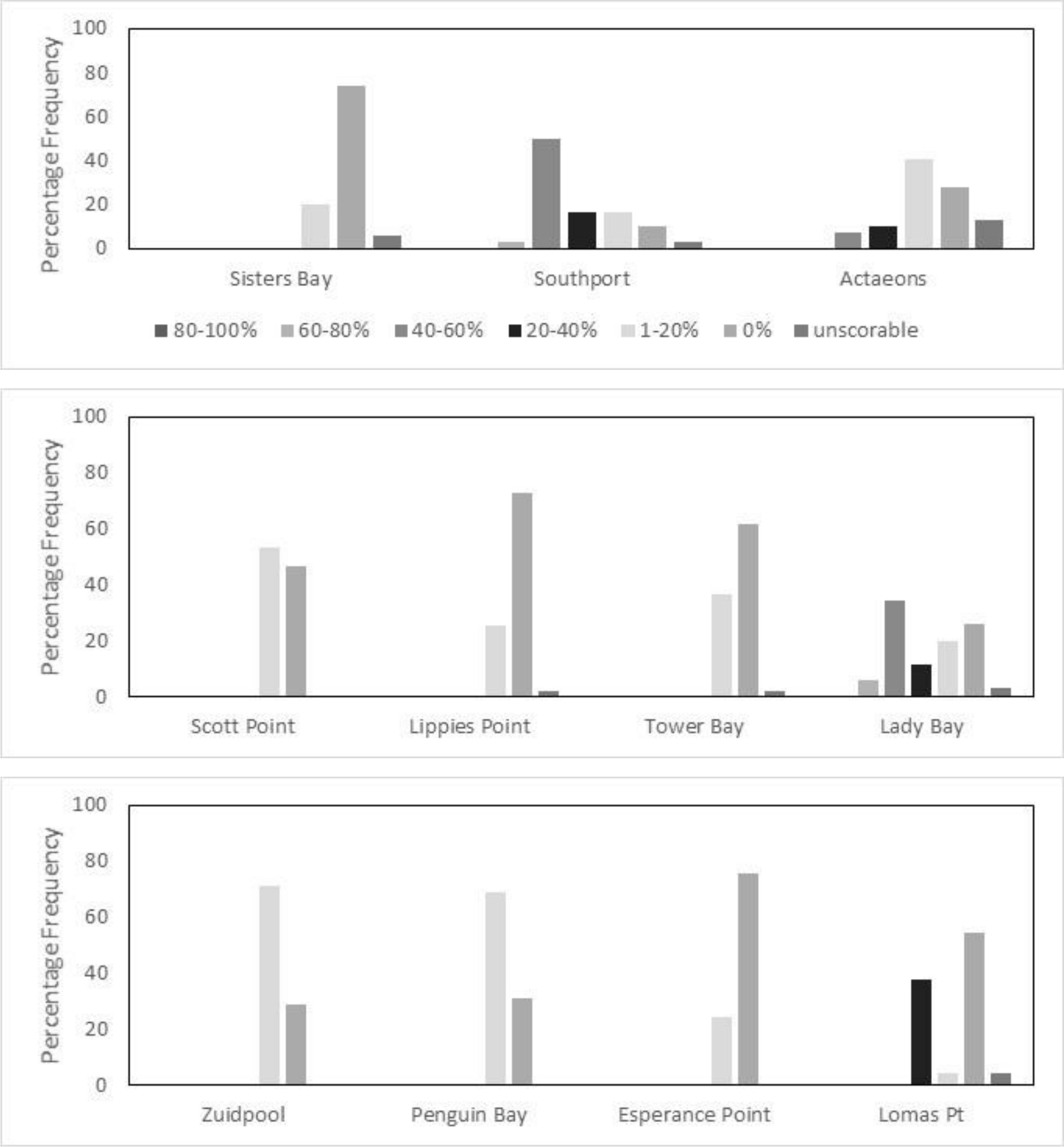

Figure 6.6-8 Percentage frequency of epiphytic algae growth as determined using towed camera in the southern D'Entrecasteaux Channel area.

The spatial analysis supported the categorical data, with Lady Bay and Southport having highest levels of epiphyte growth within the survey area (Figure 6.6-9). At Lady Bay, levels of epiphytic algae ranged from 1-20% to 60-80% and to 40-60% at Southport. The northern sites of Zuidpool and Penguin Bay also showed areas of consistent low epiphytic algal growth, with contiguous sections of video transect recording values of 1-20% (Figure 6.6-9).

A gradient in epiphytic algae was observed at Lomas Point with the westernmost section of the video transect showing greater levels of epiphytic algae (20-40%) compared to the sections of the transect further to the east. Similarly, at Esperance Point the northernmost section of the video transect showed the presence of epiphytic algae (1-20%) (Figure 6.6-9). The sections of these two transects that recorded the presence of epiphytic algae growth were situated in closer proximity to potential nutrient enrichment sources. Such sources were identified as river inputs (Esperance River, Huon River respectively) and anthropogenic sources such as salmon farming activities at Lomas Point and Esperance Point. The higher levels of epiphytic algae observed at Southport are harder to explain, given our spatial coverage was limited to a small survey area on the point. Any repeat surveys in this area should look to extent spatial coverage further into Southport and around the coastline to provide greater context to this result.

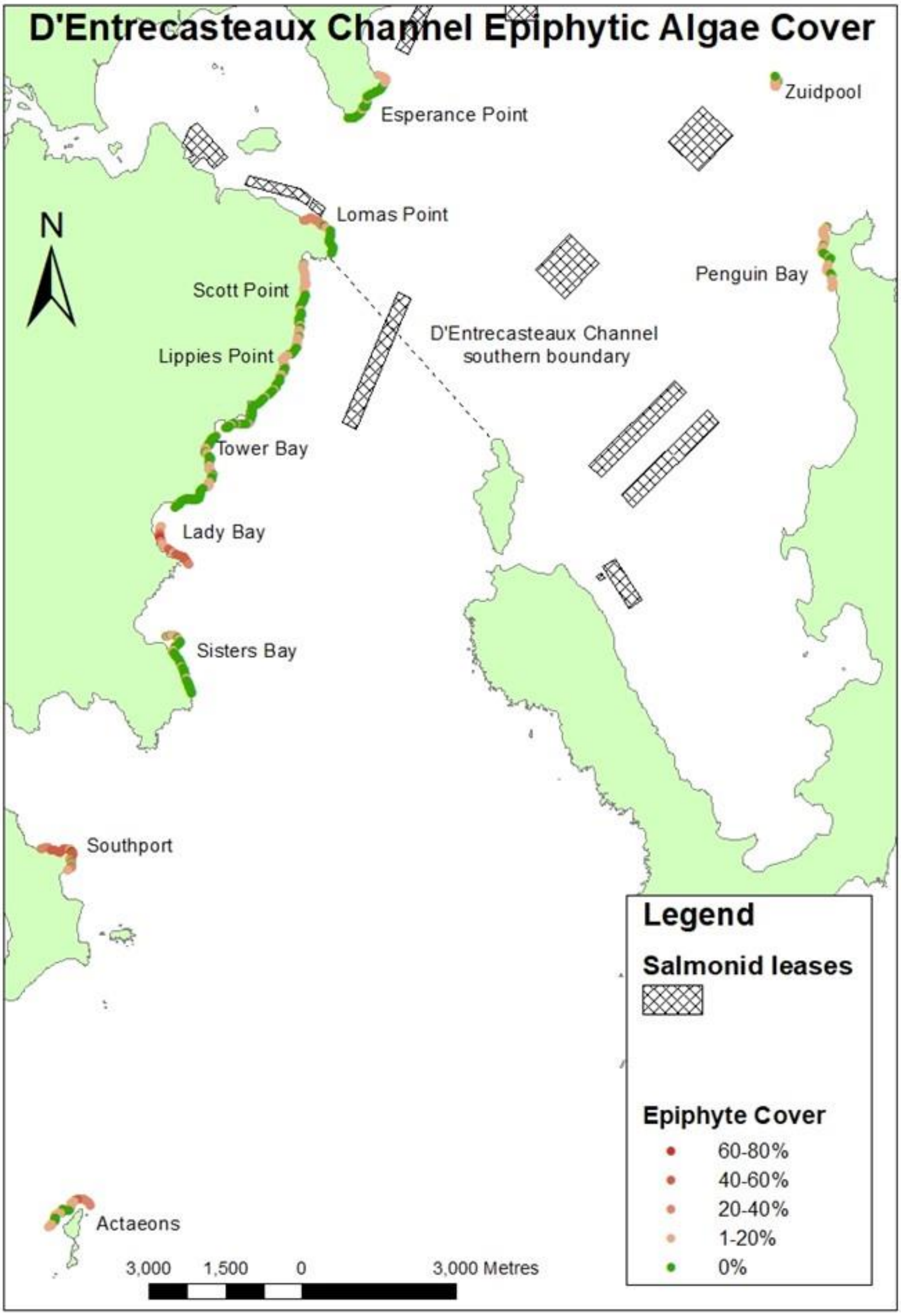

Figure 6.6-9 Epiphytic algae cover percentage as determined using towed camera in the southern D'Entrecasteaux Channel area in February 2018. Areas marked as red show increased levels of epiphytic algal growth, while those shown as green have no epiphytic algal growth.

6.6.3.4 Evaluation of canopy condition

The northern site of Penguin Bay had the highest patchiness score (11.30) of all sites assessed. The other northern sites of Zuidpool (7.34), Lomas Point (5.80), and the southernmost sites of Southport (8.84) and Actaeons (7.83) all recorded moderate patchiness for canopy algae cover using this measure (Table 6.6-3). Scott Point, Lippies Point, Tower Bay and Sisters Bay all had very low scores for patchiness (<3.00) (Table 6.6-3).

Table 6.6-3 Canopy cover patchiness relative to transect length. Patchiness score is calculated as a function of the number of patches and transect distance (km). Those sites with a higher patchiness score have greater variation in canopy cover levels.

Site	Number of Patches	Transect Length (km)	Patchiness Score
Zuidpool	4	0.55	7.34
Penguin Bay	17	1.51	11.30
Esperance Point	5	1.49	3.36
Lomas Point	7	1.21	5.80
Scott Point	5	2.06	2.42
Lippies Point	7	2.51	2.79
Tower Bay	3	2.28	1.32
Lady Bay	9	1.83	4.93
Sisters Bay	3	2.01	1.49
Southport	11	1.24	8.84
Actaeons	12	1.53	7.83

When the canopy cover of each patch was examined, Esperance Point, Scott Point, Lippies Point, Tower Bay and Sisters Bay all had a high average percentage patch length of resilient canopy cover (60-100% cover; Figure 6.6-10). In contrast, Penguin Bay, which also had the highest patchiness score (11.30 patchiness) also had the shortest average percentage patch length of all sites, with patches being a combination of both high and low canopy covers (Figure 6.6-10). The southernmost sites of Southport and Actaeons also had relatively short average percentage patch lengths, however had longer average percentage patches of resilient canopy (Figure 6.6-10). The northernmost site of Zuidpool (7.34 patchiness score) had a greater average percentage patch length of the intermediate group (20-60% cover) than any of the other surveyed sites. Patchiness data appear to highlight a moderate exposure gradient, in that those sites in the middle section, i.e., the most exposed sites (Esperance Point to Sisters Bay, see Figure 6.6-10), tended to have lower patchiness values when compared to the more sheltered northern sites.

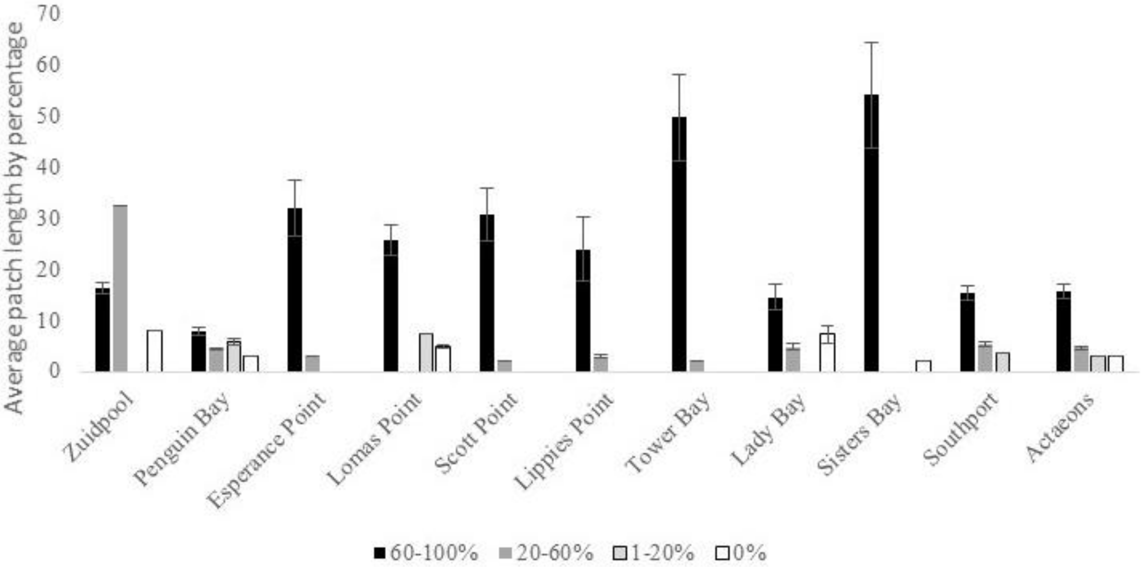

Figure 6.6-10 Canopy algae patchiness grouped into resilient (60-100%), patchy (20-60%), low (1-20%) and no canopy categories ±SE for the southern D'Entrecasteaux Channel.

6.6.4. Discussion

This study showed it is possible to use the remote techniques of towed camera and photo quadrat analysis to assess canopy cover and composition over a broadscale efficiently and comprehensively. Both canopy cover and composition revealed a low level of uncertainty at all photo quadrat sites and at all towed camera sites except for the Actaeons. While this result is unsurprising given previous studies (Juanes et al. 2008; Guinda et al. 2014a; Guinda et al. 2014b; Mallet & Pelletier 2014), this work provides a quantitative and robust assessment at varying degrees of exposure to environmental influences. While towed camera can capture a broader range of functional categories, this happens at the expense of accuracy. Towed camera data indicated broad categories of cover, while it was possible to obtain an actual percentage cover value using photo quadrats.

The towed camera captured footage from an oblique angle, meaning greater sections of the algae that are below the uppermost sections of the canopy could be viewed (Holmes et al. 2013; Mallet & Pelletier 2014). By comparison, photo quadrats were taken from above, and because the sampling was non-invasive or destructive, the canopy algae obscured what was directly below, thus increasing uncertainty in the assessment of sub-canopy biota (Glasby 1999; Ling 2008; Dunmore et al. 2014; Oh et al. 2015). This study found it is simply not possible to consistently assess any biota below algal canopy level using either method (Harasti et al. 2018). On several occasions, the biota beneath the canopy could be confidently assessed: when canopy algae were absent, when water movement pushed canopy algae to its side or when bare rock was the prevalent habitat. However, these occurrences were incidental and could not be relied upon to assess the sub-canopy habitat if either method was to be utilised in the future.

The spatial trends in canopy algae composition determined in this study through both towed camera and photo quadrat assessments were supported by previous works relating exposure to characteristic algae species. For example, *P. comosa* dominated sites with a higher level of exposure, while *E. radiata* occurrence increased relative to shelter (Edgar 1984). This was evident at the Penguin Bay and Zuidpool sites, the westernmost section of the Lomas Point and the northwesternmost section of the Esperance Point towed camera survey transects where the observed algae species makeup was either partly or wholly dominated by *E. radiata*. Whereas the more exposed sites of Scott Point, Lippies Point, Tower Bay, Lady Bay, Sisters Bay, Southport and some parts of the Actaeons site were dominated by either *P. comosa* with *E. radiata* sub-dominant or absent. This capacity to detect changes to canopy composition highlights the robust nature of both towed camera and photo quadrats for characterising canopy-related categories across a range of sites in differing environmental conditions.

Towed camera operation was best suited to areas where canopy cover was relatively prostrate, which was most evident where *P. comosa* and *E. radiata* dominated the canopy (Carroll et al. 2018). These algae species tend not to exceed 3m above the seabed, forming beds of relatively uniform height (Edgar 2008). Hence, at sites where these two algae species were the dominant canopy forming algae, unscorable towed camera footage was absent or minimal for canopy associated parameters. For instance, the 4.5 km transect recorded at Scott Point and Lippies Point had a dominant canopy species of *P. comosa* (100% and 94.9% occurrence, respectively) and no unscorable sections of canopy cover. Moreover, Zuidpool recorded 88% occurrence of *E. radiata* and no unscorable footage. Conversely, this also highlights the difficulties in assessing sites where canopy cover was variable, or extended to the surface, as was the case with *M. pyrifera*, which was dominant around the Actaeons. This site recorded the greatest occurrences of unscorable footage which was entirely due to entanglement with *M. pyrifera*. Large floating algal canopies on the surface made towed camera operations particularly challenging and often impossible at this site, which was also evident in the Barrett et al. (2010) study. Assessment of sites dominated by *M. pyrifera* was also problematic using photo quadrats, as the technique was unable to assess canopy that occurs on the water's surface. This was evident given the differences between the canopy cover using towed camera versus photo quadrats at the Actaeons site. The towed camera recorded canopy cover at the RVA site to be between 40-80%, whereas photo quadrat analysis recorded canopy cover of only 11.2 + 6.9%, an inconsistency that highlights a major drawback if relying on photo quadrats alone.

Previous studies into organic enrichment of rocky reefs have found in general, the greatest changes in reef habitat have occurred in areas adjacent to increased human population density or activity (Benedetti-Cecchi et al. 2001; Howarth et al. 2002; Arevalo et al. 2007; Connell et al. 2008; Oh et al. 2015). Algal canopy cover is critical in maintaining the ecological services provided by rocky reefs (Bokn et al. 2003; Oh et al. 2015). In this study it was possible to use the towed camera to detect changes in canopy structure over a large spatial area. These changes were quantified by the development of a patchiness scoring system, whereby the relative patchiness of a transect was determined by the number of changes in habitat structure over a large spatial scale.

All sites recorded the presence of epiphytic algae to some extent, and there were gradients of more dense epiphytic algae on the section of transects closest to the areas of human influence and potential nutrient exposure. Oh et al. (2015) and Juanes et al. (2008) correlated such occurrences to be characteristic of potential impacts from nutrient input, particularly from anthropogenic sources. Whilst there was a regional trend observed for epiphytic algae to decrease from north to south within the study area there are two cases that were somewhat anomalous given the categories we surveyed. The occurrence of epiphytic algae (up to 80% coverage) at the Lady Bay site appeared in isolation, and further south, Southport recorded consistent epiphyte cover throughout much of the towed camera transect. As both sites were away from obvious anthropogenic sources of organic enrichment, this highlights the risk of relying too much on one parameter collected from one survey to evaluate enrichment status at any given site.

Assessing rocky reefs on a meso-, or regional-scale is advantageous as broadscale changes may go unnoticed when change is assessed solely through diver surveys. Given the value of spatial scope for assessing algal canopy characteristics, a remote survey technique that allows coverage of 12 km per day, such as towed camera, is highly advantageous (Harriott 1999; Morrison & Carbines 2006; Leujak & Ormond 2007; Shortis et al. 2007; Juanes et al. 2008; Van Rein et al. 2009; Mallet & Pelletier 2014; Walsh et al. 2017). Previous studies have used towed camera technology to survey underwater habitats due to the greater spatial coverage and personnel cost efficiency, despite the loss of fine scale detail (Assis et al. 2008; Nichol et al. 2009; Van Rein et al. 2009; Barrett et al. 2010; Sheehan et al. 2010; Mallet & Pelletier 2014; Reshitnyk et al. 2014; Filbee-Dexter & Scheibling 2016; Logan et al. 2017). The ability to assess more than 12 km of rocky reef in a day was of great benefit for undertaking a broadscale assessment of canopy cover. For example, the 4.5 km towed camera transect at Scott Point and Lippies Point was completed in 109 minutes. This coverage is a considerable advantage over diver-centric methods that, given it takes approximately 45 minutes to assess 100m of transect for the Edgar-Barrett biodiversity method, it would take over 30 hours of dive time to assess the same area in situ. While some of this efficiency may be lost in the length of time it takes to process the video post-collection, it is unrealistic to expect that scoring of canopy at this spatial scale could ever occur through diver-based methods.

For future studies, technology such as remotely operated vehicles (ROV) for reef survey is worth investigation. While an ROV is expected to cover far less ground than a towed video, it is likely to be faster than diver-based surveys. With increased capacity for control in-water, ROV technology has the capacity to survey reef with much higher accuracy than a towed video (Sward et al. 2019). This may be of particular use at sites where diver surveys are simply not feasible (i.e., deeper reef sites) or where there are additional parameters of interest beside canopy cover (i.e. fish communities).

When assessing rocky reefs, particularly canopy forming macroalgae, a regional-scale assessment has a greater propensity to detect potentially long-lasting changes in habitat structure (Juanes et al. 2008; Carnell & Keough 2014; Guinda et al. 2014b). Early detection of such changes could lead to a greater understanding of the cause (Connell et al. 2008), building towards predicting or preventing future loss through environmental management responses if deemed to be the result of anthropogenic activities.

6.6.5. Recommendations and conclusions

In summary, towed video offers a technique that can reliably assess canopy cover of temperate reef ecosystems over a large spatial area. While it may not be recommended for use as a standalone method to assess the functional health of rocky reef ecosystems, this technique has the advantage of scanning large spatial scales relatively quickly. Although more time consuming to collect, the data gained by an in situ assessment is always going to be more thorough because the diver is in the water and able to assess parameters beyond canopy cover, which are important in terms of assessing organic enrichment. Whilst photo quadrats were shown to be highly effective for assessing algae canopy composition, the data collection through this method was limited. Given the breadth of data from both remote methods is somewhat similar, towed camera should be the favoured remote assessment method due to the spatial coverage that can be achieved and the elimination of the dependence on divers. While towed video may not be able to definitively identify organic enrichment, its capacity to assess across a broad area adds an invaluable piece to an overall monitoring program.

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6.7 Fixed artificial structures to determine impact from organic enrichment and sedimentation on nearby temperate reefs

6.7.1 Introduction

Most Tasmanian salmon production occurs in the coastal zone in the south-east of the state, alongside other marine based industries (e.g., tourism, commercial fishing, shellfish aquaculture) and recreational users. As such it is important to understand the potential interactions between the industry, other users and the broader environment to ensure environmental sustainability.

Understanding the ways in which salmon aquaculture interacts with the marine environment is also important to ensure risk appropriate management, and to safeguard broader ecosystem function. Interactions between salmon farms and soft sediment ecology are well understood, with monitoring methods established that can give early warning of adverse impacts (e.g. Crawford et al. 2001; Edgar et al. 2005; Bissett et al. 2007; Thompson et al. 2008; Edgar et al. 2010). However, interactions with other coastal environments, such as rocky reefs are less understood; thus, there is a need to develop methods of monitoring that can be used to support both farm-based and system management of these biologically and economically important ecosystems (Stuart-Smith et al. 2008; Oh et al. 2015; Valentine et al. 2016; Fowles et al. 2018a).

Here the suitability of four potential alternative methods for monitoring rocky reef environments: i) juvenile black-lip abalone abundance, ii) cryptic invertebrate assemblages, iii) epiphyte plates and iv) sediment traps, and the potential to adapt these techniques to better inform salmon farm management was investigated.

Juvenile abalone abundance, cryptic invertebrate assemblages and epiphyte plates were all assessed using artificial substrates deployed in the marine environment. Artificial substrates have previously been used to evaluate different aspects of rocky reef ecology, such as anthropogenic impacts on recruitments processes (Sams & Keough 2007, 2012a, b, 2013a, b). Several field-based monitoring techniques are already employed to assess stock status and recruitment for commercial fisheries such as abalone and rock lobster (e.g. Mundy et al. 2018). It may be possible to use or adapt some of these existing techniques to examine the potential for broader changes in overall community structure or potential impacts on key fisheries species. For example, evaluation of larval communities on existing settlement plates may provide insights not just into recruitment of the target species, but also other species as well. Artificial substrates may be a particularly useful tool to examine impacts of organic enrichment on reef ecosystems, where certain species or differences in the pattern of recruitment may reflect changing background conditions and the resilience of the system. The introduction of substrate space has been shown to provide an opening for opportunistic species to become established (Bokn et al. 2003; Teichberg et al. 2008), with several studies describing those opportunistic species most commonly associated with organic enrichment (Fowles et al. 2018b).

Temperate reef ecosystems have numerous “key species” important for both ecological and economic reasons. For monitoring purposes, key species would be those that adhere to the ‘canary in the coal mine’ principle, whereby changes to these species reflect shifts in the overall ecosystem. For this study the commercially important juvenile black-lip abalone (*Haliotis rubra*) was chosen as a key species, as concerns had been expressed by stakeholders that it might be directly and adversely affected by organic enrichment. We also examined the cryptic species assemblages associated with abalone recruitment structures. Cryptic species on reefs can be affected by sedimentation rates with high biodiversity generally indicating a pristine ecosystem (Smith & Simpson 2002). The variability in the community structure of cryptic species was examined, with a view to identifying how these communities might vary across sites. Epiphytic communities on artificial substrate may also be useful in examining impacts due to organic enrichment. Where free substrate space is available, nutrient enriched waters will support the growth of opportunistic algal species ahead of canopy forming species (Bokn et al. 2003; Teichberg et al. 2008). This provides an opportunity to assess whether the epiphyte recruitment varies spatially between regions and whether specific indicators of nutrient enrichment are present.

In addition, the physical process of sedimentation was measured. Increased sedimentation on reefs due to emissions of particles from salmon farms into the water column is frequently raised as a key concern by stakeholders opposed to expansion of the salmon industry. To address this, reliable data on sedimentation rate and how such sediments might affect key species and ecosystem function is needed. This study investigated if it was possible to characterise the response of sedimentation to nutrient sources and better understand linkages between sedimentation and targeted indicators (Schendel et al. 2004).

These responses are likely to be highly predictable across small spatial scales (i.e. 100s of metres from farms); however, this study sought to determine whether such responses could be detected on a much broader scale. Thus, the aim was to characterise the spatial and temporal variability of these indicators on reefs well beyond the farm leases (i.e. kms from farms/ point source inputs), and to determine if these indicators could be reliably used for ongoing monitoring. To be of value to an ongoing monitoring program, indicators need to be easily and reliably detectable from background variation (Underwood 1994). Consequently, species indicators would need to colonise the artificial substrate in a relatively predictable manner over time and be consistent over a broad spatial scale (i.e. at least at the site level).

The overall aim was to assess these novel approaches for potential incorporation into long-term monitoring programs to evaluate the impact of salmon farming on rocky reefs in Tasmania.

6.7.2 Methods

6.7.2.1 Study area

Monitoring was undertaken in two distinct regions adjacent to Bruny Island in the south-east of Tasmania: North Bruny Island and the South East D'Entrecasteaux Channel (referred to as North Bruny and SE Channel hereafter). The North Bruny region is on the western side of Storm Bay, with three sampling sites located at Cape Queen Elizabeth (CQE), Trumpeter Bay North (TBN) and Bull Bay South (BBS) (Figure 6.7-1). The SE Channel region is a moderately sheltered marine waterway, with the Huon River as the major estuarine input (Butler 2006). There were four sampling locations in this region: Lippies Point (LIP), Sisters Bay (SIS), George III (GIII) and Black Reef Slab (BRS). Sites were characterised for habitat complexity, as this metric is likely to affect ecology and abundance of cryptic species found (Table 6.7-1) and were surveyed three times each year with respect to the targeted indicators.

Figure 6.7-1 Map showing sites for the deployment of artificial substrate. Sediment traps only were deployed at the Actaeons (shown in blue).

Table 6.7-1 Habitat complexity at each of the seven sites

Site	**Habitat complexity**	**General Habitat Description**
North Bruny region		
Bull Bay South (BBS)	High	Medium boulders, cryptic habitat dominates
Trumpeter Bay North (TBN)	Medium	Large boulders and slab dominate, some cryptic habitat observed
Cape Queen Elizabeth (CQE)	Medium	Medium to large boulders, some cryptic habitat observed.

SE Channel region		
Lippies Point (LIP)	Medium	Medium to large boulders, some cryptic habitat observed.
Sisters Bay (SIS)	Medium	Medium to large boulders, some cryptic habitat observed.
George III (GIII)	Medium	Large boulders and slab dominate, some cryptic habitat observed
Black Reef Slab (BRS)	Low	Slab dominates. Very little cryptic habitat observed.

Table 6.7-2 Sampling events for collector plates in both survey areas

North Bruny	SE Channel
December 2016	July/August 2017
April 2017	November 2017
August 2017	March/April 2018
December 2017	

6.7.2.2 Artificial substrate sampling design and collection

Artificial substrates, both abalone/cryptic species and epiphyte plates, were installed on either slab rock or large boulders (>1 m diameter) at 5 - 10 m depth at each of the selected rocky reef sites. Plates were deployed along a line, collector string, as shown in Figure 6.7-2, with two strings of twenty abalone/cryptic species collector plates at each site. Each collector string comprised a 20 - 30 m galvanised main chain deployed by boat with divers floating them into position using subsurface lift bags. Collector plates were positioned relatively evenly across both sides of the main chain depending on appropriate substrate availability and fixed to the substrate by diver, as outlined in McAllister et al. (2019). The collector plates were a 400mm diameter, 12 mm thick HDP disc with stainless-steel spacing pins ensuring they sat approximately 30 – 40 mm above the substrate; this spacing was designed to simulate the habitat of juvenile abalone and other cryptic reef species (Figure 6.7-3). Each plate was numbered 1 – 20 where its associated tether chain joined the main chain.

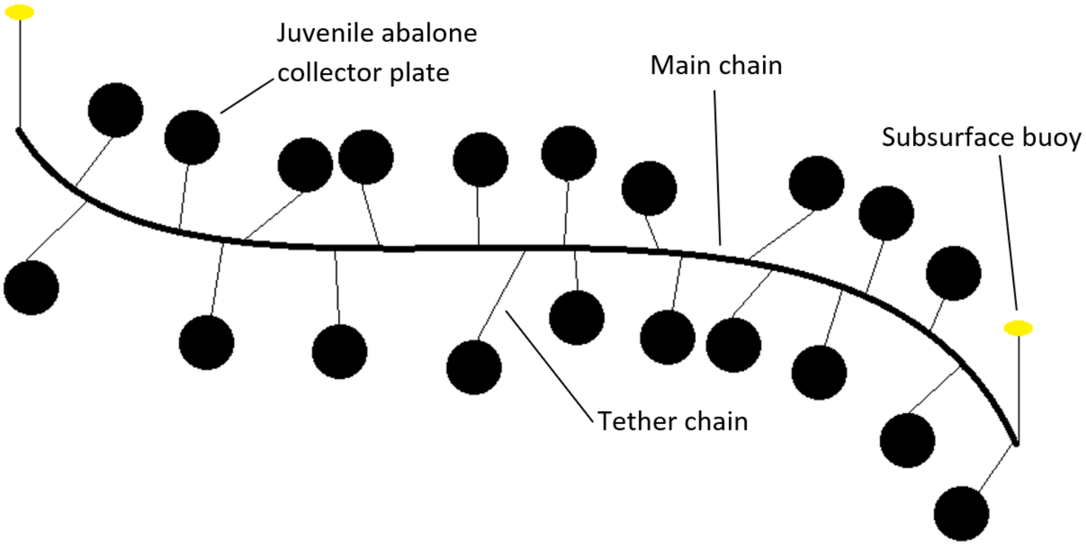

Figure 6.7-2 Diagram outlining the deployment of an abalone/cryptic species collector string showing main chain, tether chains, juvenile abalone collector plates and subsurface buoys.

Figure 6.7-3 Juvenile abalone plate A) attached to substrate, B) removed for assessment of abalone and C) with epiphyte growth plate installed on the top side of an abalone collector plate.

To determine epiphyte growth rates, separate plates were installed to the top side of the first eight abalone/cryptic species collector plates (Figure 6.7-3) on each collector string. These epiphyte plates were made from 3 mm thick clear Perspex sheets cut to a 115 mm x 115 mm square, with the upward facing side roughened to facilitate algal attachment. The first three epiphyte plates on each string were deployed for 12 months, and the next 4-8 plates were changed every three months.

The time between plate installation and initial survey varied between sites. GIII and BRS had plates installed in April – July 2015 as part of another project, with the first survey for this project occurring in July 2017. CQE, TBN and BBS were installed in August 2016, four months before their initial survey in December 2016, while LIP and SIS were installed in May 2017, about two months before they were first surveyed in July 2017.

Under each collector plate, divers using SCUBA counted and measured all black-lip abalone, while all other cryptic species were identified to the lowest readily identifiable taxonomic level and counted. The first three epiphyte growth plates were photographed using a Canon Powershot G16 12MP digital camera and the remaining five epiphyte plates were placed in Ziploc bags, and clean plates re-installed.

Back in the laboratory, the epiphyte plates were placed under water in a shallow tray and photographed. A visual assessment was undertaken, and the main characteristics recorded: algal types (colour, turfing, juvenile species where large enough to identify), encrusting coralline algae presence, and any encrusting species such as polychaete worm tubes or bryozoans. Each epiphyte plate was then scraped clean with all material collected and placed into a pre-weighed vial. These samples were then freeze dried and weighed again to obtain a dry matter weight of all epibiotic growth.

6.7.2.3 Sediment trap design, deployment and analysis

Reef sediment traps were deployed between the 5 and 10 m depth contours to characterise variability in sedimentation both within and between sites. Sites were surveyed twice in each region (January 2017, June 2017: North Bruny; November 2017, January 2018: SE Channel), with sampling times corresponding to peak and low biomass on the salmon leases. In the North Bruny region, sites were selected consistent with the locations of the collector strings (BBS, TBN & CQE). In the SE Channel region, three sites were also identified LIP, SIS and the Actaeons (ACT), with the ACT location providing data for both GIII and BRS (Figure 6.7-1).

In the North Bruny region, sediment traps were deployed on poles (Figure 6.7-4), with two canisters approximately 400 mm (sub-canopy) above the substrate (canopy) and two at 1000mm. Three trap assemblages were deployed at each site, providing six traps at each height for every site (total of 12 traps per site). The sub-canopy effect on sedimentation was only assessed at the North Bruny sites, while traps in the SE Channel only had the canopy traps (i.e. two traps at 1000 mm above the substrate on each pole, total of six traps per site).

Each sediment trap was made from 345 mm long x 55 mm diameter PVC tubing, capped at the bottom and with a baffle fixed to the top to exclude cryptic fish and octopus. The traps were deployed by divers for between 8-12 days.

Figure 6.7-4 Sediment trap installed in reef habitat (Picture: Cayne Layton).

Sediment traps were retrieved by diver, with caps placed on top of each trap in situ to avoid material loss. The contents of the sediment traps were decanted into individually labelled 250 ml jars on the boat and chilled to minimise deterioration. In the lab, sediment from each trap was isolated using a vacuum filter. Pre-dried (4 hours at 450ºC) and pre-weighed Whatman GF/F filters were placed in the vacuum filter and the samples passed through until all excess moisture was removed. Samples were then dried for 48 hrs at 60 ºC and re-weighed to provide an estimate of total suspended solids (TSS). Samples were then placed in the muffle oven for 8 hours at 450ºC to obtain a measure of organic content. All measures were converted to a daily sedimentation rate for statistical analysis.

6.7.2.4 Statistical analysis

Abundance of juvenile abalone was compared between site and survey (using two-way ANOVA, with a crossed design and both factors fixed). For this analysis, the two regions (North Bruny and SE Channel) were considered separately. Data were subject to a square-root transformation prior to analysis to satisfy the assumptions of the model. Where treatments differed significantly ($p \leq 0.05$), Tukey-Kramer post-hoc tests were conducted to detect differences among means. A post-hoc one sample, two tailed power analysis was carried out on the last survey event for sites which consistently recorded abalone counts >0 to estimate the number of abalone plates required to detect 95%, 80%, 50%, 20% and 5% changes in mean juvenile abalone abundance.

Differences in cryptic species counts between sites were compared using the multi-variate analysis (PRIMER v7, Plymouth Routines in Multivariate Research; Clarke et al. 2014). For this analysis, the two regions (North Bruny and SE Channel) were considered separately. The effects of site, survey and replicate string on cryptic species assemblage were investigated using a crossed PERMANOVA design with pairwise testing. Survey event and site were both fixed factors, while string was a random factor nested within site. Results were grouped by string, and a plate average for every species was calculated for each string. A Bray-Curtis dissimilarity index was calculated for each survey region using the dataset grouped by string, and principal coordinates analysis (PCO) and non-parametric multi-dimensional scaling (nMDS) ordinations were undertaken to visualise patterns within the data. Cluster analysis was carried out objectively to identify groupings in the data, and results overlaid on the PCO plots. Vector overlays using a Pearson correlation were used to identify any parameters which might have been influencing trends.

The effect of site and survey event on epiphyte abundance was explored separately for each survey region using a two-way ANOVA with a crossed design, in which site and survey were fixed factors. Epibiotic growth on plates deployed for 12 months and sampled once at the end of the deployment period, was also assessed using one-way ANOVA, with site as a fixed factor. Both data sets were square root transformed prior to analysis. Where epibiota abundance differed significantly ($p \leq 0.05$), Tukey-Kramer post hoc tests were conducted to detect differences among means. Once again, a post-hoc one sample, two tailed power analysis was carried out on the single deployment data set to estimate the number of epibiota plates required to detect 95%, 80%, 50%, 20% and 5% changes in mean epiphyte abundance. For each data set (i.e., both single and multiple deployments) the number of times different taxa were recorded as the dominant cover on plates were noted and expressed as a percentage of the total number of plate deployments.

For the North Bruny region, the effect of site location, survey event and trap position (canopy or subcanopy) on sedimentation rate and percentage organic content was compared using three-way ANOVA, with a crossed design and all factors fixed. In the SE Channel region, trap position was excluded from analysis as only canopy traps were deployed. The effect of site location and survey event on TSS and percentage organic content was compared using a two-way ANOVA, with a crossed design and both factors fixed. Sedimentation rate in both regions was subject to a log10 transformation prior to analysis to satisfy the assumptions of the model. Where treatments differed significantly ($p \leq 0.05$), Tukey-Kramer post-hoc tests were conducted to further assess differences among means.

6.7.3 Results

6.7.3.1 Abalone abundance

In the North Bruny region, there was a significant interaction between site and survey ($F_{(6,458)}=6.8$, p=<0.0001) in abalone abundance. Average abundance differed between all sites but was markedly higher at the CQE and TBN sites (0.8 ± 0.3 and 0.5 ± 0.2 respectively) than at the BBS site 0.04 ± <0.1. In the SE Channel juvenile abalone abundance also changed between sites ($F_{(3,459)}=136.5$, p=<0.0001) but did not differ between surveys ($F_{(2,459)}=0.7$, p=0.4948), with no significant interaction ($F_{(6,459)}=1.1$, p=0.2948; Figure 6.7-5). Whilst BRS and GIII were significantly different to each other, with mean abundances of 4.7±0.2 and 3.2±0.2 abalone per plate respectively, they had very much greater recruitment than the other sites in the SE Channel, where the abundances were much lower, mean 0.2±0.1 (LIP) and 0.4±0.2 (SIS) abalone per plate respectively, and did not differ significantly from each other.

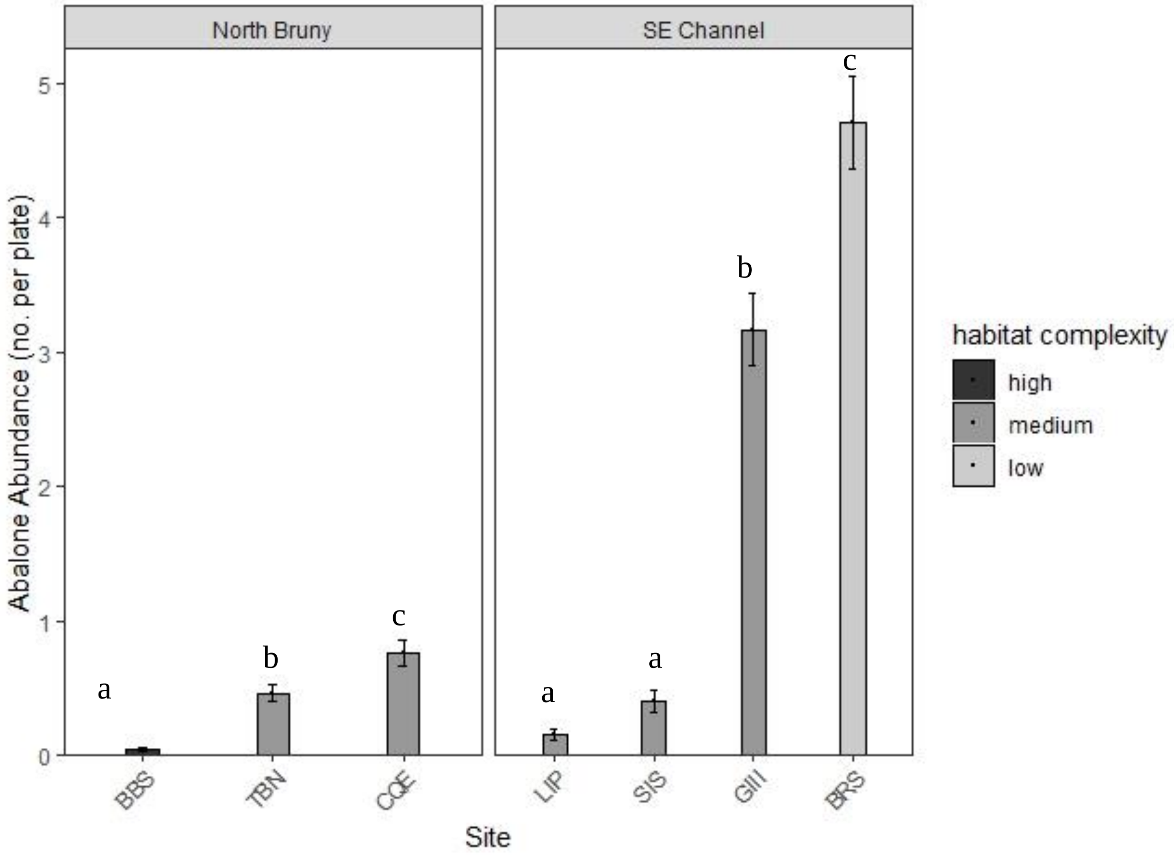

Figure 6.7-5 Total mean abalone abundance ± SE per plate for each study site, with data pooled across survey. Lower-case letters indicate significant differences (P≤ 0.05) between sites based on Tukey's post-hoc testing.

Mean abalone counts were consistently zero (or close to) on the first survey but increased over time. The only exception being BBS which never showed any increase. GIII and BRS had less zero counts than other sites, and although there were differences in the numbers recorded per plate, the frequency distributions were consistent over time. However, it is important to note there was a two-year gap between installation and first survey at these sites (compared with 2-4 months at all other sites), and as such they would appear to display the characteristics of more established populations from the first survey (Figure 6.7-6).

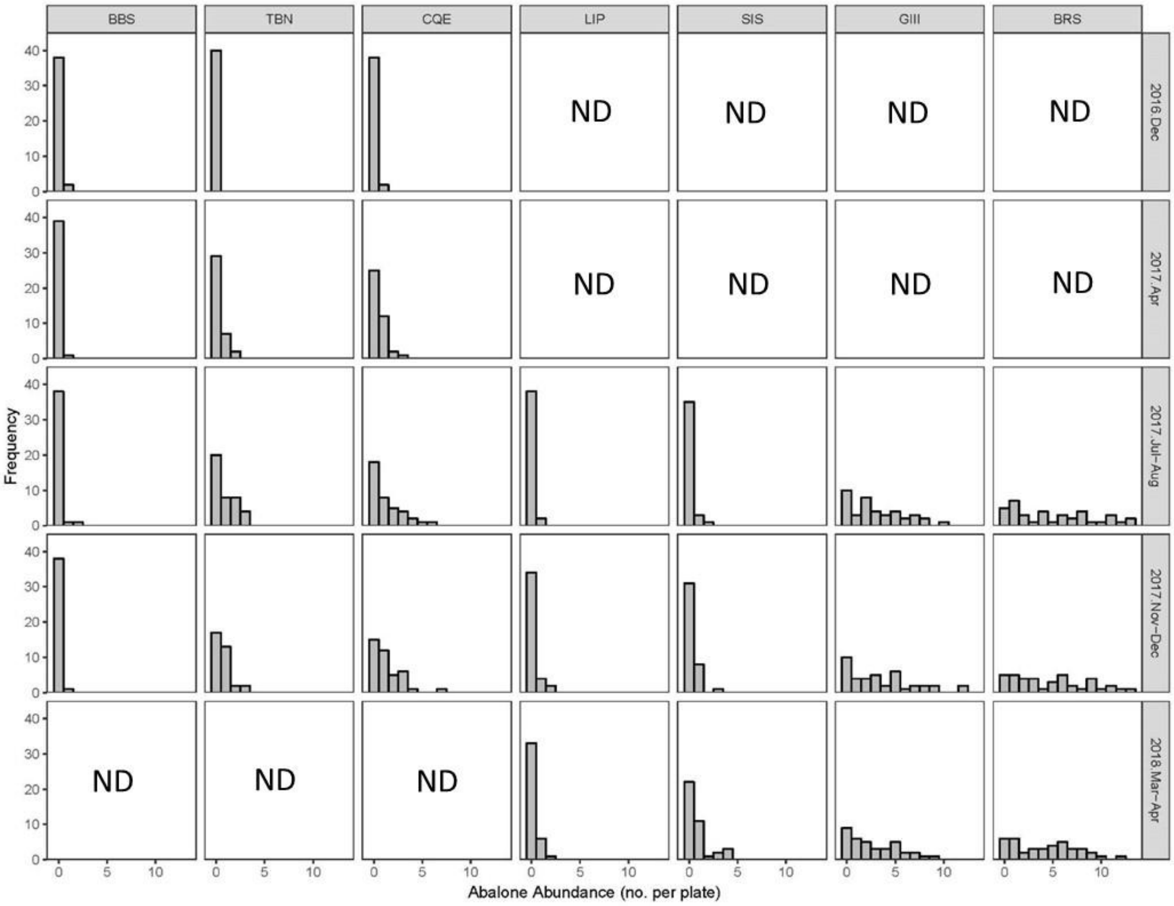

Figure 6.7-6 Abundancy frequency of abalone recorded on collector plates for each survey event across separated by study site. ND: Indicates no data collected at that site for that survey.

The post-hoc power analysis suggested that the current survey design (n = 40) was able to detect 67%, 151% and 120% change in mean abalone abundance for CQE, BRS and GIII respectively (Table 6.7-3). The power analysis results also suggested that to detect change of 80% at a BRS, 133 abalone plates would need to be deployed. At GIII the number of plates required to detect an 80% change in mean abundance was 85, while at CQE where fewer abalone were recorded, the number of plates required would be 29 (Table 6.7-3).

Table 6.7-3 Estimated number of plates required to detect different percentage changes in mean abalone abundance. Significant level (α) = 0.05, power (ϐ) = 0.80 for all sites.

Site	Number of plates (n)	Sample effect size (δ)	Number of plates (n) needed to detect change:				
			δ = 95%	δ = 80%	δ = 50%	δ = 20%	δ = 5%
CQE	40	0.67	21	29	71	432	6879
BRS	39	1.51	95	133	338	2100	33573
GIII	37	1.20	61	85	214	1328	21225

6.7.3.2 Cryptic assemblages

In the North Bruny region, cryptic species assemblages varied significantly depending on survey event ($F_{(3,430)}$= 4.194, p (perm)=<0.05) and string (nested in site; $F_{(3,430)}$=8.157, p (perm)=<0.05). Site itself did not affect species assemblage ($F_{(2,430)}$= 1.744, p (perm)=0.23; Table 6.7-4). However, in the SE Channel, survey event ($F_{(2,447)}$=3.635, p (perm)=<0.05), string (nested in site; $F_{(4,447)}$=5.731, p (perm)=<0.05) and site ($F_{(3,447)}$=4.818, p (perm)=<0.05) all affected the cryptic species assemblage (Table 6.7-5). The significance of string (nested in site) in both regions indicated that the cryptic assemblages varied at relatively small spatial scales.

Table 6.7-4 PERMANOVA results for North Bruny cryptic species assemblages

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Survey event	3	1.13E+05	37702	4.1943	0.0005	9913
Site	2	79147	39573	1.7442	0.2349	90
String (Site)	3	68123	22708	8.1576	0.0001	9894
Survey event x Site	6	59414	9902.4	1.1005	0.3434	9887
Survey event x String (Site)	9	81055	9006.1	3.2354	0.0001	9819
Residuals	430	1.20E+06	2783.6			
Total	453	1.60E+06				

Table 6.7-5 PERMANOVA results for SE Channel cryptic species assemblages

Source	df	SS	MS	Pseudo-F	P(perm)	Unique perms
Survey event	2	61296	30648	3.6353	0.0064	9930
Site	3	1.87E+05	62271	4.8185	0.0287	2451
String (Site)	4	51696	12924	5.7317	0.0001	9880
Survey event x Site	6	66523	11087	1.3149	0.1614	9881
Survey event x String (Site)	8	67462	8433	3.7399	0.0001	9836
Residuals	447	1.01E+06	2255			
Total	470	1.44E+06				

Examining data by string provided an understanding of the broader influence of both survey and site on cryptic species assemblages. PCO analysis clearly showed a colonisation effect, with clusters highlighting the effect that time of sampling had on species assemblage in both regions, with overall variability between survey events reducing over time (Figure 6.7-7 and Figure 6.7-8). In the North Bruny region, the cryptic assemblages from the first survey (Dec 2016; four months post-installation of the plates) were different to all subsequent surveys (approximately 60% dissimilarity), notably due to the presence of the hairy seaweed crab *Notomithrax ursus* (Figure 6.7-7A). The remaining surveys from this region clustered together suggesting similar assemblages, but with comparatively larger average values for presence of the featherstars, *Cenolia trichoptera* and *Cenolia tasmaniae*, and the brittlestar *Ophiothrix caespitose* (Figure 6.7-7A). There were no clear differences between sites for this region; however, one string from CQE (CQE-2) consistently stood apart from the other samples. This string had unusually high numbers of hermit crabs from the family Diogenidae in all surveys, but particularly the first (Dec 2016) and second (Apr 2017; Figure 6.7-7 A,B).

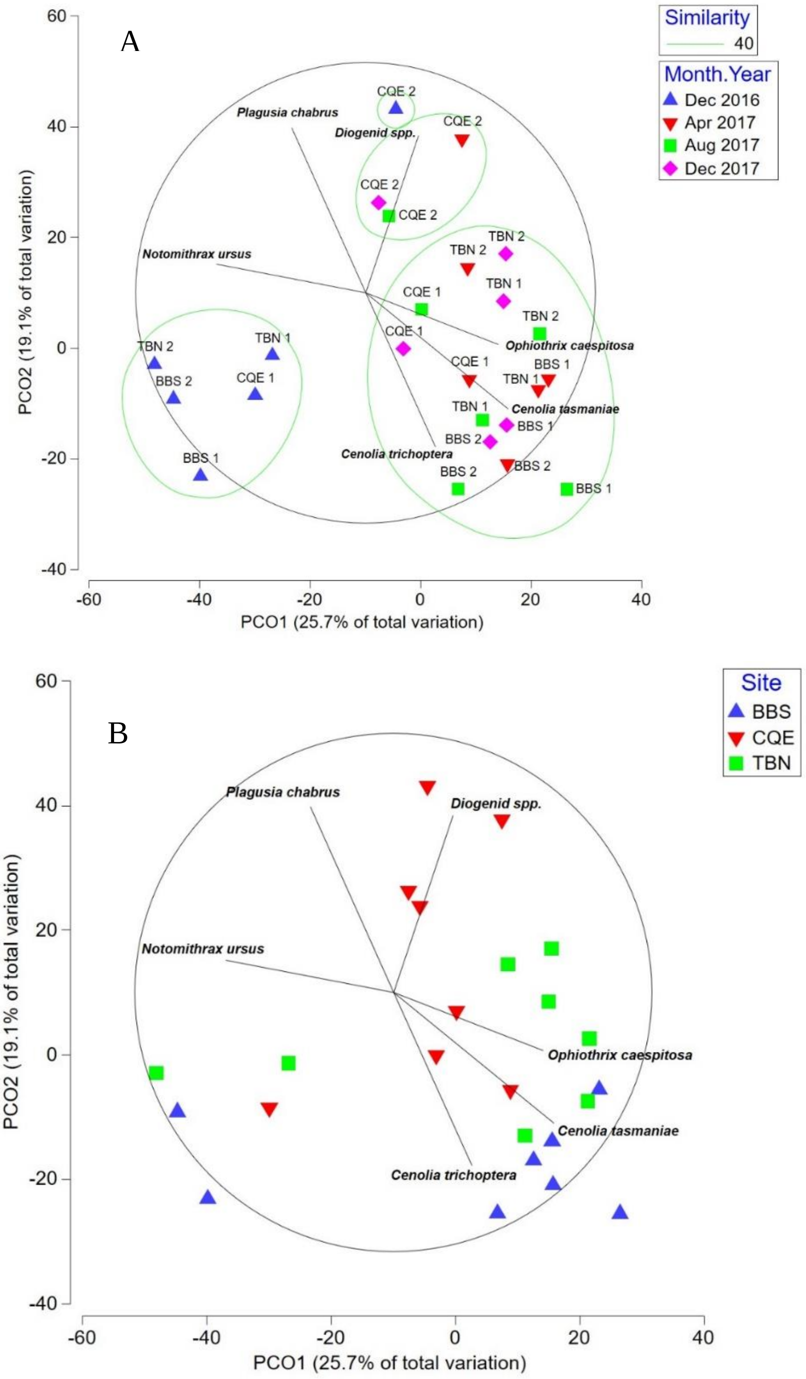

Figure 6.7-7 Principal coordinates analysis (PCO) on cryptic species across all sites and surveys in the North Bruny region presented by A) survey event (numbers on site labels refer to string 1 and 2) with cluster analysis (similarity 40) overlay, and B) site. Vector overlays contributing to principal coordinates with a base variable comparison of >0.6 are included on both A and B.

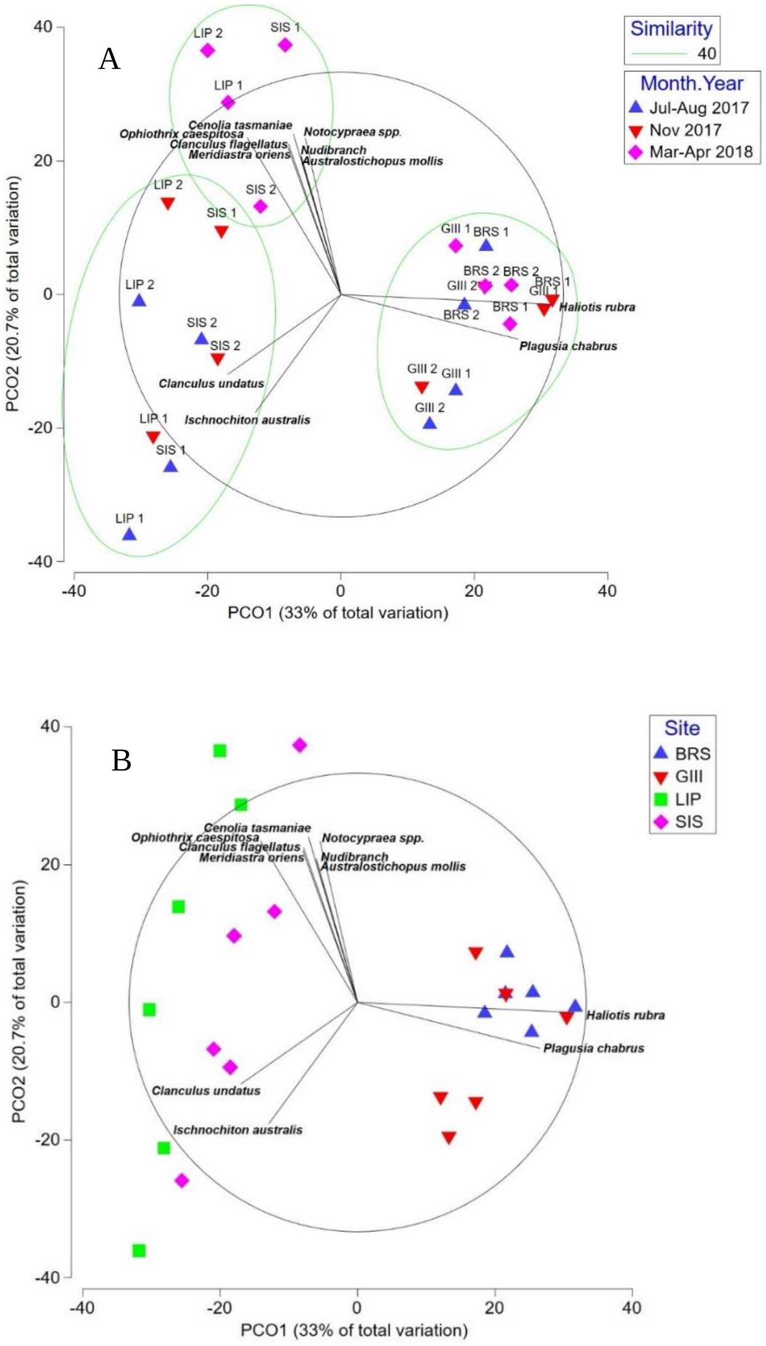

Figure 6.7-8 Principal coordinates analysis (PCO) on cryptic species across all sites and surveys in the SE Channel region presented by A) survey event (numbers on site labels refer to string 1 and 2) with cluster analysis (similarity 40) overlay, and B) site. Vector overlays contributing to principal coordinates with a base variable comparison of >0.6 are included on both A and B.

Differences between the cryptic assemblages at each survey were also evident in the SE Channel region (Figure 6.7-8). At LIP and SIS, the assemblages in the first (July/Aug 2017; 2 months post-installation) and second surveys (Nov 2017; 6 months post-installation) were dominated by the gastropod *Clanculus undatus* and chitons (Figure 6.7-8A). In the third survey (Mar/Apr 2018; 10 months post-installation), featherstars *Cenolia trichoptera* and *Cenolia tasmaniae*, and the brittlestar *Ophiothrix caespitose*, as with the North Bruny surveys were again dominant (Figure 6.7-8A). The cryptic assemblages collected from newly established sites on the first survey (LIP and SIS) were quite dissimilar (approximately 60%) from those collected from established plates (i.e. ones that had been in place for 2 years; GIII and BRS; Figure 6.7-8A and Figure 6.7-8B). The sites that had been installed for 2 years prior to the first survey (GIII and BRS) showed little change over the 10 month survey period (Figure 6.7-8A), with cryptic communities characterised by the high numbers of juvenile black-lip abalone, *H. rubra*, and speedy crabs, *Plagusia chabrus* (Figure 6.7-8B).

6.7.3.3 Epibiota - biomass and community assemblages

The three plates deployed for the full twelve months showed that epibiotic biomass varied significantly between sites in the North Bruny region ($F_{(2,14)}$=11.7, *p*=0.0010; Figure 6.7-9), with CQE recording a higher biomass than either BBS or TBN, but there was no difference in the SE Channel ($F_{(3,20)}$=1.2, *p*=0.3284).

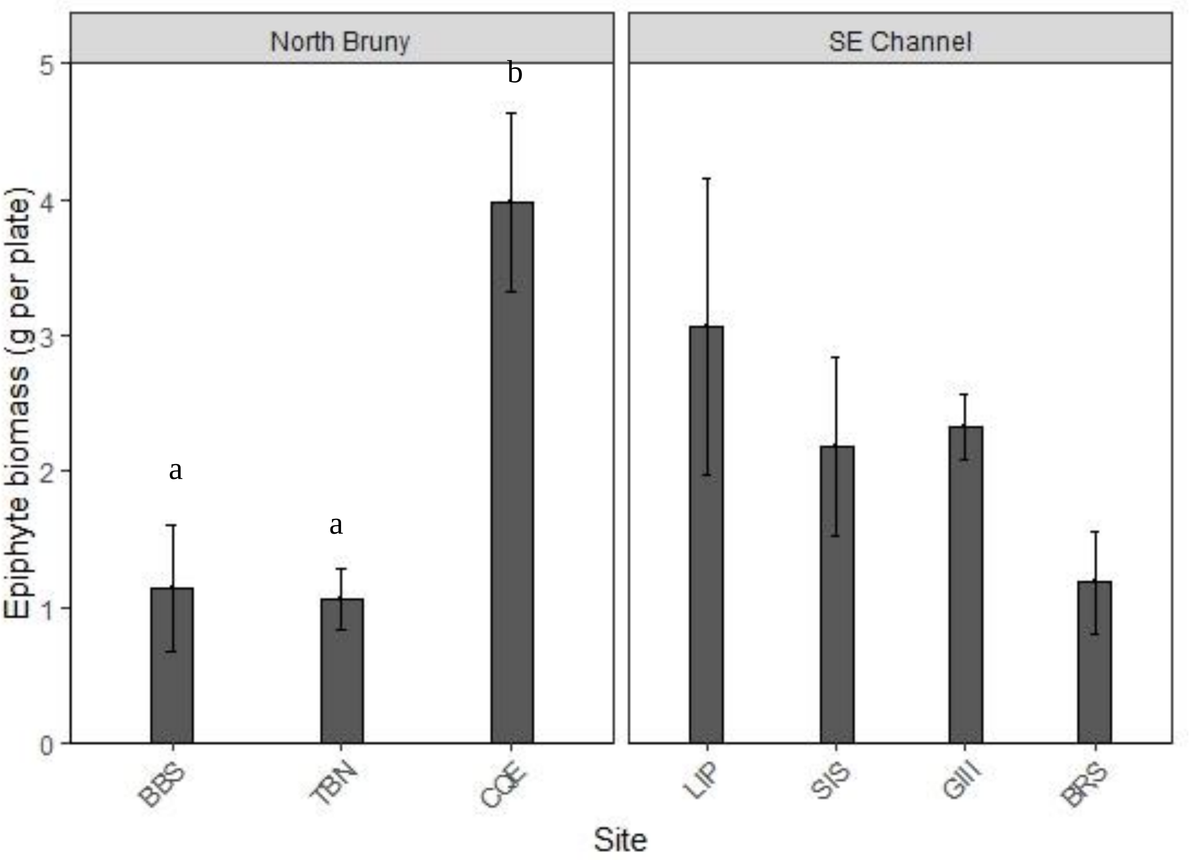

Figure 6.7-9 Mean epiphyte biomass ± SE on plates at seven locations in SE Tasmania. Lower-case letters indicate significant differences (P≤ 0.05) in mean epiphyte biomass between sites based on Tukey's post-hoc testing.

Post-hoc power analysis indicated that the number of plates needed to detect change varied depending on site (Table 6.7-6); only 13 plates were required to detect a 50% change in abundance at GIII, whilst 266 plates would be required to detect the same level of change at

LIP (Table 6.7-6). As a result, the current design (n=6) would be unable to detect a 50% change in abundance at any of the selected sites; the best able to be achieved was the reliable detection (α =0.05, ϐ =0.8) of 95% change at GIII. The number of plates required to reliably detect change increased as the mean weight of the epibiota on the plates increased, consistent with the proportionally higher standard deviation associated with an increase in biomass.

Table 6.7-6 Estimated number of plates required to detect different percentage changes in mean epibiotic biomass for sites deployed for 12 months. Significant level (α) = 0.05, power (ϐ) = 0.80.

Site	Number of plates (n)	Mean weight (g)	Standard deviation	Sample effect size (δ)	Number of plates (n) needed to detect change:				
					δ = 95%	δ = 80%	δ = 50%	δ = 20%	δ = 5%
BBS	6	1.14	1.15	1.65	14	18	43	261	4154
BRS	6	1.18	0.93	1.33	10	13	29	172	2717
CQE	6	3.98	1.61	2.31	25	34	83	511	8140
GIII	6	2.33	0.60	0.86	5	7	13	73	1132
LIP	6	3.07	2.67	3.83	64	89	226	1401	22383
SIS	6	2.18	1.60	2.30	24	33	82	504	8039
TBN	5	1.06	0.51	0.86	4.5	5.4	10	53	819

The plates deployed for 12-months suggested that the epibiota composition was highly site specific within the North Bruny region (Figure 6.7-10). At BBS juvenile brown algae and red algae were the dominant taxa, whereas CQE assemblages were characterised by mixed turfing algae. Encrusting coralline and turfing algae dominated at TBN. Epibiota composition in the SE Channel was similarly site-dependent (Figure 6.7-11), with brown algae dominant at SIS, encrusting coralline dominant at LIP, a mixed brown/encrusting coralline assemblage at GIII and mixed turfing algae and encrusting corallines the most common canopy cover at BRS.

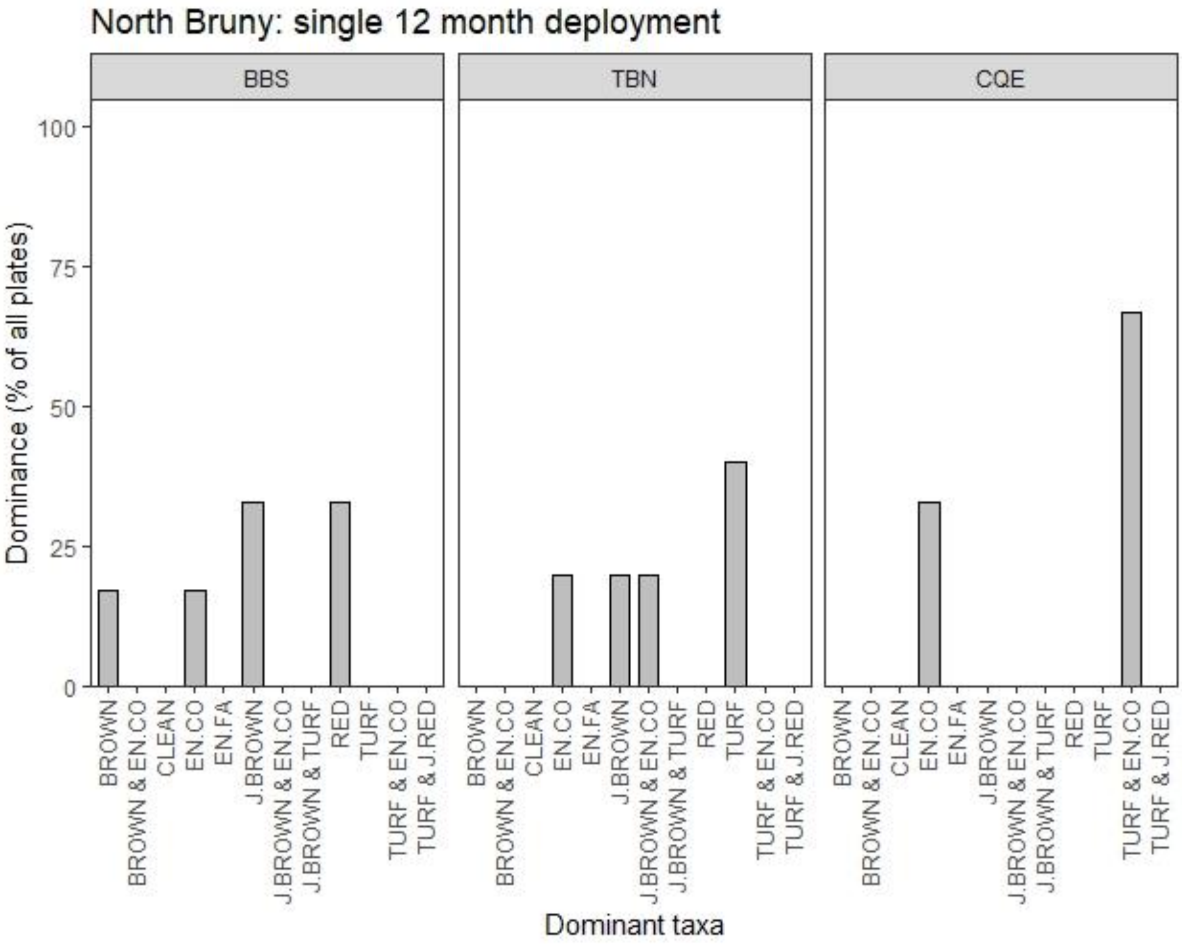

Figure 6.7-10 The frequency of dominance (estimated by percentage cover) for individual taxa identified on plates deployed for 12 months, expressed as a percentage of the total number of deployments for sites on the Nth Bruny region.

Dominant taxa key: BROWN = brown algae; CLEAN = little or no recruitment; EN.CO = encrusting coralline algae; EN.FA = encrusting fauna; J.BROWN = juvenile brown algae; J.RED = juvenile red algae; RED = red algae; TURF = turfing algae. Turfing algae is defined as algae <2cm which do not fit within any of the other categories.

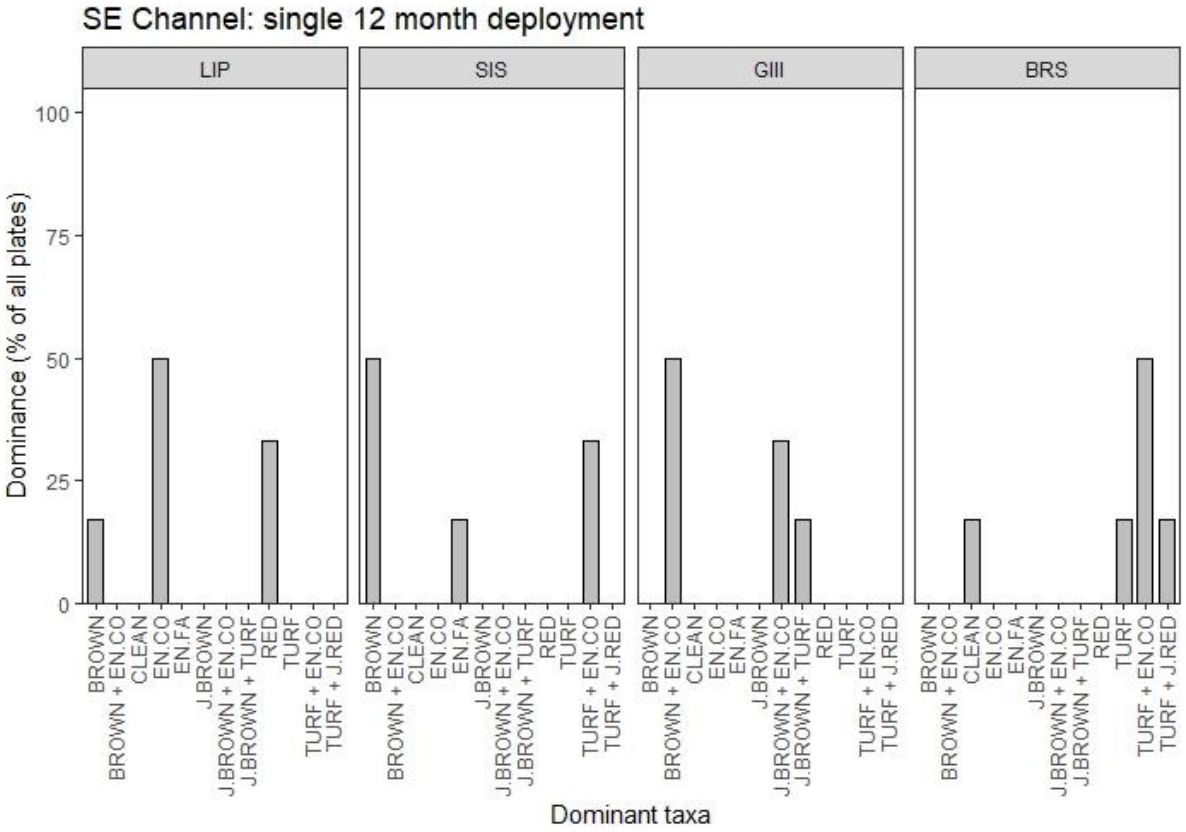

Figure 6.7-11 The frequency of dominance (estimated by percentage cover) for individual taxa identified on plates deployed for 12 months, expressed as a percentage of the total number of deployments for sites in the SE Channel region.

Dominant taxa key: BROWN = brown algae; CLEAN = little or no recruitment; EN.CO = encrusting coralline algae; EN.FA = encrusting fauna; J.BROWN = juvenile brown algae; J.RED = juvenile red algae; RED = red algae; TURF = turfing algae. Turfing algae is defined as algae <2cm which do not fit within any of the other categories.

The data from plates deployed multiple times (i.e. plates removed and redeployed every 3-4 months) showed there was a significant interaction between site and survey ($F_{(4,78)}$=3.26, p=0.0158; Figure 6.7-12) in the North Bruny region. In general, the epibiota biomass at BBS was significantly higher (0.56 ± 0.26 g per plate) than at either CQE (0.19 ± 0.07 g per plate) or TBN (0.23 ± 0.06 g per plate). In the SE Channel, while sites appeared a bit more similar, there was also a significant interaction between site and survey ($F_{(6,108)}$=3.4, p=0.0038; Figure 6.7-12).

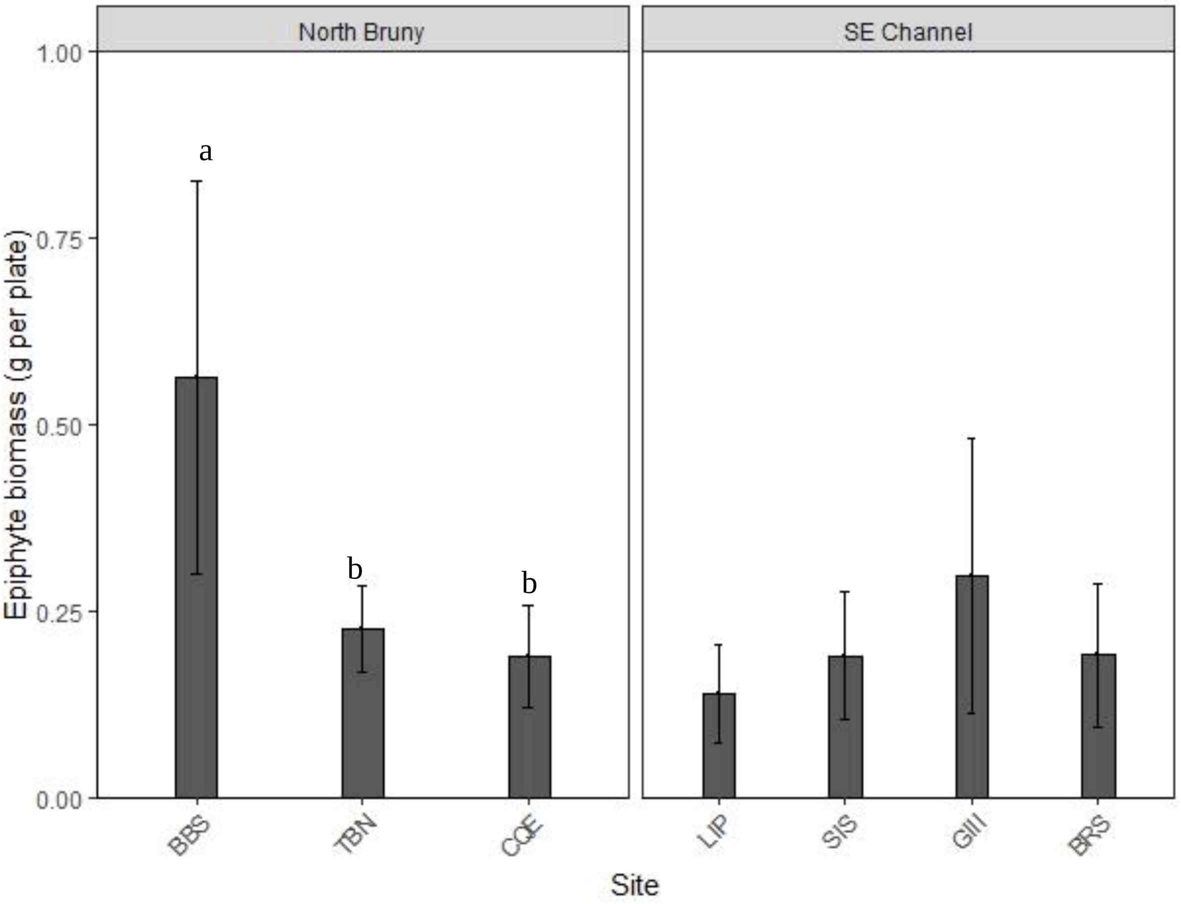

Figure 6.7-12 Mean epiphyte biomass at each site averaged over multiple survey events (n=3). Lower-case letters indicate significant differences ($p \leq 0.05$) in treatment based on Tukey's post-hoc testing.

Sites within the North Bruny region showed a general decline in biomass from March-April 2017 to November-December 2017 (Figure 6.7-13). Within this region BBS recorded the greatest change in epiphyte abundance over time, with a mean of 0.91 ± 0.24 g of biomass per plate in the first sampling event (March-April 2017), decreasing to 0.05 ± 0.02 g mean biomass per plate by November-December of the same year, and very little evidence of recruitment. Conversely, the lowest recruitment in the SE Channel region was at the first survey in July-August 2017, with the biomass increasing through November-December 2017 (Figure 6.7-13). Recruitment levels in March-April 2018 were variable, with LIP and SIS recording biomass similar to that observed in November-December 2017, whilst GIII biomass increased to a regional high of 0.63 ± 0.11 g per plate, and BRS decreased compared to the November-December 2017 peak (Figure 6.7-13).

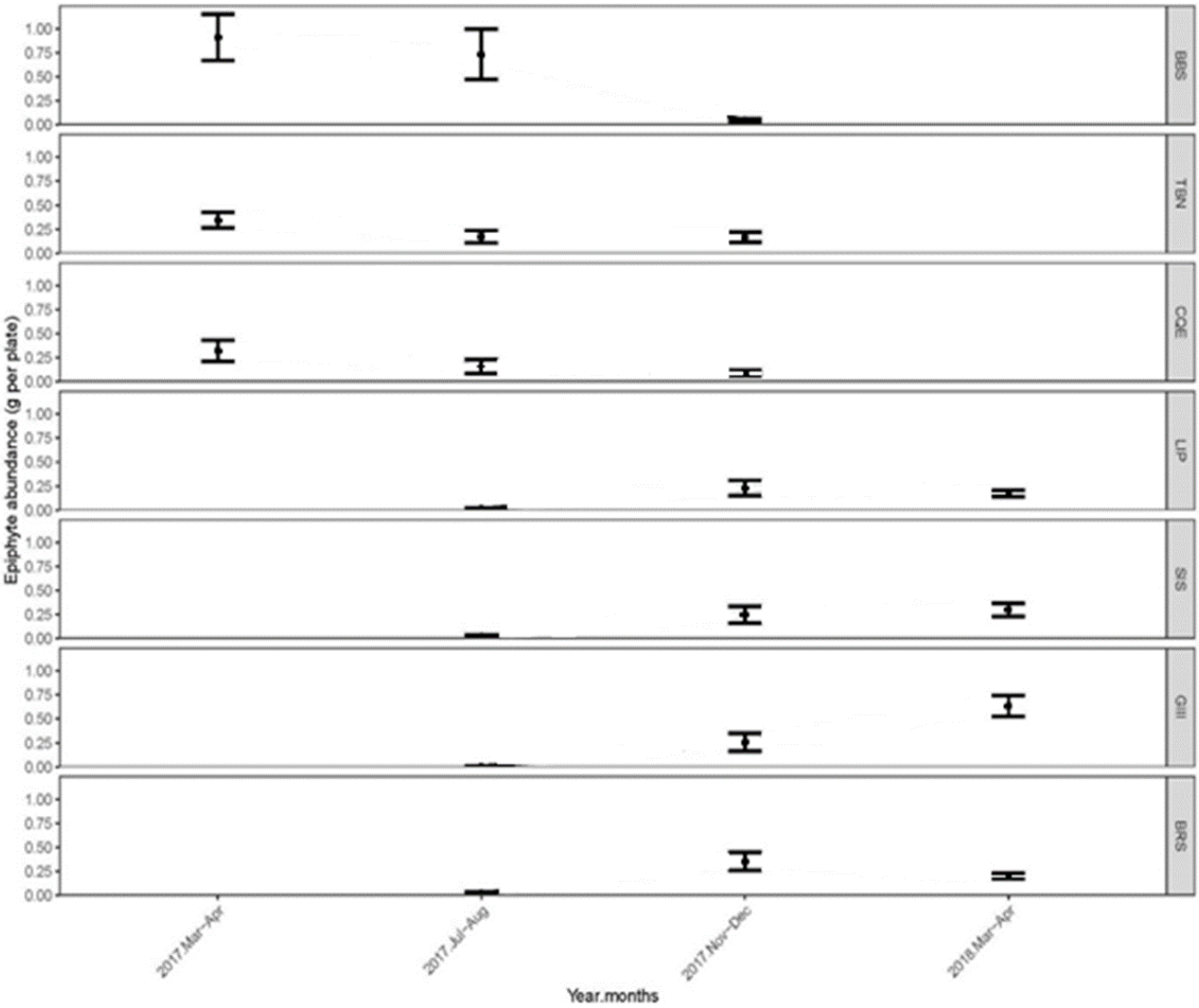

Figure 6.7-13 Mean epiphyte biomass ± SE on collector plates deployed for 3-4 month periods across study sites in both Nth Bruny (top three charts) and the SE Channel (bottom four charts).

The most common epibiotic group observed in the temporal recruitment analyses was consistent across site and time; turfing algae or no recruitment were the most common observations at sites in both the North Bruny (Figure 6.7-14) and SE Channel (Figure 6.7-15) regions respectively. In the North Bruny region, recruitment was lower in July-August 2017 and November-December 2017 than in March-April 2018. In the SE Channel no recruitment occurred more frequently in the July-August 2017 than in November-December 2017 or March-April 2018. The presence and diversity of less common dominant cover categories varied markedly between regions and survey deployments.

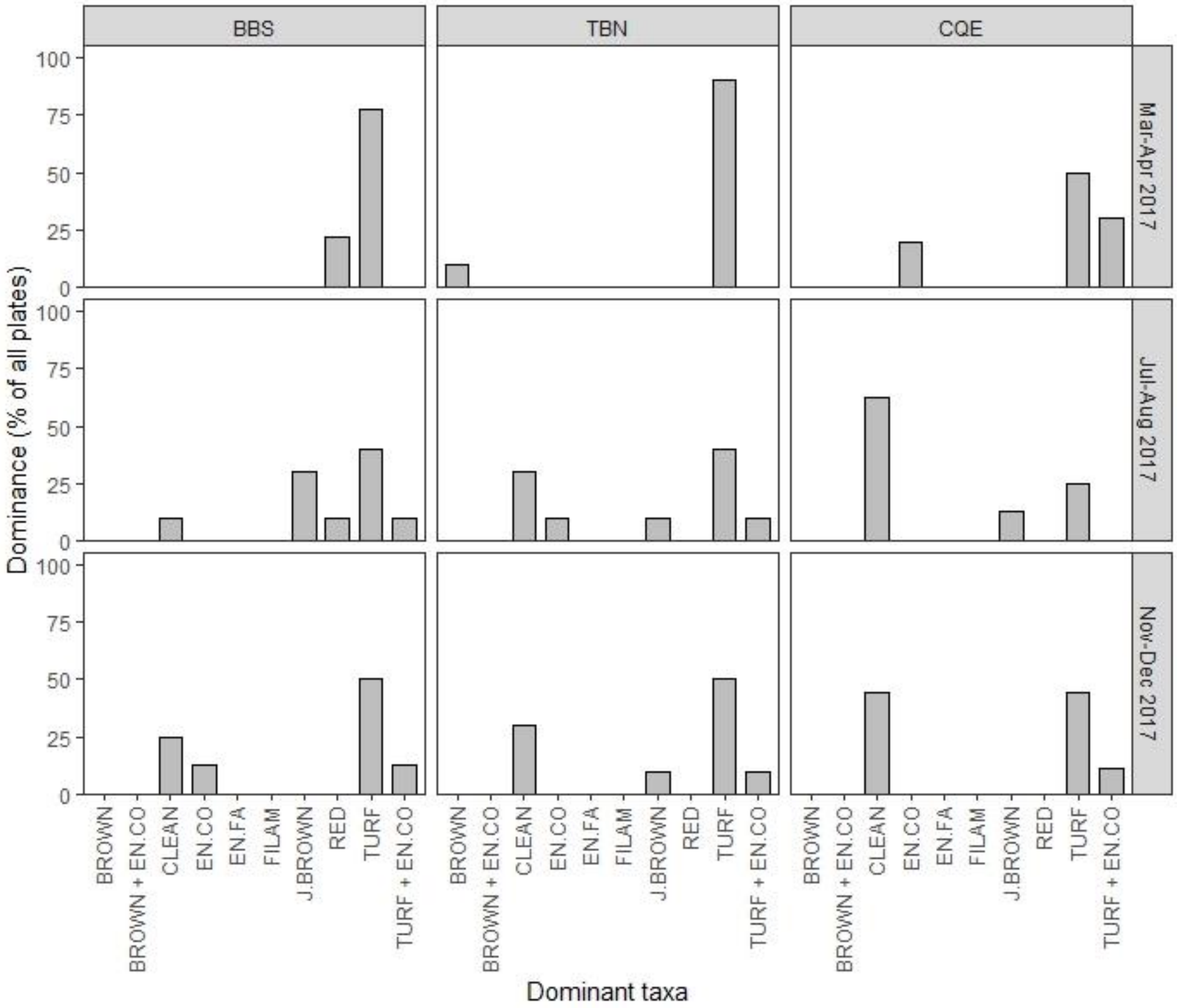

Figure 6.7-14 The frequency of dominance (estimated by percentage cover) for individual taxa identified on plates deployed for 3-4 months, expressed as a percentage of the total number of plates for three sites in North Bruny Island.

Dominant taxa key: BROWN = brown algae; CLEAN = little or no recruitment; EN.CO = encrusting coralline algae; EN.FA = encrusting fauna; FILAM = filamentous algae; J.BROWN = juvenile brown algae; RED = red algae; TURF = turfing algae. Turfing algae is defined as algae <2cm which do not fit within any of the other categories.

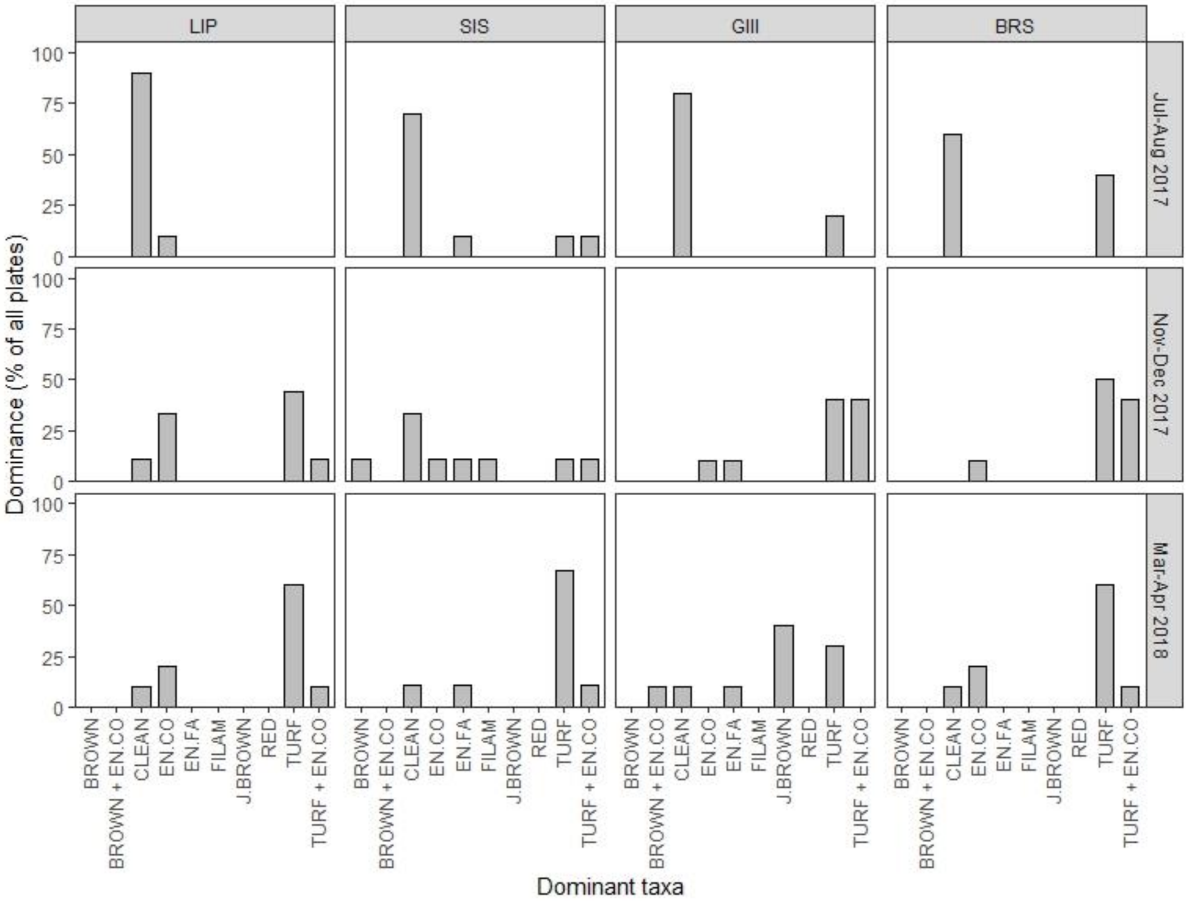

Figure 6.7-15 The frequency of dominance (estimated by percentage cover) for individual taxa identified on plates deployed for 3-4 months, expressed as a percentage of the total number of plates for three locations in the SE Channel region.

Dominant taxa key: BROWN = brown algae; CLEAN = little or no recruitment; EN.CO = encrusting coralline algae; EN.FA = encrusting fauna; FILAM = filamentous algae; J.BROWN = juvenile brown algae; RED = red algae; TURF = turfing algae. Turfing algae is defined as algae <2cm which do not fit within any of the other categories.

6.7.3.4 Assessment of sedimentation

There were significant interactions between site and survey ($F_{(2,21)}$=19.05, p=<0.0001) and survey and position ($F_{(1,21)}$=4.41, p=<0.0480) in the mean sedimentation rate (Table 6.7-7 and Figure 6.7-16). Mean sedimentation rates were consistently lower during Survey 1 (January 2017) than Survey 2 (June 2017) in the North Bruny region (Figure 6.7-16; Table 6.7-8). The highest sedimentation rate was observed in Survey 2 at the CQE subcanopy trap (717 ± 72.8 g/day/m^2), and the lowest in Survey 1 at the BBS canopy trap (6 ± 0.6 g/day/m^2). In general, BBS and TBN recorded lower sedimentation rates than CQE (Table 6.7-8). Across the North Bruny region sedimentation was greater on the subcanopy traps than on canopy traps (Table 6.7-8). This effect was particularly noticeable on Survey 2, with subcanopy traps at BBS and CQE recording approximately double the sediment of traps sitting above the canopy (Table 6.7-8).

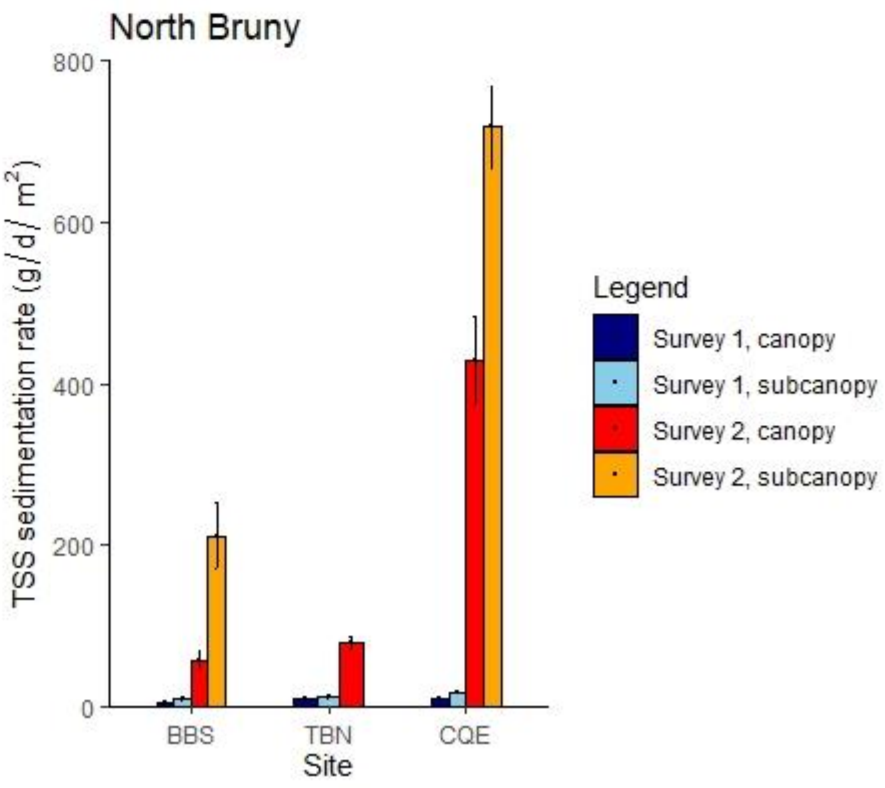

Figure 6.7-16 Sedimentation rate ± SE at three sites in the North Bruny region for survey 1 – January 2017 and survey 2 – June 2017. NB: No sub-canopy traps were deployed at TBN for Survey 2.

Table 6.7-7 Results from the 3-way ANOVA for sedimentation rate across three sites in the North Bruny region (p<0.05).

	Sum of Squares	Df	F value	P value
Site	1.41	2	62.50	<0.0001
Survey	11.57	1	1028.10	<0.0001
Position	0.51	1	45.65	<0.0001
Site : Survey	0.43	2	19.05	<0.0001
Site : Position	0.06	2	2.70	0.0901
Survey : Position	0.05	1	4.41	0.0480
Site : Survey : Position	0.03	1	3.08	0.0937
Residuals	0.24	21		

Table 6.7-8 Mean total suspended solids (n = 3) for sediment traps deployed in the canopy and subcanopy in the North Bruny region.

		Canopy		Subcanopy	
Site	Survey	Mean	SE	Mean	SE
BBS	1	6	0.6	10	1.8
	2	58	11.5	212	40.8
CQE*	1	11	0.5	18	1.7
	2	429	54.7	717	51.5
TBN	1	10	1.7	12	1.4
	2	79	6.9	not sampled	NA

NB: Units for means are g/day/m^2. During the second survey Trap 1 at CQE was not sampled.

In the SE Channel there was also a significant interaction between site and survey ($F_{(2,12)}$=10.24, p=<0.0025; Table 6.7-9). Sedimentation rates were consistently higher during Survey 1 (November 2017) than Survey 2 (January 2018). ACT showed the most extreme difference, with the rates varying between 54 ± 0.5 g/d/m^2 at SIS and 763 ± 173.2 g/day/m^2 in Survey 1 but only between 8 ± 0.3 g/day/m^2 at SIS and 35 ± 2.3 g/day/m^2 in Survey 2 (Figure 6.7-17). Comparison of canopy and subcanopy traps was not undertaken in the SE Channel region.

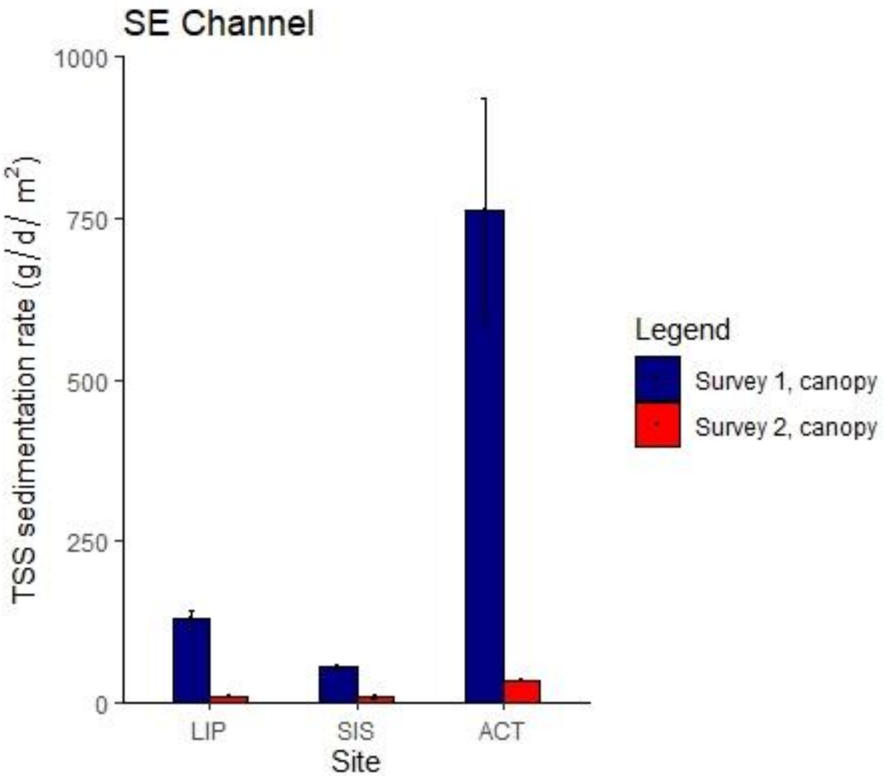

Figure 6.7-17 Sedimentation rate ± SE at three sites in the SE Channel region. Survey 1 – November 2017, survey 2 – January 2018.

Table 6.7-9 Results from the 2-way ANOVA for sedimentation rate across three sites in the SE Channel region

	Sum of Squares	Df	F value	P value
Site	2.55	2	149.639	<0.0001
Survey	5.48	1	644.109	<0.0001
Site : Survey	0.17	2	10.242	0.0025
Residuals	0.10	12		

Table 6.7-10 Mean total suspended solids (n=3) for sediment traps deployed in the canopy in the SE Channel region

		Canopy	
Site	Survey	Mean	Standard error
LIP	1	130	12.8

	2	9	1.2
SIS	1	54	0.5
	2	8	0.3
ACT	1	763	173.2
	2	35	2.3

NB: Units for means are g/day/m^2.

There was a significant interaction between site and survey in the total organic content of the sedimentation material collected from the North Bruny region ($F_{(2,19)}$=11.89, p=0.0004; Table 6.7-11). This interaction was particularly evident in the results from CQE. This site recorded higher mean values for percentage organic content than either of the other sampled locations during Survey 1 (January 2017), and lower mean values in Survey 2 (June 2017) (Figure 6.7-18; Table 6.7-12). In Survey 1 the canopy tarps at CQE recorded mean percentage organic content values of 35 ± 0.1%, compared to 28 ± 2.4% and 28 ± 0.9% for TBN and BBS respectively, whereas for Survey 2 at CQE the equivalent values were 3 ± 0.04% compared to 12 ± 0.4% and 15 ± 1.7% for TBN and BBS.

Table 6.7-11 Results from the 3-way ANOVA for percentage organic content (LOI) of sediments from the three sites in the North Bruny region.

	Sum of Squares	Df	F value	P value
Site	1.5	2	0.05	0.9518
Survey	3197.1	1	218.37	<0.0001
Position	13.0	1	0.89	0.3585
Site : Survey	348.3	2	11.89	0.0004
Site : Position	28.3	2	0.97	0.3987
Survey : Position	6.6	1	0.45	0.5111
Site : Survey : Position	15.4	1	1.05	0.3178
Residuals	278.2	19		

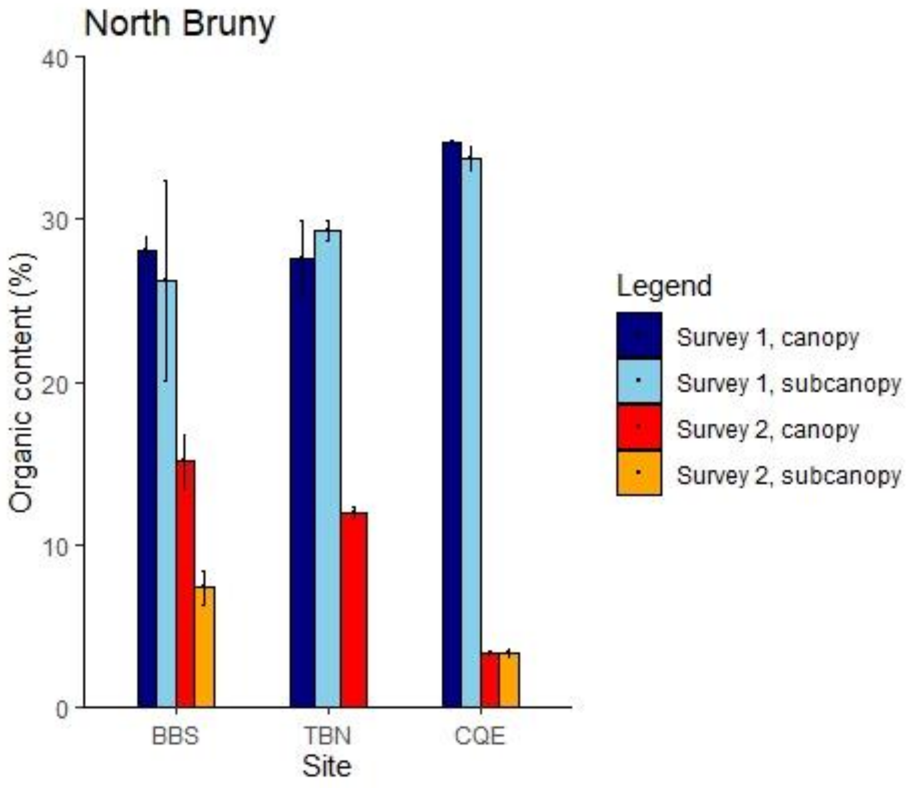

Figure 6.7-18 Total mean organic content (LOI) expressed as a percentage of total suspended solids ± SE for the three sites from North Bruny Island. Note there were no sub-canopy traps deployed at TBN during Survey 2.

Table 6.7-12 Mean organic content (n=3) for sediment traps deployed in the canopy and subcanopy in the North Bruny Island region.

		Canopy		Subcanopy	
Site	Survey	Mean	Standard error	Mean	Standard error
BBS*	1	28	0.9	26	6.1
	2	15	1.7	7	1.1
CQE*	1	35	0.1	34	0.7
	2	3	0.04	3	0.3
TBN*	1	28	2.4	29	0.6
	2	12	0.4	not sampled	NA

NB: Units for means are g/day/m^2. During the second survey Trap 1 at CQE was not sampled.

In contrast, in the SE Channel region there was no significant interaction between site and survey. There was a significant difference between the organic content of sedimented material at each survey event ($F_{(1,12)}$=43.17, p=<0.0001), but this did not differ between sites ($F_{(2,12)}$=0.37, p=0.6993; Figure 6.7-19; Table 6.7-13 and Table 6.7-14). Organic content was consistently higher in Survey 2 (January 2018), with values of 27 ± 1.1%, 27 ± 0.6% and 29 ± 4.0% recorded at LIP, SIS and ACT respectively, compared to 14 ± 0.4%, 18 ± 0.4 and 11 ± 4.4% in Survey 1 (Table 6.7-13, Table 6.7-14 and Figure 6.7-19).

Table 6.7-13 Results from the 2-way ANOVA for percentage organic content (LOI) of sediments from the three sites in the SE Channel region.

	Sum of Squares	Df	F value	P value
Site	13.9	2	0.37	0.6993
Survey	811.0	1	43.17	<0.0001
Site : Survey	61.7	2	1.64	0.2344
Residuals	225.4	12		

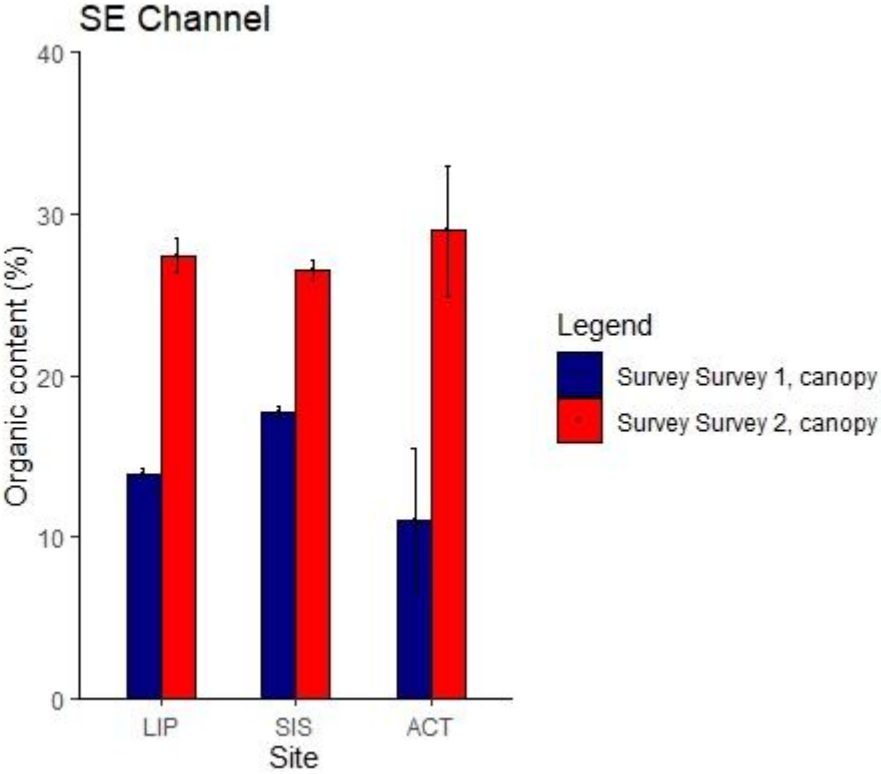

Figure 6.7-19 Total mean organic content (LOI) expressed as a percentage of total suspended solids ± SE at three sites in the SE Channel region.

Table 6.7-14 Mean organic content (n=3) for sediment traps deployed in the canopy in the SE Channel region

		Canopy	
Site	Survey	Mean	Standard error
LIP	1	14	0.4
	2	27	1.1
SIS	1	18	0.4
	2	27	0.6
ACT	1	11	4.4
	2	29	4.0

NB: Units for means are g/day/m^2.

6.7.4 Discussion

This study evaluated novel methods of artificial substrates, indicator species and sediment traps as potential monitoring techniques to better inform salmon farm management. However, the level of spatial and temporal variability in the indicators examined suggests they are unlikely to represent reliable measures of impact from salmon farming on temperate reefs. The inherent spatial and temporal variability that exists in any potential indicator will determine how useful it may be for adoption into a long-term monitoring program (Underwood 1992; Underwood 1994). Cryptic species colonisation showed the most predictable response across sites and time, with little difference between sites after 8-10 months. Abalone recruitment was also relatively predictable, but the sites differed significantly in terms of abundance, indicating strong localised environment or habitat influences (Valentine et al. 2010; Aguirre & McNaught 2012).

6.7.4.1 Assessment of spatial variability in indicators

The best indicators for monitoring programs are those which are sensitive to impact but have low spatial variability (i.e., where the results can be readily reproduced across a range of locations). Understanding the spatial variance associated with an environmental measure or potential indicator can provide some guidance to its suitability as an indicator (Underwood 1994). Unfortunately, all the metrics in the current study displayed spatial variability.

Abundance of juvenile abalone was highly variable both between sites and within sites. Studies of juvenile abalone show high levels of spatial variation in abundance across a range of scales (Mundy et al. 2018). All three sites within the North Bruny region and three of the four sites within the SE Channel were statistically distinct. As such it was not possible to establish control sites or determine which community structure best represented “natural” or unperturbed conditions. When an indicator is highly sensitive to localised environment or habitat influences, it is not possible to distinguish between impact and control sites, and change due to an impact will be difficult to detect (Underwood 1992; Underwood 1994). Juvenile abalone are known to be highly sensitive to variation in habitat, both in terms of site complexity (Aguirre & McNaught 2012; Mundy et al. 2018) and algal assemblage (Valentine et al. 2010), so determining change related to other causal factors would be very difficult. Detecting the impacts of salmon aquaculture on abalone would require a very large number of both impact and control sites to cover all other levels of variation.

There was also high within site variation in abalone abundance, effectively highlighted through power analysis. At sites where juvenile abalone abundance was highest (i.e. Cape Queen Elizabeth, George III and Black Reef Slab), the number of plates needed to detect a 50% change in population density was 71, 214 and 338 respectively, making this approach completely unrealistic as a monitoring tool. Previous studies have shown abalone patchiness and small-scale variability to be a challenge for fisheries management and population assessments (McShane 1998). The sampling effort required to overcome these high levels of both within and between-site variation would be prohibitive for any monitoring program. It is far easier to detect a given impact if the within-site spatial variability is low, as the power of any sampling design will increase for relative effort (Underwood 1994; Underwood & Chapman 2003). While juvenile collector plates have proven useful for stock assessment purposes (Mundy et al. 2018; McAllister et al. 2019), the inherent spatial variability of abalone settlement clearly suggests this approach is not appropriate for broadscale monitoring of aquaculture impact.

Overall, cryptic species assemblages were less variable between-sites than abalone abundance. Differences between sites (i.e. between Lippies Point and Sisters Bay compared to George III and Black Reef Slab) were confounded by the deployment time (i.e. months vs years) making a comparison between sites difficult. Cryptic assemblages at George III and Black Reef Slab were characterised by high numbers of juvenile abalone and speedy crabs (*Plagusia chabrus*), common mobile invertebrate species on Tasmanian temperate reefs (Edgar & Barrett 1997; Alexander et al. 2009; Barrett et al. 2009).

While it is possible George III and Black Reef Slab are inherently different to all other sites, the cryptic assemblage communities at the other sites in the SE Channel and the three sites in the North Bruny region were relatively similar at 8-10 months post-substrate deployment. At these sites, the assemblages were characterised by featherstars (e.g. *Cenolia tripchoptera* and *Cenolia tasmaniae*) and brittlestars (e.g. *Ophiotrix caespitosa*). The only outlier to this was one of the strings at Cape Queen Elizabeth, which consistently recorded a high number of diogenid hermit crabs, along with the featherstars and brittlestars. Featherstars and brittlestars are common and abundant on temperate rocky reefs, and in the case of *Cenolia* spp., they can dominate cryptic habitat (Selvakumaraswamy & Byrne 2000; Barrett et al. 2009; Naughton et al. 2014). There was a high degree of variation in cryptic assemblage at the within-site level, with individual plates often recording distinct assemblages. While this is averaged and variation generally smoothed within a string, it reflects the micro-scale habitat associations that occur for fauna of this size in this environment (Wehkamp & Fischer 2013). Factors such as size of gaps and crevices (Alexander et al. 2009; Hunter & Sayer 2009; Liversage et al. 2017), localised accumulation of sediment (Callaway 2018), algal cover (Robinson & Tully 2000) and substrate type (Callaway 2018) can all influence cryptic invertebrate assemblages over very small spatial scales. The present study could represent a good initial assessment (baseline) of community condition for these sites.

Given most sites appeared to reach a relatively similar faunal assemblage within 8-10 months post-deployment of substrate, cryptic assemblages could provide a useful monitoring tool to address change on a site by site basis, with specific changes in the largely echinoderm dominated (i.e. brittlestar and featherstar) community a potential indicator. Previous studies have suggested that cryptic assemblages may be sensitive to organic enrichment, or at least changes in sedimentation regimes (Callaway 2018; Shelamoff et al. 2019), but it is important to note that these relationships are likely to be inherently complex (Moore 1972). They may be confounded by site differences and therefore the ability to compare between sites could be diminished. For example, increased sedimentation on kelp holdfasts can cause the species assemblages to decrease in biodiversity and become dominated by polychaetes (Smith & Simpson 2002); however, reef habitat offers more complexity than kelp holdfasts. Interestingly, as assemblages in this study were dominated by filter feeders (e.g. featherstars) or omnivores (e.g. brittlestars) increases in organic particles could provide an additional food source, increasing these species in the community up to a certain threshold. However, different species may elicit different responses and that too could be used to provide insights into change (Wing & Jack 2012). Further understanding of how a cryptic assemblage may respond to organic enrichment is required before adoption into a monitoring program.

Spatial variability was difficult to characterise for epiphyte plates. Whilst the community assemblage was quite variable across sites, the biomass on the plates did not vary greatly between sites, either for short (3-4 month) or long (12 month) deployments, with far higher variation within site. As previously noted, there were several highly localised factors that would likely influence species assemblage and therefore total biomass on a plate by plate basis. Light availability, habitat complexity and physical abrasion all play a role in shaping variability on these small scales, often with conflicting effects (Connell 2005; Flukes et al. 2014). For example, on complex reefs, upward facing plates are more likely to be dominated by macroalgae or articulated coralline. A red turfing/coralline matrix or even bryozoans are more likely to dominate plates that are on an angle and receive less light, with concomitant effects on epiphyte biomass (James et al. 2013). Physical abrasion by kelp can reduce the number and type of settling species and the impact of this can differ even between individual plates, with extent depending on the patchiness of the canopy (Connell et al. 2014; Shelamoff et al. 2019). Density of the canopy is also linked to light availability, with algal turfs and articulated coralline algae interacting positively with higher light exposure (Connell 2005). There was evidence of physical abrasion effects on plates in this study, with some plates completely clean when removed from the water after four months of deployment.

Physical abrasion can also play a role in shaping the epiphyte community at the site level (Irving & Connell 2006; Flukes et al. 2014). For example, high energy sites such as Cape Queen Elizabeth or Black Reef Slab, which have dense canopies dominated by *Phyllospora comosa* and are subject to quite high levels of physical disturbance, had epiphyte assemblages dominated by encrusting coralline algae. In contrast, at Bull Bay South which is lower energy with a canopy dominated by *Ecklonia radiata* and *Sargassum* spp., and where the physical disturbance is likely to be considerably less, juvenile algae were more common on the plates. All these factors will influence both species assemblage and biomass on the plates over small and larger spatial scales.

The power analysis indicated the sampling design was not powerful enough to capture meaningful changes from plates for the bulk indicators selected. For example, the results suggested for most sites, upwards of 50 plates would be required to capture a 50% change in biomass, a level of replication not feasible for incorporation into a long-term monitoring program. However, the plates could represent a good tool for looking at enrichment gradients or impact versus control through other metrics, as there were considerable complexity and spatial variability on the epiphyte plates that were not explored. For broadscale monitoring, the ability to detect a response to environmental effects at a level able to be meaningfully related to management action would require, for the indicators selected, an unrealistically large effort in plate replication at the site level.

6.7.4.2 Assessment of temporal variability in indicators

Targeted indicators that have predictable colonisation and that are robust to stochastic processes are ideal for monitoring. Predictable endpoint communities will limit temporal (and spatial) variability that may mask impact. Colonisation effects were observed in all the key indicators tested; however, the degree of variability differed between indicators. While juvenile abalone abundances between sites were widely different, juvenile abalone followed a relatively predictable pattern of colonisation, with counts of zero regularly recorded in the first 2-3 sampling efforts but populations gradually increasing over time. Previous studies using this method to assess juvenile abalone abundance have suggested that a stabilization period of approximately 15 months is necessary (Mundy et al. 2018). This was unfortunately outside the timeframe of this project, but we had hoped that there would still be relatable patterns in the colonisation and abundance data. Assessment of populations at Black Reef Slab and George III using established plates provided an indication of performance after the stabilisation period, and as expected there was little variability in size class structure at these sites over time. The stability in populations post-colonisation suggested this technique may be appropriate for monitoring, as long-term data may be used to characterise individual sites. However, the lack of power in the current design means that only very large changes could be detected with confidence (Mundy et al. 2018). Meaningful monitoring programs should aim to detect an effect or a loss in resilience prior to catastrophic change, so consequently approaches for assessing reef health that have greater power relative to effort are recommended.

Colonisation effects were also evident in cryptic invertebrate assemblages and in most cases the communities were relatively similar at 8-10 months post-deployment. It was unknown if communities stabilised by this time, with previous studies of macrobenthic communities on artificial reefs suggesting colonisation times of anywhere between one and ten years (Perkol-Finkel & Benayahu 2005; Schroeter et al. 2015; Liversage et al. 2017). In this study, there was some evidence supporting more rapid stabilisation of particular components of the community at 6-8 months. The first survey (four months) in the North Bruny region was markedly dissimilar to all others, with gastropods and chitons dominating; however, the dissimilarity decreased over the next two surveys (eith months and 12 months respectively), with assemblages dominated by featherstars and brittlestars. In the SE Channel, featherstars and brittlestars were dominant by the third survey (ten months). This echinoderm-dominated community may be the ultimate endpoint, a hypothesis that could be further explored.

It is possible that cryptic assemblages continue to vary over time responding to different environmental factors independently and synergistically. Such assemblages would be unsuitable to use as indicators of a sustained environmental impact unless it was possible to understand and predict those interactions. However, Schroeter et al. (2015) suggested that direct physical variation may have less impact on assemblages over time, as biotic communities such as algae or biofilms on artificial substrate become more stable and established. In that case, cryptic assemblages might remain relatively stable unless there was a large scale disturbance event that affected the broader temperate reef biota – exactly the sort of circumstance where they would be of use as indicators. Clearly, more work is needed to assess this.

Of all the indicators, epibiotic communities growing on the plates had the most complex temporal response patterns. Epiphytic biomass varied greatly between surveys, which is unsurprising, given that the inherently opportunistic taxa that respond to newly available space are also likely to be responsive to environmental changes associated with season, environmental gradients and depth (reviewed in Wahl 2009; Sams & Keough 2012a).

Although there was some variation between sites, particularly in the North Bruny region, lower biomass was generally observed on plates deployed over winter (June 2017) and higher biomass from the plates deployed over summer/autumn period (January 2017/18). This indicated that factors such as light availability, temperature and potentially the stability of the hydrodynamic conditions (i.e. fewer storm events) may have been playing a role in shaping epiphyte recruitment dynamics (Sams & Keough 2012a; Nelson 2018; Carvalho et al. 2020).

While exchanging plates every 3-4 months provided an insight into temporal fluctuations over a 12-month period, the overall epiphytic biomass on plates deployed for this period was typically very low (i.e. <1 g of biomass/plate), even over summer (January 2017/18), with clean plates a common occurrence. It was extremely difficult to assess the community on plates with such little growth. Previous studies have suggested that communities will become more similar, despite differences in settlement over initial months (Bram et al. 2005; Sams & Keough 2012a), so deployment timeframe and ensuring that the communities are past the initial stabilisation period are critical to limit variability. In this case, the results suggested a deployment period longer than four months is necessary to be able to make meaningful comparisons.

As expected, plates deployed for a 12-month period had much greater biomass of epiphytic growth than those deployed for only four months. At the end of 12 months Cape Queen Elizabeth recorded the highest biomass, in contrast with the four month deployment where Bull Bay South had the greatest growth. This reflects the nature of the communities that dominated the plates at either site; coralline communities, which are slower growing but achieve a higher biomass over 12 months, were dominant at Cape Queen Elizabeth, whilst lighter, patchy juvenile brown and red macroalgae were the most common community at Bull Bay South. Duration of deployment and the resultant effect on epiphyte community composition and biomass are particularly important considerations when reviewing indicators of salmon farming interaction, as increased epiphyte biomass could potentially be a key sign of nutrient enrichment (Legge 2017). However, the significant interaction between site and survey observed in both regions indicated that this relationship is not straightforward and that recruitment processes on temperate reefs are complex and variable. Consequently, further understanding of those confounding factors is required before epiphyte plates could be considered a useful monitoring tool in a broadscale context.

6.7.4.3 Assessment of sedimentation on reef ecosystems to inform impact

Higher amounts of sedimentation were observed at the more exposed sites; however, the percentage organic material was consistent across sites. There were also differences between sampling times, with lower sedimentation in January than June in the North Bruny region, but the samples in January had higher organic content. This may reflect the cumulative effect of inherent differences in the natural processes, with hydrodynamic differences between sites influencing sediment resuspension and seasonal differences in productivity influencing the nature of the sedimented material. For example, greater sediment resuspension might be expected at the more exposed sites (Booth et al. 2000) but increases in resuspension might also be expected in winter (June), and productivity might be expected to be higher in summer (January).

This sediment trap experiment was unable to capture these complex processes effectively. The sediment traps were initially designed to assess the nearfield effects of sedimentation around salmon farms (i.e. Trumpeter lease in North Bruny and Lippies lease in the SE Channel) and as such the design was focused on capturing relatively high loads over short timeframes. The near-field research suggested sedimentation had returned to background within 250-500 m from cages (Chapter 7.2). Given that none of the reef sites examined in this study were that close, evidence of direct impact was not expected. Although results did not suggest that farm derived sedimentation was a major risk for far-field reef systems, the intent was that traps would provide valuable information on background conditions, enabling the establishment of a universal baseline or baseline categories that could help refine a risk-based approach to far-field monitoring. Instead, these results suggest it would be very difficult to modify this approach to detect the influence of farming on sedimentation. The influence of natural variables (e.g. storm events, exposure, productivity) on sedimentation was significant, and both spatially and temporally variable. Thus, the approach would need to be uniquely tailored for each region and specifically modified to address broadscale environmental influences. For example, in this study, storm events featured in several of the deployments which skewed results, with very large amounts of sediment collected in traps at some of the more exposed sites.

It is important to understand both the source of organic matter and the effect it might have on reef ecosystems to accurately attribute risk. Whilst the data from this study did not suggest a significant effect of direct sedimentation from farms on far-field reef systems, the results showed that there would be value in future studies characterising both the natural sedimentation and assimilation processes on reef ecosystems to identify their resilience more accurately. This information could then be used to assess and prioritise risk-based sampling approaches that reflect key interactions at a site-level. A better understanding of temporal influences on both the physical and biological processes in the regions where farming occurs would enable sampling frequency to be tailored to best reflect the relevant timescales. It is clear that sampling must be better aligned with major temporal events such as storms, recognising that these sorts of events can “reset” ecological processes and may change perceptions of sensitivity/ resilience.

Whilst the results from this study showed little difference in% organic carbon content between sites, the composition of this material was not analysed and given what is noted above about the potential sources of sedimentation, it must be acknowledged the organic material at each site may be quite different in composition, and it is really this element that is likely to be most significant from a biological perspective. It is also key to determining the likely influence of farm derived material. It is also important to understand both the source of sedimented material and how that material interacts with the reef ecosystem. Sediment represents a direct food source for sessile invertebrates and filter feeders (Coma et al. 2001; Wing & Jack 2012), but also can have indirect effects. For example, sediment is known to bind to turfing algae and potentially preclude recruitment of macroalgae (Airoldi & Cinelli 1997). Size fractionation of sediments may be useful to examine, with sites that have a larger proportion of fine particles more likely to be susceptible to smothering (Airoldi 1998) whilst analysis of biochemical metrics such as fatty acids or stable isotopes could provide specific information on the likely source of organic matter (White et al. 2017; Woodcock et al. 2017). Incorporation of these approaches would elucidate the physical and biological processes associated with sedimentation on rocky reefs, how these may be influenced by interactions with salmon farming and the risk levels associated, so that risk appropriate monitoring might be developed.

Finally, it is worth noting that the algal canopy itself plays an important role in shaping the effects of sedimentation onto the reef substrate. Consequently, it is important to understand what is being measured at different levels through the canopy. Here, traps accumulated sediment at a higher rate below the canopy than those level with the canopy. This is likely to be due to reduced water velocity below the canopy resulting in a higher settlement rate into the traps compared to the traps exposed to higher water movement level with the canopy (Eckman et al. 1989; Layton et al. 2019). It is also plausible that the traps at the lower level are capturing more resuspended material. Of note, position of the trap had little effect on sediment composition, with percentage organic content generally similar between canopy and sub-canopy traps. Biochemical characterisation of the organic material would aid in the interpretation of this data, with resuspended material likely to have a different stable isotope or fatty acid composition to new material into the system. Further investigation is needed to understand how the canopy and substrate might influence the impact of sedimentation and, as a result, any risks associated with farming interactions on the reef environment.

The sedimentation study results did not suggest strong evidence of a direct impact of sedimentation from salmon farming on far-field reefs. The approaches assessed were not sufficiently sensitive to determine generalised or regionally specific sedimentation rates that could be adopted into a broadscale monitoring program. A better understanding of the natural processes mentioned above might enable a more targeted and effective monitoring approach to be considered. However, given the spatial and temporal variability observed in the current study, it is anticipated that a high level of replication over both space and time will be needed to identify any meaningful level of change with certainty. The trade-off will be cost; comprehensive characterisation of sedimentation on rocky reefs would be a relatively resource intensive exercise, and as such it may be prudent to refine the approaches prior to undertaking further work.

6.7.5 Conclusions

This study aimed to test the suitability of four potential indicators of impacts from salmon farming on rocky reef environments: i) juvenile black-lip abalone abundance, ii) cryptic invertebrate assemblages, iii) epiphyte plates and iv) sediment traps. While each of these approaches had different strengths and weaknesses, in all cases the sampling effort required to identify meaningful levels of change with certainty was beyond the means of this research. Thus, these indicators cannot be recommended to form part of a routine monitoring program without further refining.

While abalone are of commercial importance and there is concern from the fishery regarding the potential interaction between salmon aquaculture and abalone populations, the results suggest it would be a more robust approach to monitor reef health in general in areas of high abalone value, as the inherent variability of abalone populations precludes the identification of any direct causal link with salmon farming. Where broadscale monitoring of reef health can overlay with ongoing stock assessment programs, long-term monitoring programs may help in understanding the influence of aquaculture in combination with other influences (e.g. overfishing, marine heat waves). Similarly, the numbers of epiphyte plates needed for adequate power to detect a meaningful change is well in excess of what could be routinely sampled for monitoring. The complex spatial and temporal interactions between a range of environmental factors, biomass and composition made it very difficult to separate any potential impact from natural variability. Epiphyte plates may be better deployed in gradient studies over short distances from farms as a proxy for local-scale nutrient enrichment, where spatial and temporal effects are likely to be less influential. While cryptic invertebrate assemblages generally showed predictable patterns across space and time, understanding the ecology of these cryptic species would need to advance markedly to predict how they might respond to organic enrichment. At present it is not possible to recommend the adoption of this technique into a monitoring program. Furthermore, there must be a better understanding of the natural process of sediment deposition onto temperate rocky reefs before an attempt to identify the risk or potential impact of salmon farming on these systems using sediment traps can be reliably made.

Overall, the results suggest an approach that assesses the overall health of the reef ecosystem, rather than focusing on targeted indicators would be the best option for long-term monitoring.

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6.8 Synthesis

6.8.1 Interactions between reef ecosystems and salmon farming in Tasmania

Temperate reef ecosystems may interact with outputs from salmon farming in several ways, both directly and indirectly, which can be difficult to predict. The extent to which organic enrichment is likely to impact a reef will depend on a suite of environmental factors, including whether the interaction is acute or diffuse, episodic or sustained, the ambient nutrient levels, localised exposure and water movement, combined or synergistic effects of other disturbances and environmental tolerances of keystone species within the ecosystem. Thus, understanding how any given temperate reef system will respond to outputs from salmon farming requires a detailed understanding of the structure and function of the local ecosystem, and the capacity to relate this understanding to the prevailing regional and/or global pressures.

There is a suite of methods available for characterising the ecology and biology of reef ecosystems, ranging from diver-based census to the use of remote technology such as ROVs or AUVs generally designed to characterise either biodiversity or functionality, or sometimes both. All approaches have benefits and limitations, with the choice of approach usually determined by the nature and limitations of the environment in question and practical considerations such as budget and safety, as well as the research question of interest.

Within Tasmania, assessment of rocky reefs at shallow depths has generally been undertaken using a diver-based underwater visual census method developed by Edgar & Barrett (1997, 1999). This approach provides a comprehensive inventory of fish, invertebrate and algae species. There are numerous benefits to employing this method as part of a monitoring program: i) it is a complete census of marine biota, and provides an ideal baseline for any site, ii) it provides data across multiple trophic levels which will help to detect any structural change in the ecosystem over time and iii) as the methods are standardised the resultant data can be compared across studies providing a broader spatial and temporal context. The results of this study showed that reef systems in the northern Bruny and SE Channel regions were broadly similar to reef systems elsewhere in Tasmania, but also that there were some distinctive elements that may need special consideration.

The Edgar-Barrett surveys indicated that the temperate reef ecosystems in these regions are generally healthy and dominated by a macroalgae canopy of either *Phyllospora comosa*, *Ecklonia radiata* or less commonly, *Macrocystis pyrifera*. The dominant canopy forming species at most sites appeared to be driven by exposure levels at those sites, with *P. comosa* and *M. pyrifera* at the more exposed sites and *E. radiata* at sites with lower wave energy, as per the exposure models developed by Edgar (1984) and Hill et al. (2010). Invertebrate and fish assemblages were typical of southern Tasmania and quite consistent across sites.

The Edgar-Barrett surveys provide a detailed characterisation of reef ecosystems and are invaluable for establishing a baseline with which to review change over time and background variability. However, they are labour intensive to implement and have limited ability to detect fluctuations in short-term nutrient indicator species, such as epiphytic or filamentous algae. This is consistent with the findings of Valentine et al. (2016), who suggested that the development of a more targeted assessment approach that could detect fluctuations of nutrient indicator species applied across a greater spatial area was needed. Consequently, it is recommended that an integrated monitoring program should include both the biodiversity surveys and a more targeted approach, with the two approaches designed to complement and inform each other: the targeted approach aimed at detecting short-term impacts or the potential for loss of resilience and the Edgar-Barrett surveys providing an understanding of baseline conditions and long-term change.

In lieu of the recommendation from Valentine et al. (2016), a novel rapid visual assessment (RVA) technique informed by results from the Edgar-Barrett approach, along with known indicators of organic enrichment on temperate rocky reefs was developed and trialled. The RVA technique took a functional assessment approach, as it was determined that this could provide results more quickly than using biodiversity alone as the key assessment metric. Edgar-Barrett surveys indicated all reefs surveyed within northern Bruny Island and southern Channel were dominated by a strong macroalgae canopy, which underpins ecosystem function, influencing all other ecosystem components (Teagle et al. 2017). Consequently, assessment of canopy cover, sub-canopy and substrate were key components of the RVA technique. Assessment of epiphytic, filamentous and nuisance algal species was also incorporated into the RVA technique, as per the recommendations of Valentine et al. (2016), as well as changes in abundance of filter feeders which may be directly affected by greater rates of organic sedimentation. The RVA technique was designed to be undertaken biannually, both to ensure timely identification of change and to enable some separation of short-term functional response to acute enrichment (i.e. as a result of increased river flow following rain events or spring pulses of nutrient) from longer term sustained enrichment (i.e. that more likely to be due to anthropogenic inputs). It was determined that multi-variate analysis of the resultant data was essential. In these complex systems, relying on a single parameter was insufficient and it was necessary to be able to demonstrate shifts in multiple parameters simultaneously.

The RVA method was continually refined during the project, with a final version implemented in the southern Channel region and used to successfully detect a broadscale enrichment gradient. At present the method shows great promise as a simple and dependable tool for monitoring organic enrichment on temperate reefs. While the RVA technique can be used with confidence to detect change in reef function, it needs reliable baseline data for comparison at suitable spatial and temporal scales, if it is to be used to assist in the determination of attribution or source of impact. Consequently, further work is recommended to a) examine performance of RVA surveys over a longer time series, b) assess the responsiveness of the technique to acute impact/ enrichment gradients and c) determine if the method can be used to examine recovery following the removal of an enrichment source. This information will allow a more detailed assessment of RVA surveys’ ability to detect any loss of resilience or functional changes on temperate reef ecosystems due to salmon farms or other enrichment sources.

Whilst RVA techniques such as the approach designed and tested can increase spatial coverage by reducing dive time at each site, it is acknowledged that fully remote assessment techniques can increase spatial coverage further by eliminating the reliance on divers altogether. With this in mind, towed video was tested, based on the RVA criteria, as part of this project. Towed video was selected because it has a much lower set-up cost and requires less expertise to operate than either ROVs or AUVs. Towed video enabled a substantial increase in spatial coverage, although the trade-off was a considerable loss in both the number of parameters that could be assessed and the accuracy with which they could be assessed. Canopy cover was the only variable that could be measured with any level of accuracy using towed video, but even in this case only extremely large changes could be detected due to the categorical nature of video scoring. This method is less likely to detect a loss of resilience or effects of organic enrichment on reef systems; however, it could be useful as an investigative approach to provide a quick assessment of findings from other approaches. For example, if both RVA and Edgar-Barrett surveys indicated there had been canopy loss at a particular site, towed video could be used to assess whether this was localised or more regional in nature. Further work on remote techniques using ROV or AUV as a “middle ground” between diver-based approaches and towed video need to be investigated further. Given the greater control in operating an ROV, it is possible that use of this equipment may help to overcome some of the issues associated with assessment of towed video, albeit on a smaller spatial scale.

Assessing key indicators as potential “canaries in the coal mine” is an alternate approach to monitoring that may also reduce costs, but only if the indicators are both reliable and timely. In this study the effectiveness of abalone plates, cryptic species assemblages and epiphyte plates were evaluated as potential indicators. Unfortunately, all approaches were found to be highly variable, both spatially and temporally, with power analysis indicating that the replication levels that would be required in order to be able to detect meaningful change were unrealistic and simply could not be achieved within the scope of resources and time. However, some of the indicators such as juvenile abalone were of interest because of the potential for direct interactions with salmon farming. In this case, we suggest that assessment of overall ecosystem health that is possible through biodiversity or RVA surveys would be a better measure of potential risk to the abalone fishery than monitoring juvenile abalone alone. Interestingly, while using epiphyte plates to monitor broadscale impact was impractical, there was some evidence that they may have merit if deployed along an enrichment gradient or to examine acute effects.

Increased sedimentation has been used as a measure of the localised footprint of salmon farming in many research studies (Keeley et al. 2013; Keeley et al. 2019). Sediment traps were deployed to identify whether change in sedimentation rate and composition could be used as an indicator of potential impact at each of the reef sites. While there were some interesting results, proximity to farming did not have a direct effect on the volume of sedimentation at any of our sites. However, sedimentation is complex and, whilst we were unable to establish a direct relationship, the indirect links between sedimentation and key organic enrichment processes may warrant further investigation.

While the aim of this study was primarily to develop methods of monitoring reef ecosystems for impacts from salmon farming, results from this project do provide some further insight into some of the key factors that that are likely to influence the interactions of reefs ecosystems with organic enrichment from farming. Proximity to the source of enrichment is known to be a key determinant of the effects on reef ecosystems; Oh et al. (2015) detected nutrient associated increases in epiphytic and ephemeral algae on reef systems 400 m to 1 km from farms in Tasmania. In this study there was no evidence of any direct effects of organic enrichment on reefs that were more than 1.5 km from fish farms. The RVA method was able to detect a functional response to organic enrichment at sites closer to fish farms, but as described elsewhere it is not possible to ascertain causation given the same sites are exposed to other sources of nutrients (both natural and anthropogenic) and a range of other environmental drivers. This is addressed in the recommendations for future monitoring and assessment below. Wave exposure is a notable example of an environmental factor that was shown to strongly influence reef assemblages in this study. This was evident when examining the patterns in the dominant canopy forming species in the northern Bruny Island and southern Channel regions. Importantly, wave exposure will likely influence both the degree of exposure to nutrients and the characteristics of the community through dilution and dispersal of nutrients, which will determine the inherent sensitivity and resilience to enrichment. It is recommended that monitoring approaches take exposure into account, both in terms of fundamental ecology and biodiversity, as well as how monitoring efforts may be aligned spatially and temporally to the characteristics of the receiving environment.

This project represents the most comprehensive investigation of approaches for monitoring the interactions between Tasmania's temperate rocky reefs and salmon farming to date. We sought to build on existing knowledge and previous studies in Tasmania, with a view to establishing approaches that could be readily incorporated into existing salmon farm monitoring programs, but which would also align with the broader current ecological monitoring of reefs in Tasmania. Many approaches were reviewed and evaluated and whilst some were deemed inappropriate, others may be considered worthy of further assessment. The results will inform monitoring programs and management of these systems into the future.

6.8.2 Recommendations for future monitoring & assessment

Research outcomes from this project have highlighted the value of biodiversity (Edgar-Barrett) and targeted functional (RVA) approaches as tools for monitoring reef ecosystems and for assessing the potential impacts from organic enrichment. Implementation of a combination of these survey methods has the potential to form the basis of a robust reef monitoring program.

It is recommended the following be included in any program to assess the broadscale interactions between salmon farming and rocky reefs in south-eastern Tasmania:

- Biodiversity surveys following the "Edgar-Barrett" method of underwater visual census should be undertaken in each region prior to the commencement of farming to establish baseline conditions.
- Although the nature of reef ecosystems does not easily allow for assessments over a gradient of impact, where possible, surveys should be undertaken at multiple sites at varying distances from the aquaculture, from within 1.5 km to a farm to broadscale to establish regional baselines.
- RVA surveys should be undertaken prior to the commencement of farming at the same sites as the Edgar-Barrett surveys were undertaken to validate and correlate the parameters.
- Once farming has commenced, RVA surveys should be undertaken biannually with a view to evaluating ongoing ecosystem health and to provide an early warning of any potential deterioration or loss of resilience in reef systems. Whilst this approach may not be able to determine direct attribution on its own, it can potentially identify those system changes early enough to inform management decisions (including the need for further investigation using other tools) and mitigation measures that can ensure the long-term sustainability of both reefs and coastal salmon farming.
- Additional evaluation using remote tools (i.e., towed video or ROV) could be undertaken with a view to establishing the value of this approach as a potential rapid response to further investigate broadscale extent of any observations of impact from the RVA or biodiversity surveys.

Where no change has been detected using RVA assessment:

- Edgar-Barrett surveys should be repeated over a regular timeframe (e.g., 5-7 years) to provide an indication of any broader ecosystem changes that may not have been detected through RVA assessment.

Where significant functional change or loss of resilience has been observed through RVA assessment:

- The data should be immediately reviewed to establish significance of observations and a plan for a targeted response to determine specific causal effects should be implemented.
    - This would likely include the use of other available tools/evidence to determine causation, such as CONNIE dispersion modelling described in Chapter 7.3, comparison with patterns observed in other reef monitoring programs, and/or the investigation of local vs regional water quality parameters.
- Edgar-Barrett surveys should be undertaken as soon as practicable to determine the extent of any functional change on community and ecosystem health.
- If RVA and Edgar-Barrett surveys both indicate widespread canopy loss, the use of towed video could be considered to examine the spatial extent of canopy loss.
- A management and mitigation plan should be implemented, with related monitoring (RVA and Edgar-Barrett surveys) undertaken at an appropriate timeframe to assess recovery performance.

While it is anticipated the monitoring program outlined above should be sufficiently robust for detecting change, it is important to note that baseline data are critical to establish both natural variability and the level of change in any specific factor. Ideally, there would be multiple data points prior to any source of change being introduced to the system. However, it is appreciated that more often there is only a single temporal baseline data-point followed by a time-series of post-change data. As temperate reef ecosystems will be influenced by a complexity of environmental factors, providing comprehensive baseline data and a more consistent time-series will markedly increase the ability of any monitoring program to detect change.

In addition, monitoring should be adaptive. Results need to be continually reviewed and assessed, as does the robustness of the overall monitoring program. The longer the duration of a monitoring program, the more faith can be placed in the data. This enables review of data to improve reliability assessments and the sensitivity of the tools recommended. It is also important to regularly review the data for other or extended management tools and applications, such as tipping points and threshold values for key parameters in ecosystems. A longer term understanding of the system response may also facilitate the formation of quantitative metrics such as an organic enrichment index to evaluate ecosystem status. As the monitoring tools and their reliability is improved, sampling effort may be reduced or increased as necessary, depending on the long-term stability of the ecosystem. As noted, the monitoring program itself needs regular re-evaluation and should certainly be reviewed following significant changes to farming practises or other inputs to the system, or before applying in other regions.

We hope that these results and recommendations will help to better understand the importance of temperate reef ecosystems, and how their ongoing condition (sensitivity/ resilience) might be best assessed and monitored. While there is scope for improvement, the results presented here provide a solid basis with which to establish a monitoring program for the future management of temperate reef systems.

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7 Modelling

Assessing two modelling approaches for use in the Tasmanian salmon industry to manage dissolved and particulate waste

7.1 Introduction

Mathematical models are widely used in research of the environmental impacts of salmon aquaculture. These models come in many different forms and underpin management decisions around key issues such as production levels and site selection. This study reviews how modelling is presently used in aquaculture generally and re-examines the needs and expectations for modelling in monitoring and management of salmon aquaculture in Tasmania. Finally, the key objectives of this chapter are to assess two modelling approaches for use in the Tasmanian salmon industry to manage dissolved and particulate waste.

7.1.1 Review of modelling for aquaculture

Nutrient enriched waste (uneaten feed and faeces) output from open-cage salmon aquaculture can affect the local ecology in several ways. Particulate waste sinks to the seafloor and interacts with the biogeochemistry in and on the sediments (Pearson and Rosenburg,1978), whilst dissolved wastes are assimilated through a broader suite of processes (e.g. phytoplankton uptake, nitrification). Monitoring programs often assess these interactions through empirical observation; however, there are many cases where this is not possible, including where the scale of sampling (temporal and/or spatial) would be prohibitively expensive or where sampling or experiments cannot adequately reflect the dynamic nature of the environment or ecological processes. Whilst empirical studies can provide important information about the processes through which an ecosystem assimilates waste, understanding how these processes function across a broad range of environmental conditions (spatially and temporally) requires the use of models.

Computer based simulations and numerical models enable the description of processes difficult to measure, and so are important tools for making quantitative assessments regarding the state of an ecosystem and predicting how it might respond to change. Modelling approaches need i) a clear and unambiguous statement of the mathematical problem (model context), and ii) reliable data/ information for model validation. However, there is often uncertainty present in both the understanding of underlying processes and the data used to validate models, rendering ecological interpretation difficult (Petrovskii and Petrovskaya 2012). Quite often uncertainty can be accounted for and factored into the model through randomness measures or other mathematical approaches.

Issues such as stability and convergence in numerical models are now less important with improved computational capacities. Current mathematical and computational methods have improved to the point where the qualitative analysis of the results that is now the dominant issue in the application of model outputs (Petrovskii and Petrovskaya 2012), making it especially important to provide clear guidance on interpretation. Improved understanding of ecosystem processes incorporated into increasingly sophisticated modelling approaches, results in the production of models more reflective of the complex interactions that exist between salmon aquaculture and the environment, which in turn reliably informs both production and environmental management decisions.

Often the same models are used both for management and research. Research models generally require a lot of data as they need to provide a detailed understanding of ecological and/or physical processes, whilst management models can often function with much less data, particularly where they are based on well understood processes. In Tasmania the environmental models used to inform management of the salmon aquaculture industry have typically been complex research tools; recently, simpler models specifically designed for use by managers have started to appear. These new models need to be validated and their use needs to be well defined.

A broadscale environmental monitoring program (BEMP) was initiated in 2009 throughout the D’Entrecasteaux Channel and Huon Estuary (Ross & Macleod 2012) and has subsequently been used to assess the health of this region and potential impacts of farm production. The data collected through the BEMP provides both baseline data (Thompson et al. 2008) and ongoing condition assessments, but the data also provide a valuable resource for validation of both research and management models. The BEMP program requires collection of both benthic data (chemical and biological) and water quality parameters shown to be indicative of shifts in the system. Broadscale monitoring programs are active in all farming regions around Tasmania except for the Tamar Estuary. Whilst all major salmon farming countries have some level of environmental monitoring embedded within their regulatory frameworks (Wilson et al. 2009) few collect and manage such comprehensive data as required in Tasmania. Countries invariably employ benthic monitoring to assess impacts at a local scale and some countries including Scotland, Canada, Norway and New Zealand require monitoring to be performed at control sites to allow broadscale impacts to be assessed (although none of these countries have mandated broadscale monitoring programs). Whilst such broadscale data are important for routine monitoring, the value of a broad spatial and temporal dataset for model development, calibration and validation cannot be understated.

A biogeochemical (BGC) model of southeast Tasmania was developed as part of a major research project (Volkman et al. 2009) to quantify impacts of a proposed salmon farm expansion in that region (Wild-Allen et al. 2010). The model findings supported several key management decisions, including imposing an industry cap on nutrient inputs to the system, and all production within the region still operates within that restriction. More recently, a Fisheries Research and Development Council project (FRDC-INFORMD2, Condie 2017) enabled the development of modelling tools that utilise some of the underpinning parameterisation and outputs from the southeast (BGC) model and associated hydrodynamic model (Wild-Allen et al. 2010) to undertake more rapid simulations able to inform and support management decisions.

The first of these tools was CONNIE (CONNectivity InterfacE), particle tracking software that runs via a web interface. CONNIE calculates particle distributions for a substance (particle) released from one or many point sources over a nominated period. CONNIE was designed to help in track farm debris, dissolved and particulate waste as well as biosecurity modelling for the salmon farming industry. CONNIE can only be used in regions where there is a hydrodynamic model to which it can connect.

The second tool developed through INFORMD2 was MAREE, a statistical emulator for estimating change in water quality parameters (at the system level) based on a defined change in production levels at one or more farms. Development of MAREE was based on defined statistical relationships between multiple runs of the BGC model calibrated and validated in the D’Entrecasteaux Channel and Huon Estuary (Wild-Allen & Andrewartha 2016).

Both tools rely on having a related hydrodynamic model of water movement for dispersal modelling to be conducted in the regions where they are applied. There are several hydrodynamic models around coastal Tasmania fit for this purpose: ETAS on the east coast (Herzfeld et al. 2015), SETAS in Storm Bay (Jones et al. 2012) and DHD in the D’Entrecasteaux Channel and Huon Estuary (Wild-Allen & Andrewartha 2016). A hydrodynamic model also exists for Macquarie Harbour; however, it focuses on understanding the water column oxygen dynamics (Ross et al. 2017) and particularly the role oxygen dynamics play in response to enrichment. The Macquarie Harbour oxygen model has been validated with real time data, including saturated oxygen data, collected from an array of sensors deployed in the Harbour.

Tasmania is at the forefront of studies on modelling and predicting environmental impacts from salmon farming at the system level. A recent collaboration between CSIRO and the Chilean government saw some of this local expertise used to develop an Integrated System for Aquaculture Management (SIMA 2018) to provide predictive oceanographic climate models for aquaculture. These models integrate a range of sub-models and were designed to provide information to support management and mitigation of environmental impacts at a system wide level. In Norway, Canada and Scotland hydrodynamic modelling and particle tracking have been used alongside biological information to understand biosecurity issues and the risk of disease transmission between farms (Salama & Rabe 2013 Hydrodynamic, MIKE3, FVCOM etc.). Ecosystem models (ECOLab, ERSEM etc.) also have been used to examine system impacts from salmon aquaculture (Falconer et al. 2016) in over 30 studies, with the focus ranging from determination of phytoplankton response to establishing connectivity between and within farms. DEPOMOD is another particle tracking platform widely used to model deposition of solid waste around farm leases (Cromey et al. 2002). Providing DEPOMOD output to establish the benthic footprint of farming operations is a regulatory requirement in Canada and Scotland (Fisheries & Oceans Canada 2016; SEPA 2008). In more recent studies, DEPOMOD outputs were extrapolated to link depositional fluxes in the particulate waste (feed and faeces) to specific benthic impacts (Keeley et al. 2013). DEPOMOD has been validated for salmon farms and farming environments in Scotland, New Zealand and Canada (Cromey et al. 2002; Keeley et al. 2013; Currie et al. 2013).

7.1.2 Objectives

The key objectives of this study were to:

1. Validate the depositional tool DEPOMOD in locations around Tasmania with a view to assessing its amenity for pre-emptive management (Section 7.2).
2. Evaluate the capacity of dispersal modelling from either a hydrodynamic or particle tracking model to calculate the dissolved nutrient footprint emanating from coastal salmon aquaculture and thus provide a low-cost alternative to modelling near field interactions with the environment (Section 7.3).

7.1.2.1 Validation of DEPOMOD

Validation of DEPOMOD requires both the model's depositional flux predictions to be corroborated and some of the model's key characteristics (e.g. currents, sediment type, bathymetry etc.) to be calibrated for Tasmanian conditions. The depositional fluxes returned by DEPOMOD can be compared and associated with a benthic response (e.g. shift in macrofaunal community) to establish a relationship. The overall aim was to establish whether depositional modelling (and DEPOMOD in particular) could be a useful tool to help improve management of local scale impacts.

Hydrodynamic models have an advantage over simpler particle tracking (Lagrangian) models since they can simulate both dissolved and particulate discharges, and represent heterogeneous hydrodynamic fields (Perán et al. 2013). However, these models need more computational power and expert knowledge to both build and operate. Whatever the management motivation for modelling (connectivity, depositional or full biogeochemical), undertaking some hydrodynamic modelling during the early stages of planning and development can improve understanding of the potential interactions and risks associated with marine farm development, enabling farmers and managers to devise/prepare strategies/measures to mitigate environmental impacts. Accordingly, pre-emptive modelling should be considered a good strategy to select sites. Presently, only in Scotland, British Columbia, Tasmania and New Zealand is depositional modelling a prerequisite in salmon farming management strategies. To date there is no mandated requirement for ecosystem or broader biogeochemical modelling anywhere, although this is under consideration in Tasmania.

7.1.2.2 Dispersal modelling for management

As previously noted, modelling research to support salmon farming in Tasmania has clearly shown the need to understand the scale of interaction of interest, to ensure the model is suitable for that level of interaction. Near scale interactions with reefs is an area of contemporary interest, but there is very little research or data in the literature around this topic, particularly from a farm management perspective. Whilst BGC models can be valuable for identifying issues at the estuary scale, many are focused on specific issues e.g. points of accumulation, or interactions between farms, and so may be limited in their application. For example, the ability to model in detail reef-farm interactions would require the major processes underpinning those interactions to be well understood and captured in a BGC model. This would significantly increase the scope and complexity of the model as most would predominately be designed to resolve estuary/ regional scale interactions. It is possible to reduce the scale of the BGC model; however, this is a costly process and may not be necessary when alternatives exist. A possible alternative approach that would be less costly and less time consuming is to use dispersal modelling from either a hydrodynamic or particle tracking model to calculate the dissolved nutrients footprint. This information could be used in conjunction with empirical studies on the impacts of nutrification of reefs around existing farm sites as a pre-emptive modelling tool to establish a risk-based analysis of reef-farm interactions which in turn could support management decisions and monitoring strategies.

7.1.3 References

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7.2 Evaluating depositional modelling of salmon aquaculture waste for a range of sites in Tasmania

7.2.1 Introduction

Extensive work has been done in Tasmania and globally on understanding the benthic impacts of solid waste outputs from salmon aquaculture. Nutrient enriched solids deposit on the seabed and/or suspend in the water column, changing sediment chemistry and providing resources for benthic macrofauna which can lead to marked changes in the benthic assemblages (Karakassis et al. 1999; Hargrave et al. 2010; Pearson & Rosenburg 1978). Although there are processes to regulate waste assimilation, studies have shown that benthic impacts can still vary under similar rates of deposition primarily as a result of differences in sensitivity or resilience in the receiving environment (Macleod et al. 2007; Keeley et al. 2013).

Organic enrichment from farm waste often results in increased sulphate reduction, as oxygen is depleted (Black & Nickell 2014; Hargrave 2010; Jørgensen 1982). As soluble sulphides are toxic to most animals, higher concentrations are associated with reduced ecosystem health (Black & Nickell 2014; Hargrave 2010; Keeley & Taylor 2015). Consequently, the concentration of free sulphides (S) is commonly used to quantify impact around salmon farms (Holmer & Christensen 1992; Chang et al. 2013). Site specific attributes such as sediment particle size and current speeds can have a positive or negative effect on oxygen availability in the sediments (Holmer & Christensen 1992; House 2003), which in turn will influence the rate at which sulphide reduction takes place. A study of sulphide concentration around salmon aquaculture sites in Canada indicated that farm production level and mean current speed accounted for only 37% of the overall variation in sulphide levels (Chang et al. 2013), which suggests that although these influences are important, there is still a significant component of variation (63%) unexplained, and that the relationship between deposition rate of organic waste and sulphide production is site dependent.

Macrofauna are a key indicator of change in benthic condition, with opportunist species often appearing at sites with high organic enrichment. Consequently, total abundances, number of species, and the abundance of key species have all been used to characterise impact level. These features have increasingly been incorporated into impact specific indices such as the Infaunal Trophic Index (ITI) (Word 1978) and AZTI Marine Biotic Index (AMBI) (Borja et al. 2000). These indices include information on the composition of functional groups that reflect the gradient of enrichment under cages and are indicative of the overall health of the system (Macleod & Forbes 2012; Cromey et al. 2002; Keeley et al. 2013). There is a strong relationship between macrofaunal composition, associated condition index, and sediment chemistry. Brooks & Mahnken (2003) showed that the number of taxa at sites with sulphide concentration < 100 μM was double that at sites with sulphide concentration greater than 1000 μM. Hargrave (2010) provided benthic condition ratings for sediments associated with specific sulphide values and identified that variation in impact level was site specific and a result of a variety of factors that may not necessarily be captured by a single index. This study sought to use a range of criteria to accurately evaluate the benthic conditions under salmon pens in Tasmania, basing the assessment on a suite of criteria originally proposed by Macleod & Forbes (2004).

The deposition rate of organic material is an important variable in empirical studies of benthic impacts around salmon farms. Deposition models are commonly used to predict deposition temporally and spatially. These models typically characterise a site using environmental parameters such as water flow and bathymetry, although they can incorporate other physical parameters (e.g., sinking rates for particulates), farm production levels and cage positions. The type of model that is used is dependent on the purpose of the study. Chary et al. (2021) used an ensemble of models (FINS, ROMS and NewDEPOMOD) to develop a methodology for assessing salmon farm site suitability within a region. Bannister et al. (2016) used a 3D hydrodynamic model coupled with a particle tracking model to examine near and far field dispersal of salmon waste, and to look at sustainability within a broader ecosystem. In this study we modelled local scale deposition of particulate salmon waste and assessed how well the results could be used to predict benthic response. To achieve this, we utilised DEPOMOD, the most commonly used deposition model for this type of study. It was developed by the Scottish Association of Marine Science (SAMS) to model the impact footprint associated with solid wastes released from salmon farms (Cromey et al. 2002). This model has been validated at a range of sites around the world (Keeley et al. 2012; DFO 2012; Currie et al. 2013; Cromey et al. 2002). It simulates the movement of particulate matter using a particle tracking module and calculates proportional deposition to determine benthic condition through empirical relationships. DEPOMOD was recently upgraded, with improvements to both the sediment transport and resuspension processes in NewDEPOMOD (Black 2016) but the accuracy of this new model has not yet been tested within Tasmanian sites.

In Tasmania, an application for a new marine farming development plan (or an application for a draft amendment to a marine farming development plan) requires an Environmental Impact Statement (EIS) to be prepared. This may include an evaluation of the deposition zone for the proposed salmon production level. DEPOMOD/NewDEPOMOD could be a very useful tool for this, provided model validation in the local region is performed. In this study the impacts of depositional particulate waste (uneaten feed and faeces) released from open-cage salmon aquaculture at a range of sites in Tasmania using both the old and new versions of the model were characterised and model results validatedl against empirical measurements i) to determine the relationship between predicted deposition rates and benthic indices (biological and chemical) and ii) to evaluate the effectiveness of DEPOMOD for predicting deposition.

7.2.2 Methods

DEPOMOD and NewDEPOMOD require bathymetry and water flow data associated with specific leases alongside farm specific production information (cage dimensions, grid layout, production levels). These model platforms use particle tracking to simulate movement of particulates released from a salmon pen driven by water currents, under the influence of gravity and subject to resuspension once they have settled on the benthos. Benthic impact is quantified through an empirically determined relationship between deposition rate and the Infaunal Trophic Index, an index of change. As this index is not well understood in Tasmanian coastal waters, we determined a relationship based on indices used locally to detect shifts in benthic condition around salmon leases.

Both modelling platforms use an extensive set of parameters that characterise physical processes involved in the production and deposition of particulate waste from salmon farms. Many of these parameter values can be user defined to characterise a specific farm site/operation. They can be determined through empirical measurement or model calibration against observation. However, they have a physical meaning and so care must be taken when changing their value. The most important parameters for production and deposition are presented in this section; however, for a full description of parameters and modelled processes see NewDEPOMOD User Guide (2020) and Cromey et al. (2002).

7.2.2.1 Model test location

Southeast Tasmania

Salmon farming in southeast Tasmania occurs in the Huon Estuary, the D'Entrecasteaux Channel and more recently Storm Bay (Figure 7.2-1). Two farm locations were included in the model testing: Storm Bay One (SB1) and East of Lippies (EL). The SB1 lease was first stocked in 2015 and is situated in Storm Bay off the east coast of North Bruny Island. The lease is in 35-40 m of water with a gradual increase in depth from west to east. There were 12 circular cages on the lease; cages 1 – 6 are in the north grid and 7 – 12 in the south (top right, Figure 7.2-1), with each cage being 76 m in diameter and 24 m in depth. Bathymetry for each study site was collected either as part of zone assessments or through broader benthic habitat mapping programs (Barrett et al. 2001; Lucieer et al. 2007, 2012); high-resolution bathymetry (~1 m) was interpolated onto a 25 m resolution model grid. The bottom sediments around SB1 are characterised as mostly fine sand (particle size < 0.125 mm).

The East of Lippies (EL) lease is located at the southern end of the D'Entrecasteaux Channel in depths of around 45-50 m and has been stocked since 2016. The benthic region is relatively flat, and the lease has 32 cages arranged into two grids: cages 1-16 to the north of the lease and cages 17-32 towards the south (bottom right, Figure 7.2-1). Each cage is 53 m in diameter and reaches down to 18 m in depth. Sediments around the EL sites are characterised as mostly fine sand (particle size < 0.125 mm).

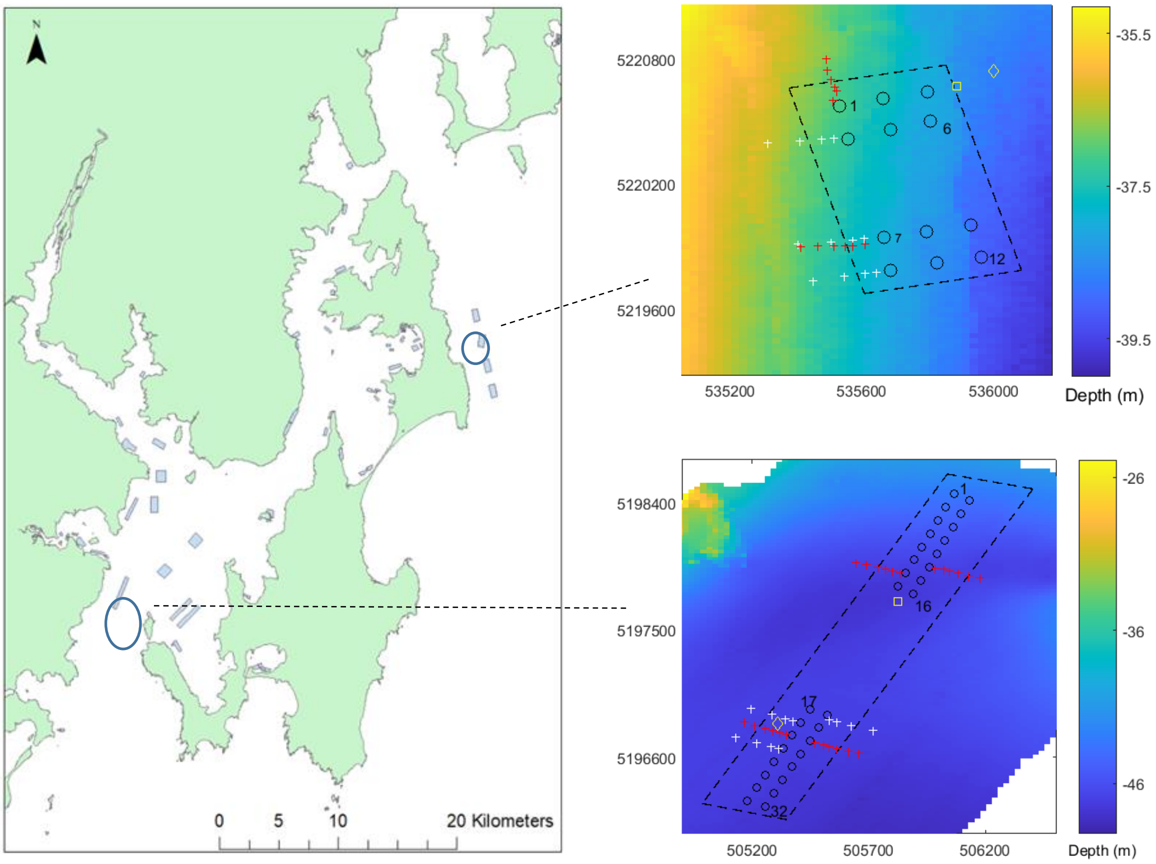


Figure 7.2-1.The two aquaculture sites in southeast Tasmania used in this study were Storm Bay One (SB1) lease located in Storm Bay (top right) and the East of Lippies (EL) lease, located in the southern D'Entrecasteaux Channel (bottom right). SB1 has 12 cages: 1-6 in the northern grid and 7-12 in the southern, while EL has 32 cages: 1 -16 in the northern grid and 17-12 in the southern. The red crosses mark the locations of sediment trap deployments. The white crosses indicate benthic survey locations: lower (southernmost), middle and upper. The yellow circle/square mark the positions of the ADCP deployment at SB1 for January/June. At EL, the diamond/square represents the November/January deployment.

Macquarie Harbour

Macquarie Harbour is located on the west coast of Tasmania (Figure 7.2-2, left). It is a large body of water and the location of aquaculture operations since the late 1980s. The Franklin (FR) lease, in the south of the harbour (top right, Figure 7.2-2), is characterised by a relatively uniform and shallow area on the southern side and a channel of deeper water running through the north side with depths ranging from 27-45 m across the lease. This site contained 40 circular cages, with cages 1-20 located in the northeast section of the lease and 21-40 in the southwest grids respectively. Each cage was 38 m diameter and 15 m deep. Destocking at this site commenced in November 2016 with all sites vacant by mid-April 2017, and restocking commenced in early July 2018.

Table Head lease (TH), in the north of the Harbour, is closer to the narrow entrance to the harbour (bottom right Figure 7.2-2). Depths at this site range from 14-32 m, grading out from shallow close to the shore to a steeper drop off in the northwest corner of the lease. There are 60 cages arranged in two grids; cages 101-115 and 201-215 are in the northwest grid, with cages 301-315 and 401-415 in the southeast. Each circular cage is 32 m in diameter and 10 m deep. Sediment in Macquarie Harbour sites is characterised as mostly silt (particle size < 0.063 mm).

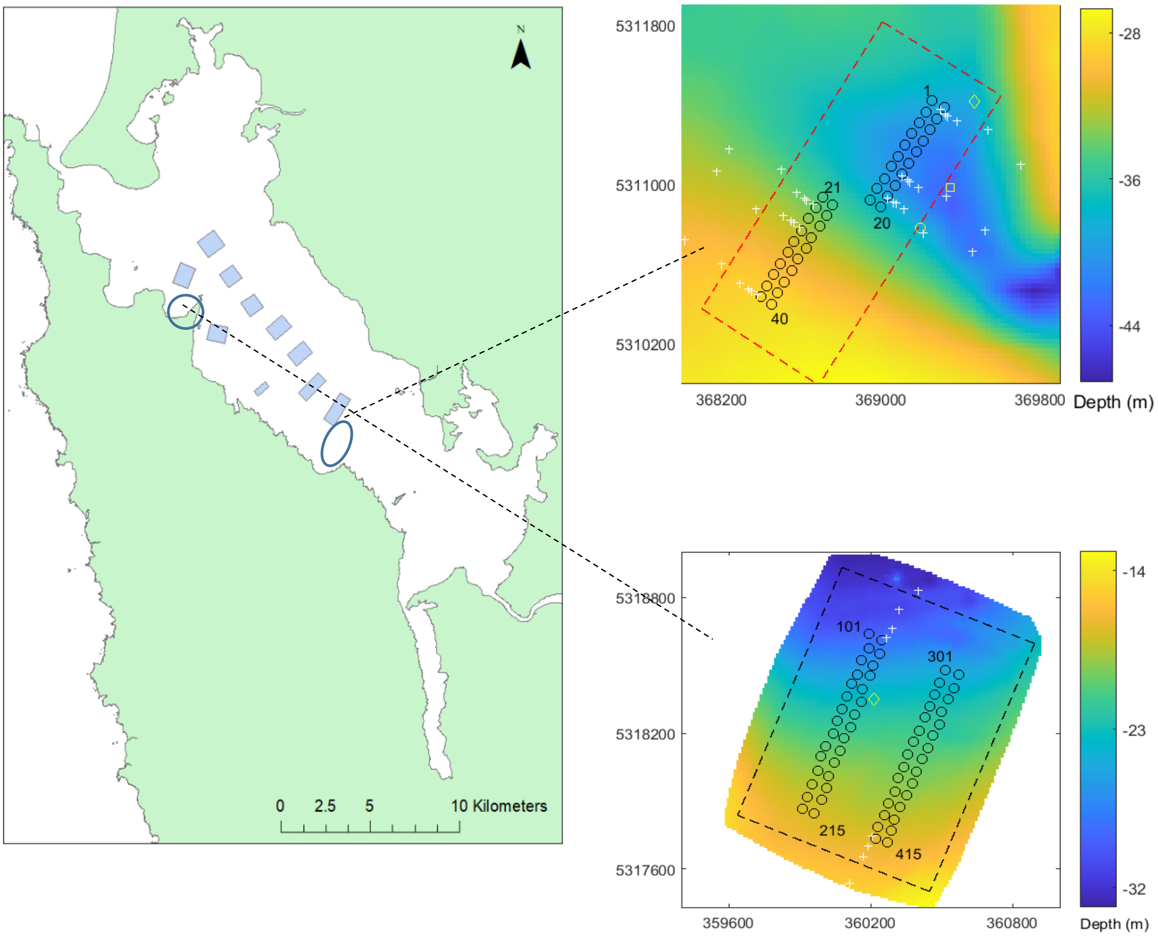


Figure 7.2-2. There are two aquaculture sites in Macquarie Harbour used in this study. The Franklin (FR) lease (top right) is the southernmost lease in the Harbour nearest the World Heritage Area (WHA) and Gordon River. FR has 40 cages: cages 1-20 in the northeast and 21-40 in the southwest. The Table Head (TH) lease is located nearest to the entrance to Macquarie Harbour. TH has 60 cages: 101 -215 in the western grid and 301-415 in the eastern. The white crosses mark sites of the transects at both leases; at FR sediment traps were placed at the same locations of the transects. At FR there were 6 transect in all, 3 to the northwest (NW): lower (southernmost), middle and upper and 3 to the southeast: lower (southernmost), middle and upper. TH had two transects northeast and southwest. The ADCP deployments at Franklin are shown by a circle (March 2019), square (November 2018) and diamond (October 2014) deployment. There was only one ADCP deployment at TH represented by the diamond (in April 2017).

7.2.2.2 Hydrography

The D’Entrecasteaux Channel and Huon Estuary are micro-tidal regions, with a neap-spring period of approximately 14 days. Long term flow regimes are driven by the Huon River, with surface flow exiting the north D’Entrecasteaux and only weaker flow moving toward the southern boundary. The bottom flow enters predominantly via the south D’Entrecasteaux. Average currents around EL range from 0.1 ms^{-1} at the surface to 0.01 ms^{-1} on the bottom (Herzfeld et al. 2010). Depth averaged flow in Storm Bay is generally low (< 0.01 ms^{-1}) and is driven by both connectivity to the Derwent Estuary and D’Entrecasteaux Channel, and offshore gyres (Herzfeld 2008). SB1 lease is considered an exposed, “offshore” environment as Storm Bay can be subject to high wave action and storms.

Water currents in Macquarie Harbour have been shown to be complex, a result of the narrow entrance, bathymetry and influence of the King and Gordon rivers. The upper harbour and entrance are generally dominated by surface flow with weaker sub-surface flow increasing towards the Gordon River end of the harbour; however, this regime varies according to river influence and tides (Koehnken 2002).

DEPOMOD assumes spatially invariant water currents and as a result only a single Acoustic Doppler Current Profiler (ADCP, NortekAQ Aquadopp 400KHz) deployment was required for each lease to provide flow data. Current profiles were collected using a bottom mounted ADCP attached to a tripod and an acoustic release buoy (Vemco VR2AR) via a weighted snag line. Deployment times are detailed in Table 7.2-1. Bin sizes for all deployments were 2.5m with the velocity at each depth (Table 7.2-1) assumed to be the middle of the relevant bin. Magnetic declination was either set in the instrument prior to deployment or accounted for manually in post processing of the data. All data were averaged for each hour. The variance in water currents due to tidal influence was calculated using t-tide, a MATLAB program (Pawlowicz et al. 2002) (Table 7.2-2). The analysis showed that flow at all sites was largely driven by factors (wind, river flow) other than tidal forces. Tidal ranges for each region were taken from the Bureau of Meteorology website (BOM 2019) and included in the models (Cromey et al. 2002).

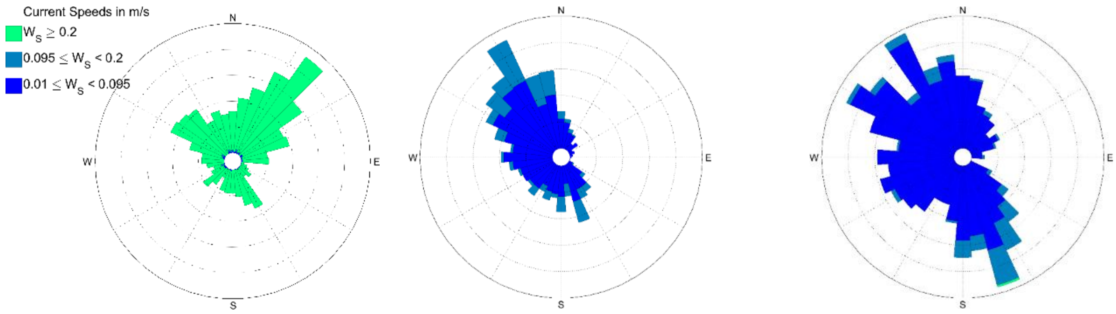
Table 7.2-1 Current data taken from ADCP's located at all sites over a range of dates. Tidal components were estimated using t_tide (Matlab 2019a). Each period of current data was used in one or more scenario. These scenarios were designed i) to investigate depositional footprints (D), ii) to validate sediment trap observations (S) or iii) to provide a comparison of benthic indices with deposition (B).

Site (Deployment)	Scenario	Date	Layer	Depth (m) above seabed	Mean speed (ms^{-1}) full (residual)	Mean direction (˚) full (residual)	Variance due to tide (%)
SB1 (January)	D, S, B	17/01/2017	surface	38.5	0.78 (0.71)	161 (140)	13.9
			mid	18.5	0.07 (0.065)	244 (167)	22.3
		–	bottom	3.5	0.06 (0.05)	215 (186)	21.3
SB1 (June)	D, S, B	07/06/2017	surface	38.5	0.64 (0.63)	216 (114)	3.5
			mid	18.5	0.055 (0.048)	230 (178)	27.0
		–	bottom	3.5	0.065 (0.057)	223 (180)	19.2
EL (November)		24/10/2017	surface	45.5	0.248 (0.23)	200 (196)	18.0
	D, S, B		mid	24.5	0.098 (0.07)	139 (158)	36.4
		-	bottom	3.5	0.097 (0.08)	164 (174)	33.4
EL (January)		15/12/2017	surface	46	0.58 (0.50)	165 (170)	6.3
	D, S		mid	25	0.098 (0.081)	192 (190)	28.0
		–	bottom	3.5	0.088 (0.072)	170 (166)	34.9
Franklin (October)	D, B	4/10/2018	surface	40.5	0.32 (0.31)	150 (160)	3.6
			mid	22.0	0.06 (0.057)	237 (202)	6.1
		-	bottom	3.5	0.07 (0.068)	209 (238)	1.4
Franklin (March)		21/03/2019	surface	39	0.30	216	-
	S		mid	22.5	0.08	243	-
		-	bottom	3.5	0.10	224	-
Franklin (September)		24/09/2014	surface	38.5	0.60 (0.59)	163 (156)	3.3
	D, B		mid	21.5	0.042 (0.04)	188 (178)	11.6
		– 11/12/2014	bottom	4.0	0.064 (0.06)	177 (173)	7.4
TH (May)		04/05/2017	surface	20.0	0.25 (0.24)	189 (202)	7.1
	D	– 03/06/2017	mid	11.0	0.095 (0.094)	171 (187)	5.3
			bottom	2.0	0.038 (0.037)	169 (180)	4.5

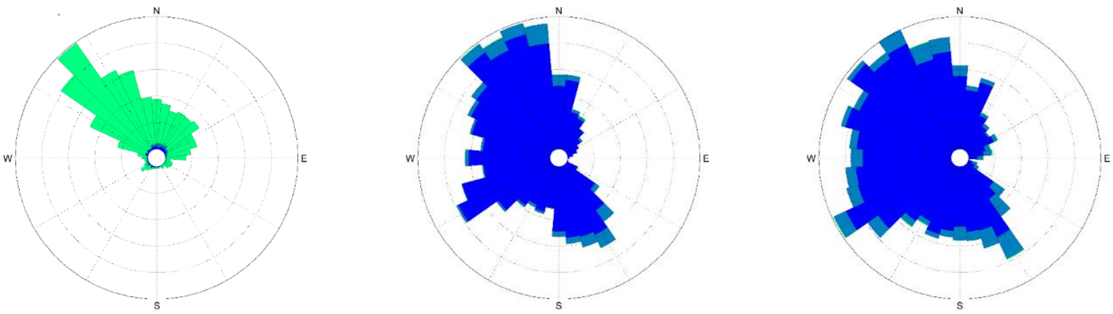
Based on Table 7.2-1, SB1, FR and TH are categorised as depositional sites based on the low mid-bottom water currents. EL is categorised as dispersive given the mean (full) bottom current speeds (0.095 ms^{-1}) are around that required for resuspension in DEPOMOD (Cromey et al. 2002). Although the March deployment at FR also had relatively high bottom current speeds, the time period was very brief (3 days) and was during a storm in the harbour. Tidal influences were low in Macquarie Harbour sites, with most water movement caused by asymmetric influences such as wind and river flow. Tidal influence was much greater in SB1 and EL; however, the main flow was still due to asymmetric (non-tidal) causes. Table 7.2-1 also provides details of which current data set was used for sediment trap comparison (S), benthic validation (B) or to calculate the depositional footprint (D) at each site.

The current data (Figure 7.2-3) show a detailed description of the range of current speeds and directions at three depths (surface, midwater and bottom). These figures have been scaled to show currents below and above 0.095 ms^{-1}, which is the speed at which resuspension occurs in DEPOMOD for material residing on the benthos for less than four days.

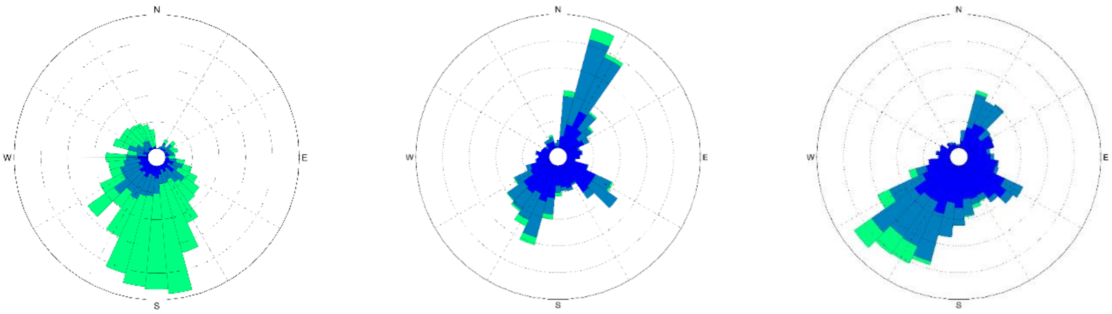
SB1 January 2017



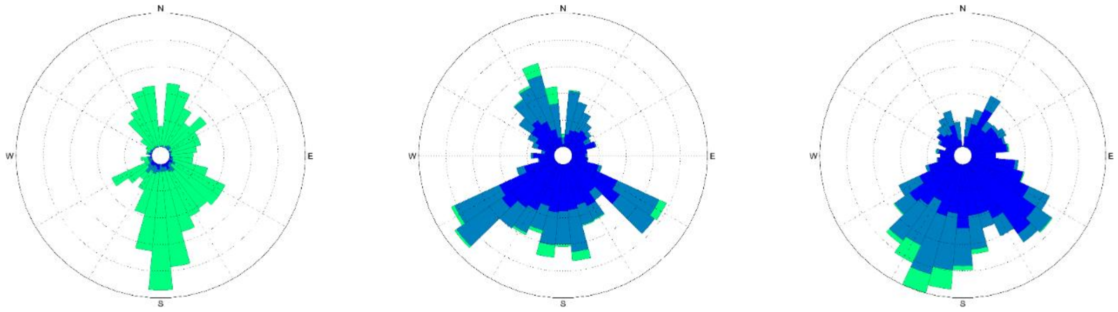
SB1 June 2017



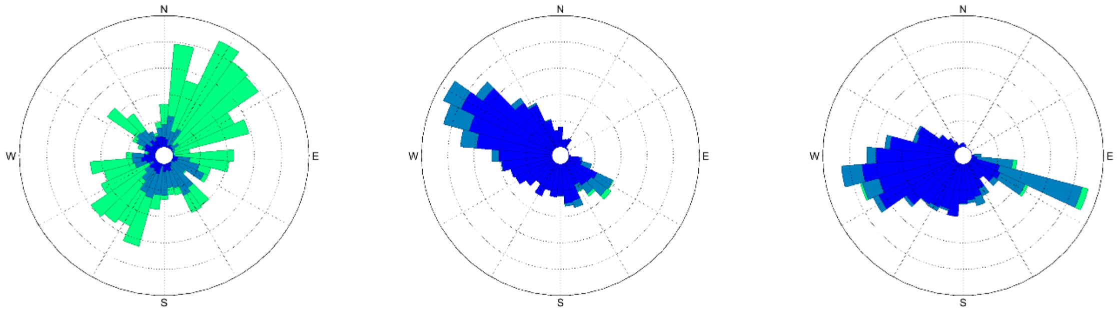
EL November 2017



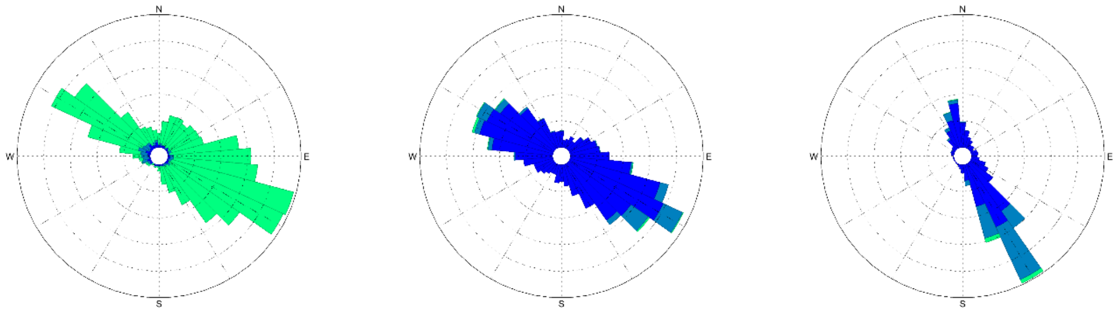
EL January 2018



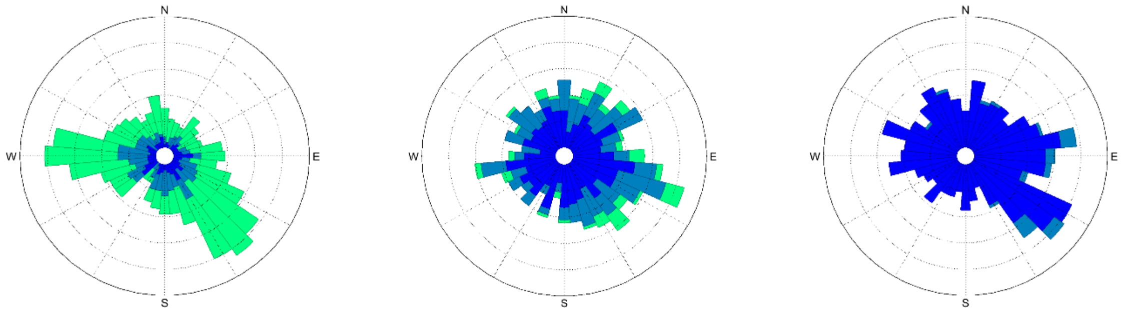
FR October 2018



FR September 2014



TH May 2017



Surface **Middle** **Bottom**

Figure 7.2-3 Water currents (speed and direction) for surface, midwater and bottom (left to right) at all sites. The current speed scale was chosen to highlight the point that resuspension occurs (< 0.095 ms^{-1}).

Resuspension has been shown to be inaccurate in DEPOMOD at dispersive sites (Keeley et al. 2013). In NewDEPOMOD resuspension parameterisation is not fixed at 9.5 cms^{-1} but can be adjusted primarily using the critical shear stress parameter (which has a default value of 0.02 Pa). It should be noted that we have not shown the current data for the March 2019 deployment at FR in Figure 7.2-3 as it was for 3 days only and is not used to characterise flow at the site. There is however a summary of the conditions seen in that deployment provided in Table 7.2-2.

7.2.2.3 Feed data

Feed input for each site was provided as daily input per lease or cage group (Table 7.2-2) and was adjusted to be input into the model at the cage level. Table 7.2-2 includes detail on which feed data were used to investigate depositional footprints (D), validate sediment trap observations (S) or provide a comparison of benthic indices with deposition (B) at each site.

Table 7.2-2 Daily feed rates for each site and period of each model run. Low, moderate and high are used to provide context across scenarios and are not representative of whether an individual lease is in a period of low, moderate or high biomass in their production cycle. Scenarios reflect where the data are being applied to i) investigate depositional footprints (D), ii) validate sediment trap observations (S) or iii) provide a comparison of benthic indices with deposition (B).

Site	Scenario	Feed (kg d^{-1})	Scenario
	Jan. 2017 South Cages	15216 (High)	D, S, B
	Jun. 2017 North Cages	14023 (Moderate/High)	D, S, B
	Jun. 2017 South Cages	12675 (Moderate)	D, S, B
East of Lippies	Nov. 2017 South Cages	22131(High)	D, S
	Jan. 2018 North Cages	3496 (Low)	D, S
	Average Mar., Sep. 2017	15981 (High)	B
Franklin	Oct. 2018 NW cages	5290 (Low)	D, B
	Oct. 2018 SE cages	4900 (Low)	D, B
	Mar.2019 NW cages	11866 (Moderate)	S
	Mar. 2019 SE cages	9766 (Moderate)	S
Table Head	Apr. 2017 NW cages	5273 (Low)	D
	Apr. 2017 SE cages	6891 (Low)	D

7.2.2.4 Model parameters

For the comparison between the two platforms, default model parameters were used. Although most parameters can be customised, this is usually done after model results are either compared to observations (e.g. horizontal, vertical diffusion) and validated, or determined empirically (e.g. sinking rates of fish faeces and different feed sizes). In NewDEPOMOD parameters used to calculate resuspension such as the critical shear stress parameter (Tau$_C$) can also be changed for more accurate predictions; however, these parameters have not yet been measured empirically in Tasmania. The key parameterisation values in this study are summarised in Table 7.2-3. For more discussion and a full list of the parameters used, see NewDEPOMOD User Guide (2020) and Cromey et al. (2002).

Table 7.2-3 Parameter values used in DEPOMOD and NewDEPOMOD (Cromey et al. 2002)

Parameter	Value
Sinking rate feed	0.083 ms^{-1}, std 0.015
Sinking rate faeces	0.032 ms^{-1}, std 0.011
Waste Feed	3%
Digestibility	85%
Percentage water	9%

7.2.2.5 Model validation

Model runs of DEPOMOD and NewDEPOMOD were validated using empirical data collected from sediment traps and benthic surveys. Sediment trap data provide a direct comparison between observed and modelled deposition rates of particulate waste, whereas benthic data enable specific comparison between deposition and ecosystem response. Sediment traps were deployed at all sites (Figure 7.2-1 and Figure 7.2-2) except for Table Head, for the periods shown in Table 7.2-4. In the southeast the traps were placed at 0, 35, 65, 100, 150, 200 and 500 m from the cage, whereas at Franklin they were 0, 35, 50, 100, 250 and 500 m. This approach enabled comparison with predicted rates and determination of an offset term equivalent to background deposition, estimated using data from the traps at 500 m from the cages. This background sedimentation rate was then added to the model results (Weise et al. 2009) to represent the net sedimentation rate.

Table 7.2-4 Sediment trap deployments and benthic surveys at all sites.

Site	Benthic Survey	Sediment Trap
SB1	02/03/2017	23/01/2017 – 31/01/2017
SB1	03/08/2017	21/06/2017 – 04/07/2017
EL	01/03/2017	03/11/2017 – 10/11/2017
EL	02/09/2017	17/01/2018 – 02/02/2018
Franklin	17/10/2018	21/03/2019 – 24/03/2019
Franklin	27/02/2019	

Two sediment cores (radius: 22 mm, aspect ratio: 6) were attached to a metal crucifix and deployed on a concrete base, 2 m from the bottom, with a sub-surface midwater buoy to keep the traps upright. A bungy line was used between the buoy and crucifix to absorb water movements. Samples were washed to remove salt and noticeable organisms were removed using a pair of fine point tweezers. Trap contents were filtered through pre-weighed glass fibre filters using a vacuum filtration system and washed with milli Q water to remove salt from the sample prior to noticeable organisms being removed using a pair of fine point tweezers. Sediment covered filters were then freeze dried to constant weight before being processed to determine% organic matter using a muffle furnace.

To assess 'model fit' we used the criteria described in Jusup et al. (2009). In their approach, a line of best fit using ordinary least squares (OLS) is constructed between observed and modelled deposition. A perfect fit between model and observation would be a line with slope k=1 and intercept l=0. Along with R^2, t- statistics were used to calculate if k and l were significantly different from 1 and 0 respectively. A Wald test was used to see if the line with k=1 and l=0 was significantly different from the fitted model. Using these results, four categories were used for model fit:

1. Very good - R^2 is significantly different from 0 and the model does not significantly differ from the line with k=1 and l = 0.
2. Good - R^2 is significantly different from zero but either k or l is significantly different from 1 or 0 respectively.
3. Fair - R^2 is significantly different from 0 but both k and l differ significantly from 1 and 0 respectively.
4. Poor - R^2 is not significantly different from 0.

We also used the mean absolute relative error (MARE) to examine model accuracy (Cromey et al. 2002) calculated by,

$$\text{MARE} = \frac{100}{n}\left(\sum_{i=1}^{n}\left|1-\frac{x_i}{y_i}\right|\right) \qquad (1)$$

where n is the number of observations and (x_i, y_i) are the i^{th} pair (of n) of model, observation values, respectively. This error measure shows the average percentage difference between observed and modelled deposition.

For benthic validation, indices were collected along transects 0, 50, 100 and 200 m from the cage (Figure 7.2-1) at EL and SB1 and at 0, 35, 50, 100, 250 and 500 m from the cage at Franklin (Figure 7.2-2). An ordinary least squares (OLS) regression model was 'fit' to modelled deposition and observed biological (total abundance (ind. m^{-2})) and chemical indices (sulphides (µM)) data, respectively. Predicted abundance and sulphides, calculated from fitted models, were then compared with observed values to examine correlation (Riera et al. 2017). Analysis was conducted using 'fitlm' in Matlab 2019a modelling with 'RobustOpts' to account for variation in the observations. It should be noted an offset term was not added to the deposition rates in the benthic validation study. Here we were only interested in how the deposition being added (by salmon farms) to the natural system is driving change.

We included two extra model runs for NewDEPOMOD at EL using a reduced current data set between 1/11/2017 – 14/11/2017 from that shown in Table 7.2-1. This reduced set was used in conjunction with the feed data (Table 7.2-2) that corresponded to the dates the benthic surveys were performed at that site. We took this step because the results gained by using the full current data set in the benthic validation runs correlated poorly with the benthic indices. Keeley et al. (2013) found that DEPOMOD results did not correlate with benthic indices in dispersive sites and furthermore results were greatly improved if resuspension was turned off in the model. Although NewDEPOMOD was developed to improve resuspension as a process this was not calibrated for any of the three sites examined in this study. The reduced period of current data used had lower mean current speeds and therefore resuspension was less of an issue.

7.2.2.6 Benthic indices

The indices used in the benthic validation were based on the work in the southeast (Macleod & Forbes 2004) and Macquarie Harbour (Ross & Macleod 2017). Other indices have been identified as indicators of change or impact around salmon leases in Tasmania; however, only a subset is used in this study with the aim to determine if modelled deposition rate could be used to predict change in these indices.

Benthic sampling was conducted at each site on the dates shown in Table 7.2-4. Macrofauna samples were collected using a Van Veen grab ($0.1m^2$), with collected sediment put through a 1mm mesh sieve and fixed in 10% buffered formalin solution. Samples were sorted, and animals removed were counted and stored in 70% ethanol before identification to the lowest practical level for calculation of biological indices.

Sediment cores for the measurement of physico-chemical parameters were collected using a boat deployed penta-corer (core radius: 22 mm) at all sites listed above. Suitable quality cores were collected with one kept for sulphide testing and the top 3 cm of another collected for loss on ignition analysis. Total free sulphides (TFS) was measured in a 2 ml sample collected from 3 cm below the sediment surface using a TPS uniProbe Sulphide Ion Selective Electrode calibrated against a sulphide standard concentration series (Wildish et al. 1999).

7.2.2.7 Comparison of depositional footprints

DEPOMOD and NewDEPOMOD were compared using their predicted depositional footprint at each study site. Previous studies have identified 2 g C $m^{-2}d^{-1}$ as a threshold for impacts (summarised in Keeley et al. 2013). It should be noted this rate is equivalent to ~ 1500 g solids $m^{-2}\,y^{-1}$ assuming solids are 50% carbon and converting from a daily to yearly rate. This yearly rate was used to compare the predicted footprints by calculating the amount of material captured within the contour bounded by this value. However, it is important to note that there is no assertion this is a threshold input level in the regions of study. It has simply been used to compare between model results.

NewDEPOMOD was the only platform used to characterise deposition at the FR lease. NewDEPOMOD was developed to be able to model sediment deposition processes including undulating bathymetry of the type seen at FR (Black et al. 2012). The current fields and feed rates used for each model run for depositional footprint analysis, are shown in Table 7.2-1 and Table 7.2-2. In the case of the sediment trap validation runs, resuspension was turned 'off' in both models. In DEPOMOD resuspension is an option whereas in NewDEPOMOD setting $Tau_c = 2$ stops material resuspending.

7.2.3 Results

7.2.3.1 Model validation

Observed sedimentation rates at the depositional sites (SB1 and FR) were consistently higher at the cage (0 m) reducing significantly with distance. In contrast, at the more dispersive East Lippies site, there was no clear pattern with distance.

The scatter plot of observed versus modelled deposition rate with regression line and 95% confidence interval, for all surveys at each site is shown in Figure 7.2-4. The statistical analysis (Table 7.2-5) shows that NewDEPOMOD provided a ‘very good’ fit at SB1 and FR (R^2 differs significantly from 0 and the regression line with slope 1 and intercept 0 cannot be rejected). In contrast DEPOMOD provided a ‘good’ fit at SB1 (R^2 differs significantly from 0 but the slope differs significantly from 1); DEPOMOD was not assessed at FR. Both NewDEPOMOD and DEPOMOD provided a ‘poor’ fit at EL based on the assessment criteria (R^2 did not differ significantly from 0). In all plots at least > 75% of all points were contained within the 95% confidence limits.

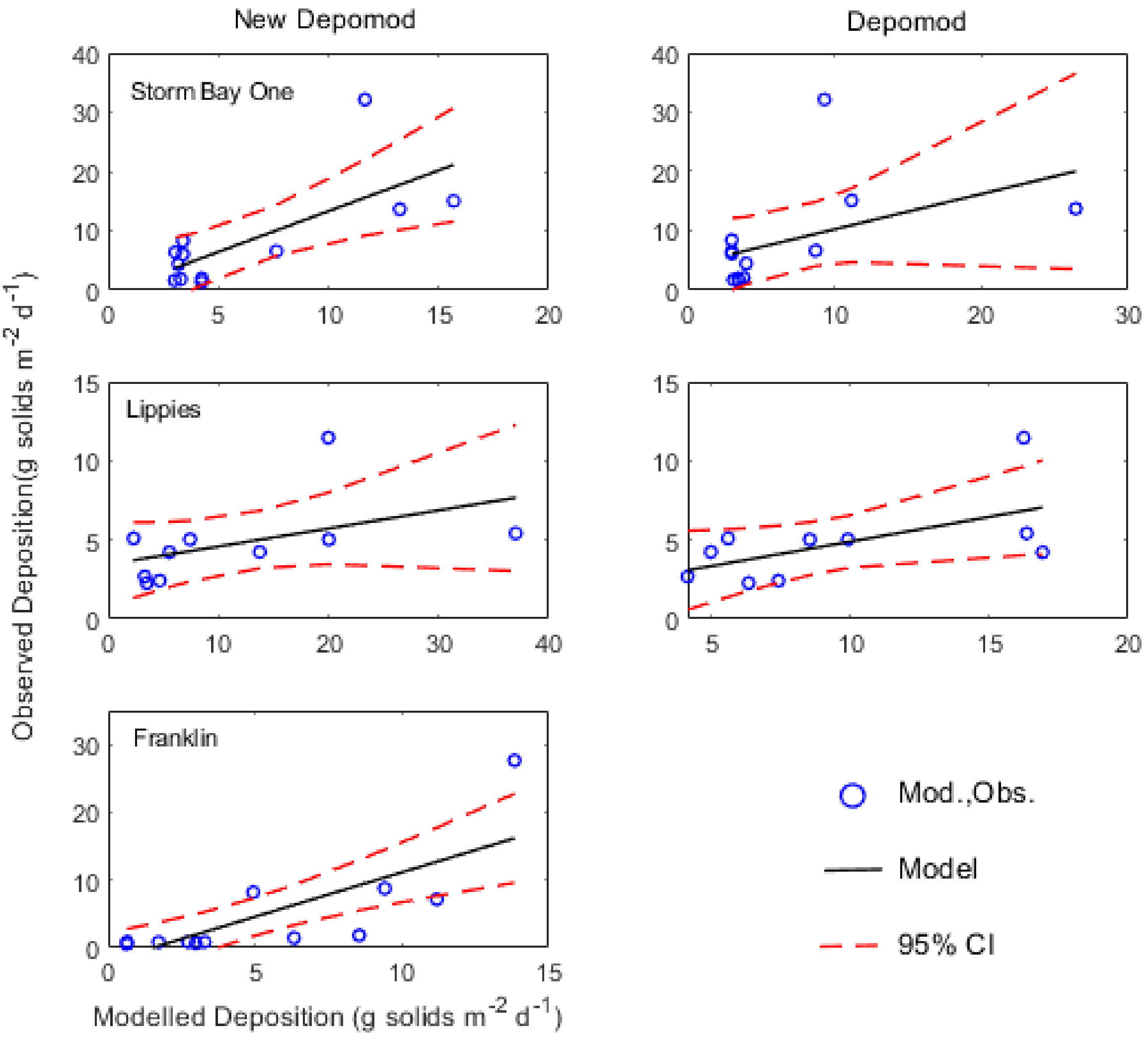


Figure 7.2-4 “Predicted vs. observed” scatter plot (blue circles) with the regression line (black) obtained by ordinary least squares method. Dashed curves (red) represent 95% prediction confidence interval. The Figure on the left/right represent NewDEPOMOD/DEPOMOD results. The top, middle and bottom rows show results for Storm Bay One, East Lippies and Franklin respectively; DEPOMOD was not run at Franklin.

Note only the sediment traps at distance of 0-100 m were used. Traps at 150 m in the southeast and 250 m at FR recorded sediment rates equivalent to the background rates measured at 500 m. The background rates added as offsets to each model output were 0.64 and 3.5 (g solids m^{-2} d^{-1}) at SB1 (January and June, respectively), 1.8 and 0.65 at EL (November and January, respectively) and 0.64 at FR.

Table 7.2-5 Summary of the ordinary least squares estimation and statistical testing for observed versus modelled deposition at each site across all surveys.

Coefficient	OLS Estimate		Standard Error		Null Hypothesis	Test Statistics		P-value	
	ND	D	ND	D		ND	D	ND	D
Slope (*k*)					k = 1				
Storm Bay One	-0.91	0.47	0.27	2.21		t = -0.35	t = -2.28	**0.73**	0.05
Lippies	0.078	0.225	0.06	0.14		t = -16.12	t = -5.64	0.00	0.00
Franklin	1.31		0.30			t = 1.03		**0.32**	
Intercept (*l*)					l = 0				
Storm Bay One	0.75	2.81	2.06	0.23		t = 0.36	t = 1.27	**0.72**	**0.23**
Lippies	3.39	2.33	0.90	1.48		t = 3.75	t = 1.58	0.01	0.15
Franklin	-1.99		1.94			t = -1.02		**0.33**	
Slope and Intercept					k = 1, l = 0				
Storm Bay One	-		-			F = 2.07	F = 4.34	**0.15**	0.04
Lippies	-		-			F = 26.81	F = 6.91	0.00	0.01
Franklins	-		-			F = 0.04		**0.85**	
Determination (R^2)					R = 0				
Storm Bay One	0.54	0.32	-			F = 13.8	F = 6.18	**0.00**	**0.03**
Lippies	0.14	0.17	-			F = 2.51	F = 2.88	0.15	0.13
Franklin	0.58		-			F = 18.7		**0.00**	

As stated previously, MARE represents the mean percentage difference between observed and modelled deposition and can be used therefore as a measure of model accuracy. Comparison between the accuracy of the models based on MARE (Table 7.2-6) shows SB1 results were the most accurate with NewDEPOMOD performing slightly better than DEPOMOD with a score of 62.89% compared to 68.69%. The EL models were less accurate with DEPOMOD (117.42%) performing better than NewDEPOMOD (148.57%) which was the least accurate of all the sediment trap validation studies. FR performed only slightly better at 135%. These results can be placed in context by comparing them to those found in similar studies by Cromey et al. (2002) at 13% and Jusup et al. (2009) at 48.9%. Overall SB1 was the only site where model accuracy was considered reasonable in comparison.

Table 7.2-6 Model accuracy results.

Study	Cromey et al. 2002 DEPOMOD	Jusup et al. 2009 KK3D	Storm Bay One NewDEPOMOD DEPOMOD	East of Lippies NewDEPOMOD DEPOMOD	Franklin NewDEPOMOD
Accuracy	±13%	±48.9%	±62.89% ±68.69%	±148.57% ±117.42%	±135%

The relationship between modelled deposition and total abundance was well represented by a 2nd degree polynomial in both the southeast (combined SB1 and EL) sites and FR respectively (Figure 7.2-5). Abundance changed significantly with deposition at the southeast site (one-way ANOVA, F = 43.6, $p < 0.000$). Correlation between abundance and deposition rate remained positive at all organic loads. Similarly, at the Franklin site abundance changed significantly with deposition rate (one-way ANOVA, F = 6.65, $p = 0.0168$). Abundance peaked at around 4 g solids m^{-2} d^{-1} and reduced thereafter indicating conditions for macrofauna were worsening. Once again > 75% of data points were within the 95% confidence interval.

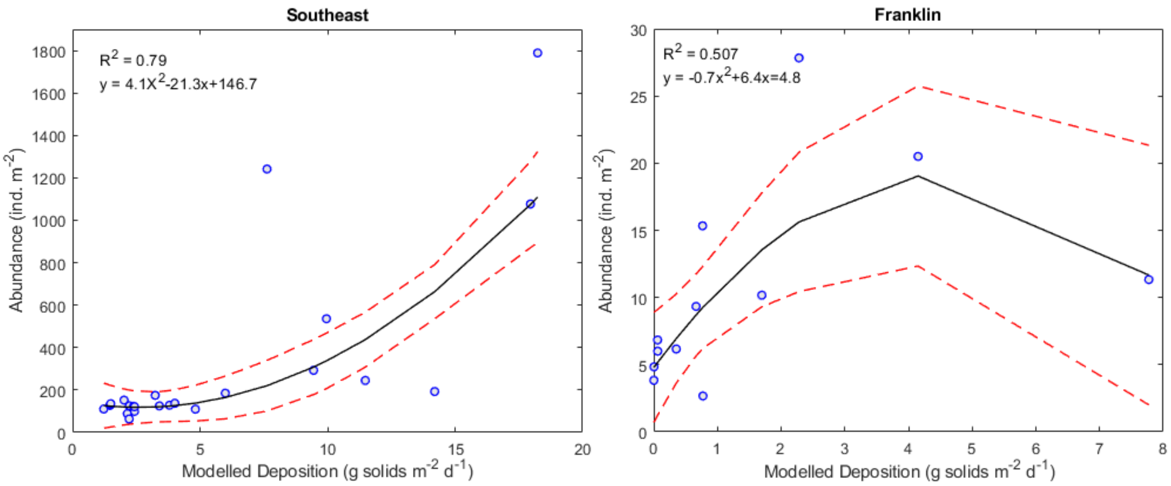


Figure 7.2-5 Scatter plot of predicted deposition versus observed abundance in southeast (combined Storm Bay and East of Lippies sites) and Franklin. The regression line (black) and 95% confidence intervals are also included.

Predicted versus observed abundance in the southeast sites (both separate and combined) was highly correlated ($R^2 > 0.7$) (Figure 7.2-6, Table 7.2-7). At FR this correlation was slightly lower ($R^2 = 0.62$). The relationship between predicted and observed abundance was higher in the combined southeast region than the independent sites at SB1 and EL.

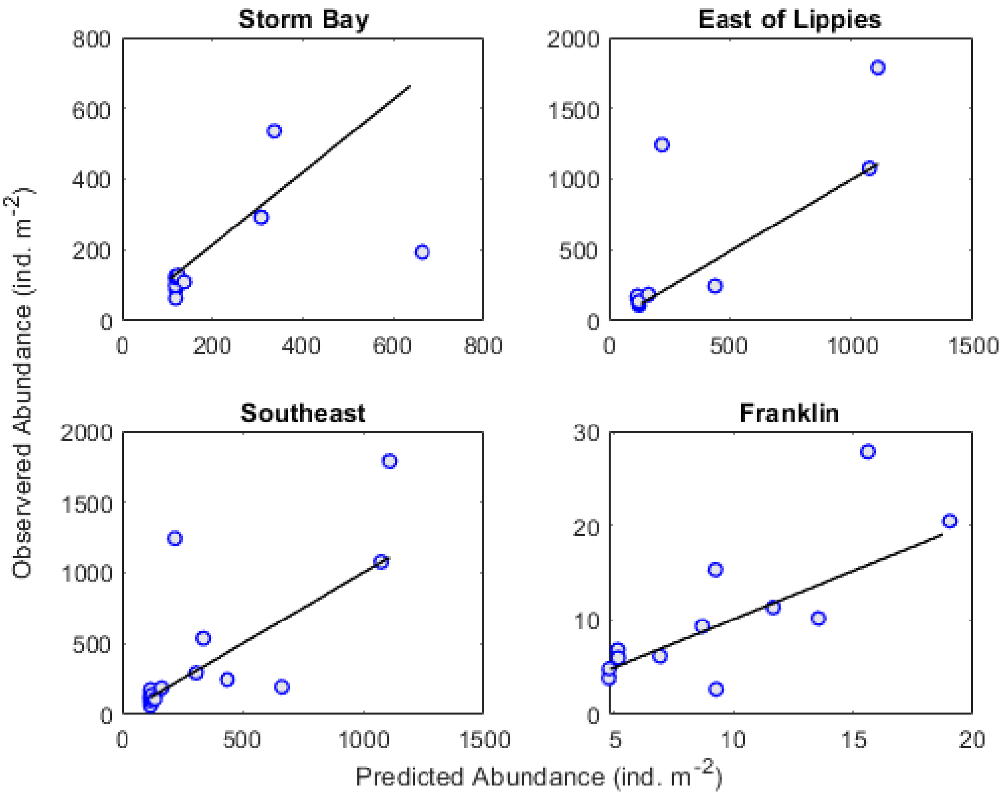


Figure 7.2-6 Scatter plots for predicted versus observed abundance for all sites including the combined southeast region with regression (black) lines.

Table 7.2-7 Statistics for relationship between predicted and observed abundance.

Site	R^2	F	P	B1(slope) t-statistic	p
SB1	0.84	60.6	<0.0001	7.7475	< 0.000
EL	0.73	30.2	<0.0002	2.2025	0.052
Southeast	0.88	166	<0.0000	12.695	< 0.000
FR	0.618	18.8	0.0015	4.3355	< 0.001

The relationship between sulphides and deposition rate was well represented by a 2nd degree polynomial in both the southeast and Franklin regions (Figure 7.2-7). Sulphide concentration correlated well with deposition rate at both southeast combined sites (one-way ANOVA, F = 4.44, $p < 0.025$) where sulphides increased to a deposition rate of ~11 g solids $m^{-2}d^{-1}$, decreasing thereafter. Similarly, at the Franklin lease sulphides correlated well with deposition rate (one-way ANOVA, F = 6.65, $p < 0.017$), increasing until deposition reached ~4 g solids $m^{-2}d^{-1}$ and decreasing thereafter. In both models, 75% of data points were contained within the 95% confidence interval.

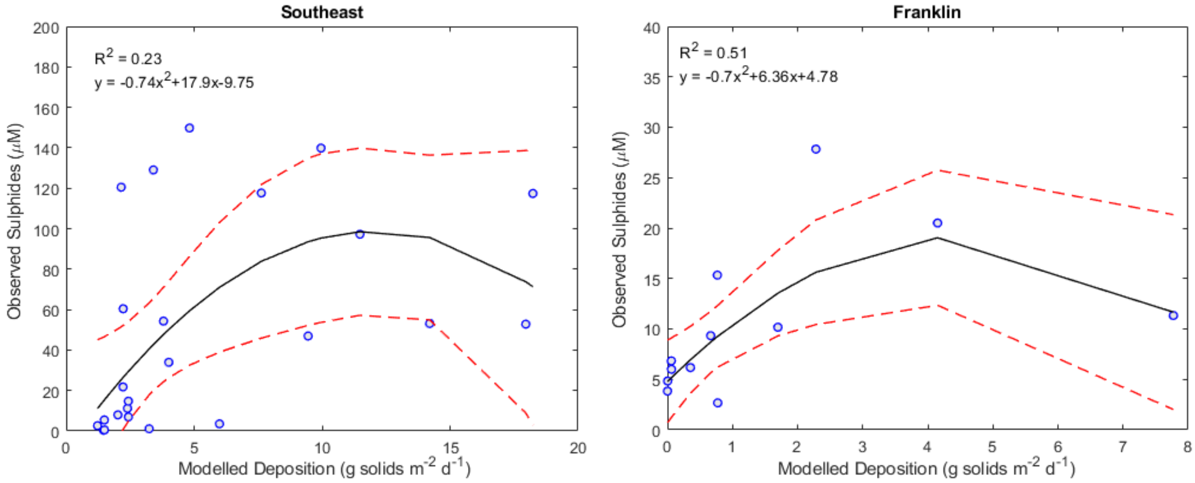


Figure 7.2-7 Scatter plot of predicted deposition versus observed sulphides in southeast (combined Storm Bay and East of Lippies sites) and Franklin. The regression line (black) and 95% confidence intervals are also included.

Predicted versus observed sulphides in the southeast sites combined were correlated (R^2 = 0.38), however this was largely driven by the EL site where the correlation was high (R^2 > 0.65), whilst at SB1 there was no correlation (Figure 7.2-8, Table 7.2-8). At FR predicted versus observed sulphides were strongly correlated (R^2 = 0.6) (Figure 7.2-8, Table 7.2-8).

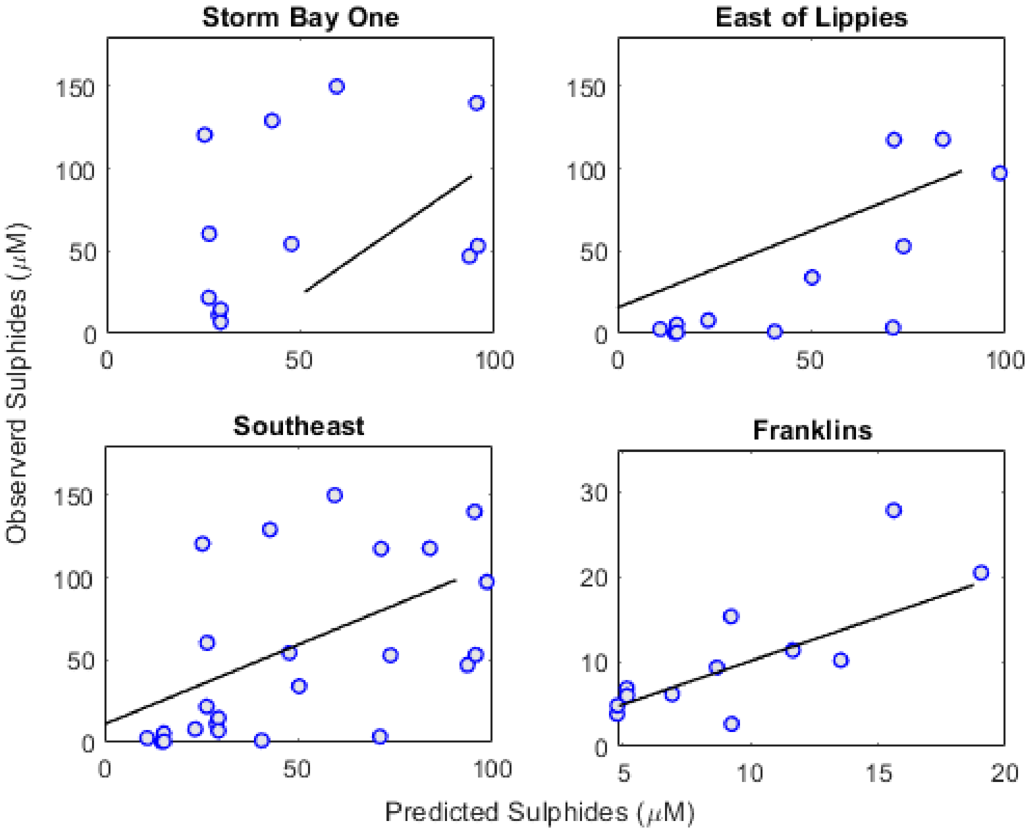


Figure 7.2-8 Scatter plots for predicted versus observed sulphides for all sites including the combined southeast region with regression (black) lines.

Table 7.2-8 Statistics for relationship between predicted and observed sulphides.

Site	R^2	F	P	B1(slope) t- statistic	p
SB1	0.001	1.02	0.34	1.0084	0.337
EL	0.652	21.6	<0.001	4.63	<0.001
Southeast	0.38	14.8	<0.0008	3.5306	< 0.002
FR	0.60	17.2	<0.002	4.04	< 0.002

7.2.3.2 Depositional footprints

Storm Bay One

The depositional patterns predicted by DEPOMOD and NewDEPOMOD at SB1 differ (Figure 7.2-9 and Figure 7.2-10), with the footprint extending slightly further to the east/southeast in the DEPOMOD model output. NewDEPOMOD predicts both a larger footprint and total mass of organic solids contained inside 1500 (g solids m^{-2} y^{-1}) contour, for all production and flow regime scenarios (Table 7.2-9).

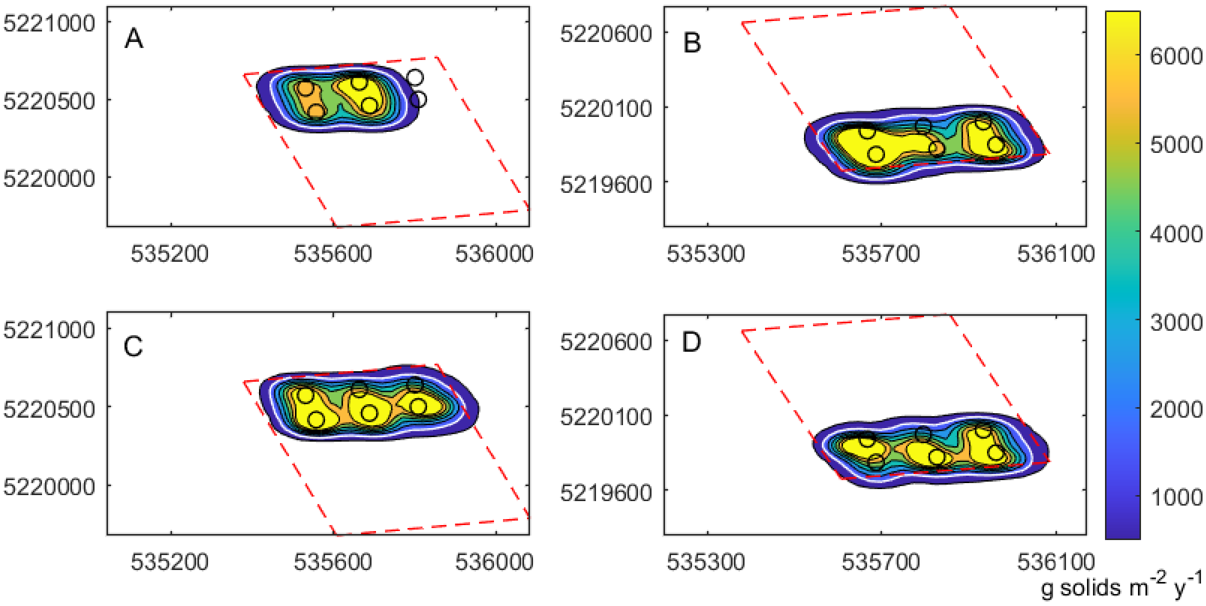


Figure 7.2-9 Deposition rates (g solids m^{-2} y^{-1}) at Storm Bay 1 as predicted by NewDEPOMOD. A represents deposition around northern cage grid for January 2017, B is southern grid January 2017, C is northern grid June 2017, D is southern grid for June 2017. The 1500 g solids m^{-2} y^{-1} is applied in each Figure as an expected benthic impact threshold (white line). The black circles are cages, whilst the red dashed line represents the lease boundary.

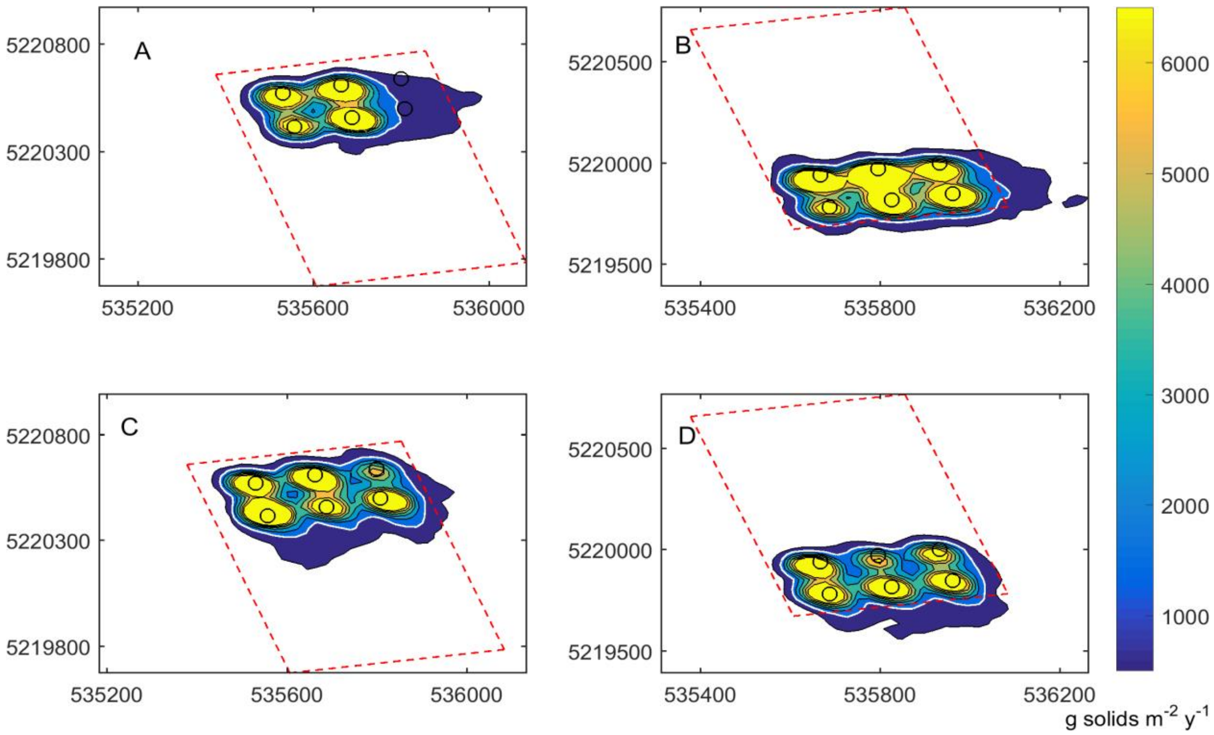


Figure 7.2-10 Deposition rates (g solids m^{-2} y^{-1}) at Storm Bay 1 as predicted DEPOMOD. A represents deposition around northern cage grid for January 2017, B is southern grid January 2017, C is northern grid June 2017, D is southern grid for June 2017. The 1500 g solids m^{-2} y^{-1} is applied in each Figure as an expected benthic impact threshold (white line). The black circles are cages, whilst the red dashed line represents the lease boundary.

Table 7.2-9 Total solids contained within the 1500 g solids m^{-2} y^{-1} contour (benthic impact threshold) and the area impacted at lease.

	DEPOMOD		NewDEPOMOD	
Date – Cage Group	Mass (Tn y^{-1})	Area (ha.)	Mass (Tn y^{-1})	Area (ha.)
Jan 2015 - North	457	8	505	10
Jan 2015 - South	856	14	896	16
Jun 2015 - North	665	12	827	15
Jun 2015 - South	579	12	736	15

East of Lippies

Both models predicted low deposition concentrations at EL (Figure 7.2-11). Only the DEPOMOD modelling showed exceedance of the nominal benthic impact threshold (1500 g solids m^{-2} d^{-1}), and only on the northern grid under the January 2018 production scenario (Figure 7.2-11A). Under the November 2017 production scenario, DEPOMOD suggested the deposition on the southern grid was less than 50 g solids m^{-2} y^{-1}, a level unlikely to be detectable. Only the northern grid showed any real deposition beneath the cages using the full current data sets and DEPOMOD (Table 7.2-10). NewDEPOMOD predicted an asymmetric footprint with low concentrations around both grids in January 2018 and November 2017, but this did not exceed the nominal benthic impact threshold (1500 g solids m^{-2} d^{-1}) on either occasion (Figure 7.2-11B, D).

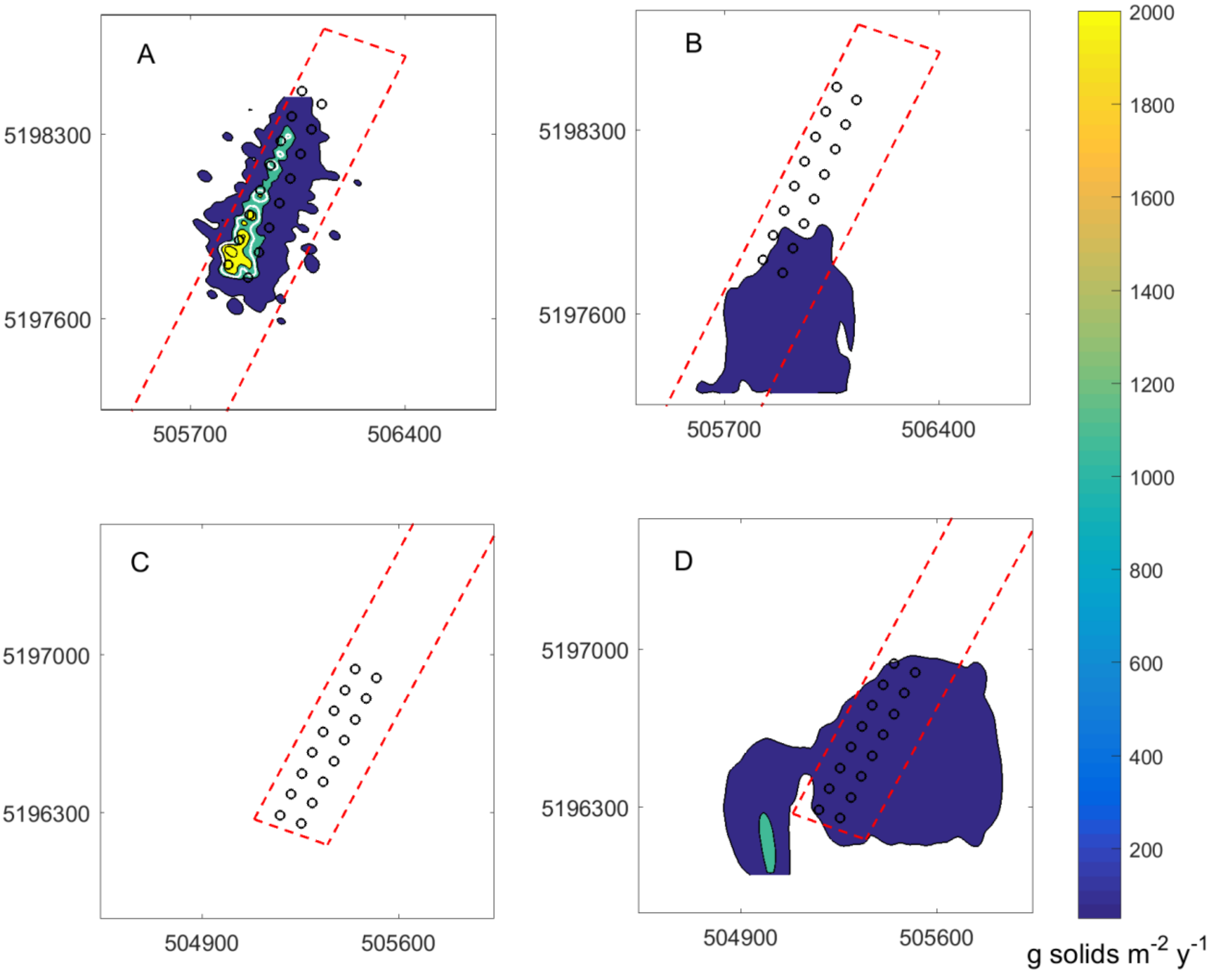


Figure 7.2-11 Deposition rates (g solids m^{-2} y^{-1}) at East of Lippies based on feed rates in Table 7.2-2. A is DEPOMOD result for northern grid in January 2018, B is NewDEPOMOD results northern grid in January 2018, C is DEPOMOD results for southern grid in November 2017, and D the NewDEPOMOD result southern grid in November 2017. The white line (visible only in A) is the 1500 (g solids m^{-2} y^{-1}) contour, or the expected benthic impact threshold. Black circles represent cages and the red dashed lines represent the lease boundary.

The footprint produced using NewDEPOMOD and the reduced current data (as described in methods), and applying the feed rates for March and September 2017 resulted in greater levels of waste beneath the cages (Figure 7.2-12) and a markedly different pattern of deposition observed using the full current data set in the November scenario (Figure 7.2-11D). The area bounded by the benthic impact threshold (1500 g solids m^{-2} y^{-1}) contour for each scenario varied appreciably between different production scenarios (Table 7.2-10). The model outputs (using NewDEPOMOD) suggested much greater deposition beneath the southern grid with the reduced time series of current.

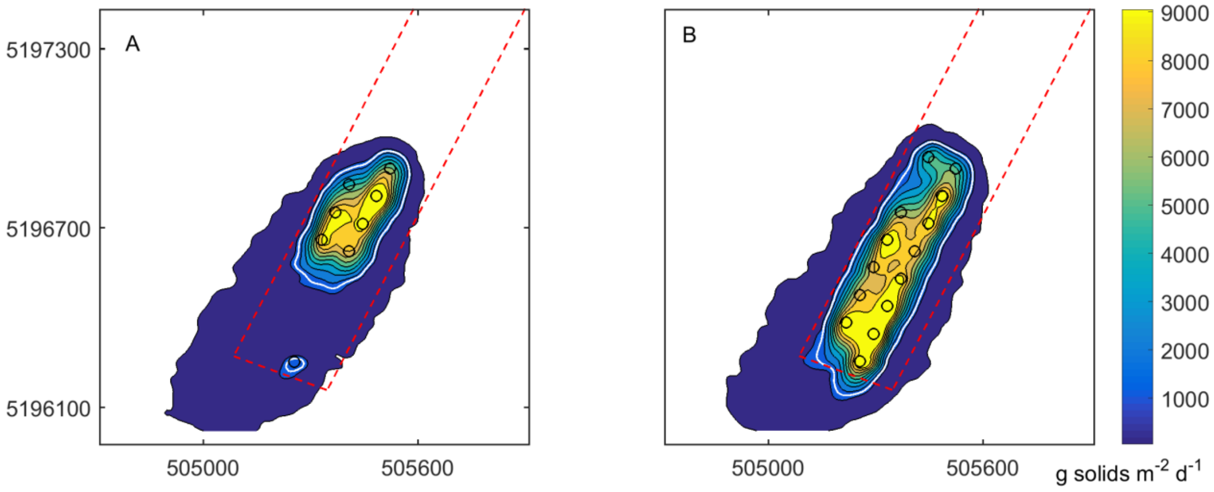


Figure 7.2-12 Deposition rates (g solids m^{-2} y^{-1}) at East of Lippies using a reduced 2 weeks of current data from 3 – 17/11/2017. A is NewDEPOMOD results (southern grid) for March 2017 feed rates, B is for September 2017 feed rates. The expected benthic impact threshold (1500 g solids m^{-2} y^{-1}), (white line) is applied in each Figure. Black circles are the cages (notice not all cages were operational in each month) and the red dashed line represents the lease boundary.

Table 7.2-10 Total solids (Tn) and area (Ha.) contained within the 1500 g solids m^{-2} y^{-1} contour and the area impacted at lease.

	DEPOMOD		NewDEPOMOD	
Date - Cage Group	Mass (Tn y^{-1})	Area (ha.)	Mass (Tn y^{-1})	Area (ha.)
Mar 2017 - South	-	-	407	7
Sep 2017 - South	-	-	838	13
Nov 2017 - South	0	0	0	0
Jan 2018 - North	63	3	0	0

Franklin

The depositional patterns calculated using the two different ADCP datasets reflected the direction of the current (Figure 7.2-13). The modelling showed the deposition footprint extended from the cages in the direction of the prevailing currents, i.e. in a north-westerly direction when using the October 2018 current data and towards the south east when using the September 2014 data. In all scenarios, most of the deposition was <500 g solids m^{-2} y^{-1} with higher concentrations being restricted to beneath the cages. Under low production (October 2018 feed rates) the area that exceeded the nominal benthic impact threshold (1500 g solids m^{-2} y^{-1}) almost doubled when using October 2018 compared to September 2014 currents, but remained very similar under the high production scenario, suggesting in this case the footprint is less dependent on currents (Table 7.2-11).

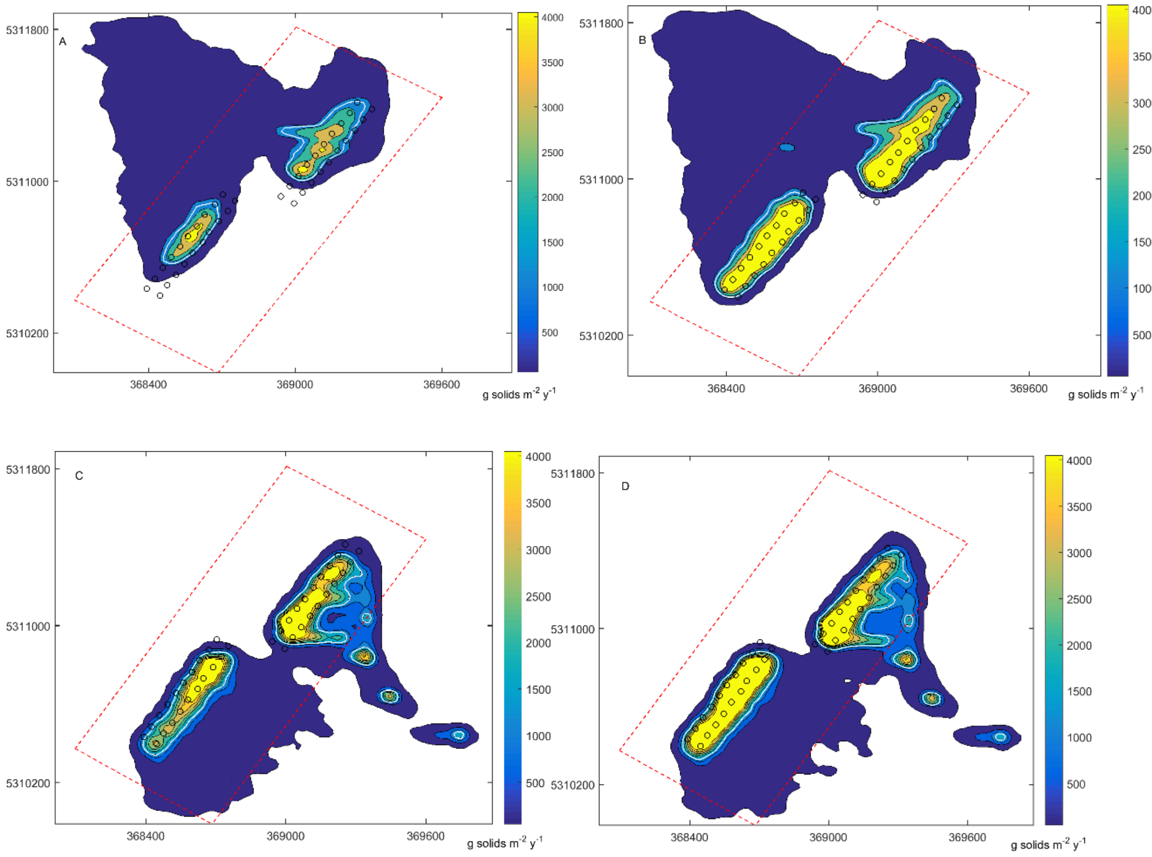


Figure 7.2-13 Deposition rates (g solids m^{-2} y^{-1}) at Franklin using NewDEPOMOD only. A shows results for October 2018 feed rates and current data, B is for March 2019 feed rates and October 2018 current data, C is results for October 2018 feed rates and September 2014 current data, D is for March 2019 feed rates and September 2014 current data. The expected benthic impact threshold (1500 g solids m^{-2} y^{-1}) (white line) is applied in each Figure. Black circles are the cages and the red dashed line represents the lease boundary.

Table 7.2-11 Total solids contained within the 1500 g solids m^{-2} y^{-1} contour and the area impacted at the lease.

	NewDEPOMOD			
	October 2018 currents		September 2014 currents	
Feed	Mass (Tn y^{-1})	Area (ha.)	Mass (Tn y^{-1})	Area (ha.)
October 2018	87	4	170	6
March 2019	255	7	256	7

Table Head

The deposition pattern for Table Head was calculated using both DEPOMOD and NewDEPOMOD (Figure 7.2-14). A comparison using the current data obtained at Table Head (Table 7.2-1) and feed rates from April 2017, a period of high salmon production at the lease, showed more local deposition using DEPOMOD (Figure 7.2-14A) with a more dispersed footprint including low level concentration (< 100 g solids $m^{-2}y^{-1}$) using NewDEPOMOD (Figure 7.2-14B). Only a subset of cages was operational at this time (A and B in Figure 7.2-14). In the comparison between models the depositional concentration that exceeded 1500 (g solids $m^{-2}y^{-1}$) was markedly different with DEPOMOD predicting over double the area and mass of waste within this contour compared to NewDEPOMOD (Table 7.2-12).

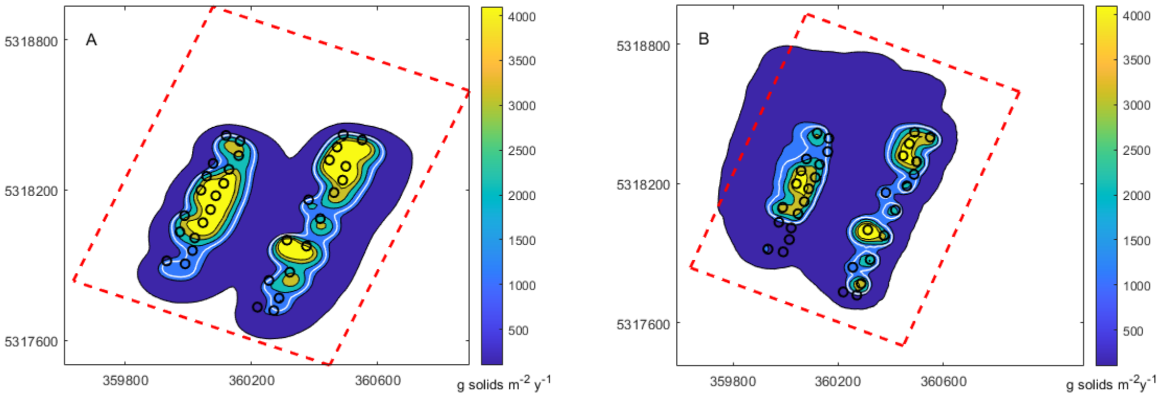


Figure 7.2-14 Deposition rates (g solids m^{-2} m^{-1}) at Table Head. A shows results for April 2017 feed rates and May 2017 current data using DEPOMOD, B is for the same data using NewDEPOMOD. The 1500 (g solids m^{-2} m^{-1}) contour (white line) is applied in each Figure. Black circles are the operational cages and the red dashed line represents the lease boundary.

Table 7.2-12 Total solids contained within the 1500 g solids m^{-2} y^{-1} contour and the area impacted.

	DEPOMOD		NewDEPOMOD	
Feed	Mass (Tn y^{-1})	Area (ha.)	Mass (Tn y^{-1})	Area (ha.)
April 2017	415.8	12.3	173.3	6.6

7.2.4 Discussion

7.2.4.1 Model validation

Modelled versus observed deposition

Using sediment traps to validate the depositional modelling had mixed success. The EL deployment did not show a good relationship between modelled and observed deposition. Although the relationship at the other sites was much more in agreement, an assessment of the percentage difference between observed and modelled deposition (i.e. MARE test) indicated lower accuracy than that reported in the literature (Cromey et al., 2002; Jusup et al. 2009). Those comparative studies were shorter in duration and did not test model accuracy over as large an area as was done in this study. In the initial validation of DEPOMOD (Cromey et al. 2002) sediment traps were deployed for 24 hours and at a maximum distance of 65 m from the cage. Similarly, the Jusup et al. (2009) study had traps placed at a maximum distance of 40 m from the cage. The rationale for undertaking the validation with traps deployed close to the organic matter source and only for a short duration was to ensure farm produced organic matter was the dominant source of deposition collected. Although a longer duration may be more indicative of the mean conditions at a site, there is the issue of storms providing major resuspension events capable of filling the traps with sediment, as was the case for the study in FR where several traps had to be discarded. Furthermore, the particles collected close to the cage are more influenced by gravity, a process better understood and possibly more accurate in the model than advection. Neither of these studies countered the influence of background sedimentation on the results with an added offset term to modelled values as in Weise et al. (2008). It is thought MARE results could be improved by changing the sediment trap design, an approach likely to increase confidence in the model. However, because both FR and SB1 offered ‘very good’ classifications for model fit (for NewDEPOMOD, good for DEPOMOD), it can be inferred that the model results for predicted deposition should be considered reliable at those two sites at least.

The ‘poor’ rating for the sediment trap validation exercise at EL was largely a result of both models overestimating the deposition rate in comparison to the observations. The poor fit was also because the observations were almost constant across the stations within each transect, with the exceptions of the 0 m stations on the southwest and northeast transects in November and January respectively, where there was a noticeable reduction in rates at the 50 m station (~constant thereafter). In contrast, both models predicted a ‘classic’ pattern with a maximum rate at the cage reducing sharply with distance. As resuspension was turned off in the modelled data, this result was anticipated. However, the observations suggest the high currents and deeper bathymetry were not only moving material quickly away from the cage but in a trajectory not captured by the sediment traps. Given the transects were deployed in an east-west direction, greater correlation may have been obtained by placing one transect in a southerly direction, as this was the mean direction of currents during the period of deployment. Alignment between observation and model may also have improved by having sediment traps closer to the pens; however, in at least two of the transects (where observations were constant) this may not have helped.

There are other calibration exercises that would help reduce potential sources of error in the depositional process and that should also be a priority moving forward. The use of model defaults for sinking rates is something that can be improved when developing site specific models. Determining accurate sinking rates for feed and faeces, which are known to change for different size classes of salmon (Reid et al. 2009) is easily done through experimentation. Alternatively, much work has been done on determining settling velocity as a mass fraction of the waste (and not size) which could be used in future work (Bannister et al. 2016). Similarly, the sediment transport model can be improved through more accurate representation of the composition of the sediments (e.g. density, particle size etc.).

Modelled deposition versus benthic impact

Given the benthic response to organic enrichment is known to vary (Cromey et al. 2002; Hargraves et al. 2010; Keeley et al. 2013), there was a need to assess this relationship locally. In this study, modelled deposition provided a reasonable fit with the chosen benthic indices. In southeast Tasmania there was clear effect of distance, with major impacts at the cage largely dissipated at 35 m. This is consistent with the criteria set in Macleod and Forbes (2004). In Macquarie Harbour, the benthic response was also typical of that expected for organic enrichment (see Ross & Macleod 2017), but with some differences in the scale of impact and species.

The impact of low dissolved oxygen (DO) in Macquarie Harbour has been shown to override the benthic response to enrichment. This was evident in this study where at FR (October 2018 survey), bottom water DO was very low (~9%) and no pattern of enrichment was seen. In contrast (February 2019 survey) where bottom water DO had improved considerably (~30%), a clear pattern emerged. Given deposition was lower in the first survey (October 2018) and that by the second survey (February 2019) a considerably greater amount of material is likely to have been deposited on the benthos, it is counterintuitive based on enrichment alone for the benthic condition to be in considerably worse condition in February 2019. The influence of dissolved oxygen in Macquarie Harbour makes interpretation of model results more difficult, which has implications for its use in farm management. Understanding the oxygen cycle in the harbour, response to enrichment in the low DO environment, and recovery of leases post fallowing are integral to interpreting how deposition will affect a lease long term.

The statistical analysis for deposition versus total abundance showed a strong correlation in both regions and across all sites. The correlation between predicted and observed abundance was also strong across all sites and the southeast region. There was a clear gradient with distance from the source in total abundance and the presence of well-known opportunists and indicators of enrichment. That these patterns correlate well with the predicted enrichment gradient is promising given the aim is to eventually derive a relationship between total abundance and deposition that would allow prediction of levels at which compliance would be breached. It should be noted for deposition we used total solids; however, it is carbon in the feed and faeces that will drive local production. The percentage of carbon (in feed and faeces) can shift depending on the feed type. Therefore, it is advisable to establish a threshold for benthic shifts based on carbon content rather than total solids. Percentage carbon (feed and faeces) can be determined for each site and put into DEPOMOD (all versions) as a user defined parameter. However, this was unnecessary for the purposes of this study as we only wanted to establish a relationship between deposition and benthic responses, not thresholds of change.

There was a strong relationship between modelled deposition and sulphides in both regions. Sulphides have been found to vary significantly between salmon aquaculture sites, with currents and production levels accounting for about one third of the variability (Chang et al. 2013). Temperature and dissolved oxygen are also known to affect sulphide levels (Holmer & Kristensen 1992). At SB1 predicted and observed sulphides were not correlated; this may reflect the lease had only recently been stocked. In the first surveys at SB1 there was no relationship between sulphides and deposition that was consistent with organic enrichment whereas in the second survey there was. More data will help to determine the long-term relationship at each site and how farm production levels are affecting this.

The benthic validation at EL showed a strong correlation with both total abundance and sulphides. The choice to use a reduced period of current records (for the benthic validation) could be viewed as arbitrary. Firstly, while a longer time is encouraged, a two-week period is the minimum advised by the developers of DEPOMOD (Cromey et al. 2002). Secondly, this choice was made as NewDEPOMOD did not correlate well using the full data set, which was most likely driven by the high mean bottom currents affecting resuspension processes in the model. The reduced current set produced a depositional pattern more aligned with the mean direction of currents observed in the complete set. If this pattern was more indicative of the actual direction of the deposition, then even if concentration levels were not correctly predicted, they would still be more likely to correlate with the observed pattern of benthic enrichment than the pattern produced using the full period of current data. However, this would mean that NewDEPOMOD is not working well in dispersive environments. This could be improved by calibration of resuspension in NewDEPOMOD in high flow environments. Alternatively, using current data provided by a validated hydrodynamic model, either in NewDEPOMOD or another particle tracking tool (e.g. CONNIE, connie.csiro.au) may be more suitable, as generally the flow field produced is not subject to the type of dramatic shift in current speeds as those recorded by ADCP's.

NewDEPOMOD was developed with improved sediment transport processes including resuspension, which were shown to be unreliable in the original model at dispersive sites with mean bottom currents above 0.095 ms^{-1}. Furthermore, this process cannot be calibrated in DEPOMOD. It can only be turned off which is unrealistic. Until resuspension has been calibrated/validated in NewDEPOMOD it cannot be discerned how accurate this process is, other than to infer from these results that it too fails at more dispersive sites using default values for the parameters controlling this process. As a final note, recent work by researchers using NewDEPOMOD in high-flow environments in Scotland has resulted in the preliminary releases of a default set of parameter values for use in high flow environments (Rebecca Weeks pers. comm.). This set could be further calibrated against observations for use in Tasmanian conditions.

7.2.4.2 Depositional footprints

There was significant variation in the depositional patterns across all sites. These differences were primarily due to changes in bathymetry and hydrography. Changes in farm production level influenced the concentrations within the depositional footprint but did not change the actual pattern of response (i.e., the shape of the footprint). The inherent bathymetry of the site is fixed but the modelled bathymetry can change when cages are shifted around a lease. This was a particular consideration in areas such as Franklin, where there are marked changes in bathymetry within the site.

Current flow has a major influence on the depositional footprint. This was particularly evident at FR where ADCP records from the same time of year and of similar duration, but for different years, provided very different footprints. As previously stated, the DEPOMOD platform proposes at least two weeks of current data should be used but suggests longer time periods would be preferable (Cromey et al. 2002). This study highlights the importance of a longer deployment period. In some locations, comparison between different times of year or across years may be warranted.

Placement of the ADCP is also an important consideration. The ADCP deployments at SB1 were some considerable distance from the southern cage grids and covered two distinct time periods (Jan - Mar and Jun – Oct 2017) in the same year. The resultant dispersal and benthic deposition patterns in this case were very similar, but potentially not representative of the hydrodynamic conditions for the cage grids modelled. It is important that the current data reflect the flow near the cages.

NewDEPOMOD results were shown to be sensitive to the water currents, particularly at FR with the varying bathymetry. Using two distinct sets of ADCP data, both from spring but different years (October 2018 and September 2014) and slightly different locations, altered the deposition patterns significantly. However, at high production the mass and area covered by deposition within the 1500 (g solids m^{-2} y^{-1}) contour and the spatial pattern of the footprint was almost identical for both sets. In contrast, when the two current sets were used to model the footprint at low production, the patterns were quite different. In this case the deposition contained within the 1500 g solids $m^{-2}y^{-1}$ contour was 50% and 70% less in mass and area respectively, dependent on the ADCP record used. In terms of management of farm impacts this result can be viewed in two ways. A significantly higher build-up of enriched matter may cause greater local impacts. However, as a clear relationship between deposition rate and benthic impact remains undetermined, it may be that lower concentrations further from the cage have an impact making it necessary in high flow sites to predict where most sediments are being transported.

One way to address issues with lack of reliable observational (current) data would be to use the currents produced from a hydrodynamic model. Using long time period current data from a validated hydrodynamic model of appropriate resolution should enable more accurate depiction of the hydrography around a lease. The major caveat would be that a fine scale hydrodynamic model needs to exist in the region of interest. There are reliable hydrodynamic models available for both SE Tasmania and Macquarie Harbour (Herzfeld et al. 2010; CSIRO 2019). It should also be noted a recent feature added to NewDEPOMOD is the capacity to use spatially varying current fields as input. However, this was not available at the time this study was completed.

Although both DEPOMOD and NewDEPOMOD produced similar footprints at SB1 and TH, this was not the case at EL. At EL, a deeper site where currents are more dispersive, there was more variation in the outputs. The currents were more likely to result in resuspension; this was particularly evident using DEPOMOD where the resuspension threshold is fixed. This suggests solids were being transported away from the site due to depth of water and currents, but it would be useful to confirm this. Previous studies have shown that DEPOMOD outputs do not always accurately reflect the benthic condition at dispersive sites (Keeley et al. 2013). This is one reason why NewDEPOMOD was created (Black et al. 2016), as it has a greater capacity to adjust the model with respect to sediment transport and resuspension (Black et al. 2012). Obtaining the data with which to define and/ or validate these elements while important, is challenging. Showing modelled sediment transport processes accurately reflect those observed at a site alongside the validation approach used in this study would offer more conclusive evidence that the model is characterising waste deposition accurately.

Finally, it is worth noting NewDEPOMOD would appear to be more sensitive to changes in bathymetry. Although not shown in these results, NewDEPOMOD was more sensitive to changes in depth on the FR and TH leases; this may reflect the improvements in model accuracy of sediment transport. When applying the resuspension module NewDEPOMOD takes account of the actual gradation whereas DEPOMOD assumes the bathymetry is flat (Black et al. 2012). This is important for leases with undulating bathymetry; sediments can potentially sink to deeper water and accumulate, rendering them less susceptible to resuspension and transport.

7.2.5 Conclusions

Depositional modelling software DEPOMOD remains a useful tool for both site selection and management of farm waste in Tasmanian salmon aquaculture regions. It is strongly recommended this tool be calibrated/validated at each site where it is deployed, particularly when moving between ecologically distinct regions. Validation can be carried out through correlation with sediment trap and benthic data. Calibration which would involve choosing model parameters suitable for each site (including empirical measurement of some parameters), will improve results. Future work relating specific deposition rates to ecological shifts within ecologically distinct regions is encouraged as this will help develop more informed management of enriched waste. However, at more dispersive sites it is recommended that resuspension thresholds are determined first, as this will increase confidence in the model outputs; this process is described in NewDEPOMOD User Guide (2020). Finally, there should be some consideration given to how the results of depositional modelling are interpreted in relation to storm events. If, for example, intense wave action or storms are likely in a region, they will potentially disperse deposited organic material, impacting the model validation and thus farm management.

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7.3 Evaluation of dispersal modelling: A non-conservative tracer as a method for determining gradients of dissolved waste emanating from open-cage salmon aquaculture

7.3.1 Introduction

Global farmed fish production now exceeds that of beef (biomass), and the domestication of marine species is occurring at an exponential rate (Duarte et al. 2007; Smeijkal & Kakumanu 2018). International trade in aquaculture species driven by the productivity of large farms, refined regulations and increasing levels of specialisation has led to consolidation of production amongst fewer larger enterprises (Little et al. 2016). One such industry is trade in Atlantic salmon (*Salmo salar*) which has grown at an annual rate of 16% since 1985 and is characterised by a highly standardised coastal water production system (Little et al. 2016).

As production levels and the number of salmon cages in Tasmanian waterways increase, there is growing concern about the sustainability and potential environmental impacts of coastal salmon aquaculture. Key amongst these concerns is the potential effect of nutrient enrichment as a result of waste feed and faeces (FAO 2019). Depending on production levels, open cage salmon farms can input significant amounts of organically enriched waste into a water body, potentially elevating nutrient concentrations close to the farm. Wang et al. (2012) estimated 45% of total nitrogen input as feed to salmon farms in Norway was released back into the environment as dissolved nutrients. Ammonia is the major source of dissolved nitrogen discharged from salmon farms (Dauda et al. 2019).

In southeast Tasmania, salmon farms are generally situated in coastal areas where there is potential for ammonia discharged from the farms to affect adjacent temperate reefs. A key step in assessing to what extent open cage aquaculture might affect coastal reef systems is developing the capacity to accurately predict the concentration of farm derived nutrients and how they change with distance from the cage. This would enable forecasting of where waste might accumulate and/or change over a given time-period, and therefore target monitoring in areas with the greatest predicted impact (Wild-Allen et al. 2010).

Dispersal modelling offers one way of assessing the footprint of dissolved nutrient waste associated with salmon farms. Passive tracers released from a point source and pushed around a model domain by water currents can be used to simulate the dispersal of ammonia from a salmon farm. By attaching a decay-rate to the tracer (i.e. making it non-conservative) losses can be simulated, such as uptake by autotrophs or conversion to other forms of nitrogen (nitrification of ammonia to nitrate), making the model more realistic. This approach is comparatively low-cost and easier to set up than more complex biogeochemical (BGC) modelling approaches which have previously been used to describe impacts of salmon farms and nutrient enrichment in southeast Tasmania (Wild-Allen et al. 2010; Hadley et al. 2018).

Two models have been used in Tasmania to determine concentrations of non-conservative tracers released from multiple point sources and to inform aquaculture planning decisions: the Southeast Tasmanian (SETas) hydrodynamic model (Herzfeld et al. 2010) and CONNIE particle tracking model (Condie et al. 2005). These models are not independent as CONNIE uses current field output from a hydrodynamic model which is a slightly higher resolution version of SETas. Both models provide information on farm derived particle (nutrient) dispersion; however, the parameterisation and calculations in CONNIE have been modified to support an input interface that is both simpler and quicker to use. In this study it is assessed whether CONNIE i) provides an equivalent output to SETas under three different operational farm conditions and ii) improves the understanding of nutrient dispersal.

For model outputs to be useful for management and planning, there are several essential criteria to fulfill. Specifically, the output must accurately reflect: i) spatial distribution (dispersion/ dilution with distance from source), ii) temporal changes (natural influences on dispersion and dilution over time), and most importantly iii) quantum of change/ impact (the actual nutrient load in the system either as total amount or additional input). In addition, there are a number of practical considerations that may need to be taken into account: model simplicity (level of information needed to run the model), ease of use (ability for a non-specialist to employ), and speed (how quickly the analysis and output information can be generated). Considering the quality and reliability of the information obtained, alongside key operational imperatives such as timing and cost, will enable a more informed decision to be made about the benefit of the different modelling approaches.

The CONNIE model provides significant advantages from a management perspective; thus, the main objective of this study was to determine whether CONNIE was a more effective and efficient means to model and determine nutrient distribution than SETas. The analysis was based around how accurately CONNIE predicts 1) the **pattern** (both space/time) of nutrient dispersion and 2) **actual** nutrient concentrations. CONNIE has a higher model grid resolution than SETas and as such is better suited to capturing concentration gradients at a finer scale. CONNIE also has a faster run-time than SETas (minutes compared to days) and is easier to setup and run by non-experts.

We modelled the dispersal of dissolved nutrients (ammonia) emanating from two salmon aquaculture sites in southeast Tasmania using the two models. We compared the results with empirical data to quantify the ammonia gradients and to determine how reliable CONNIE is as a simple and low-cost modelling tool for determining the nutrient gradient and zone of influence around aquaculture leases.

7.3.2 Methods

7.3.2.1 Study site locations

The study was conducted at 3 leases, Storm Bay One (SB1) and Trumpeter (TR) in Storm Bay (SB1), and East of Lippies (EL) at the southern end of the D’Entrecasteaux Channel (Figure 7.3-1). Storm Bay has relatively low depth averaged current speeds (<1 cms^{-1}) with flow patterns controlled by water moving out of the Derwent Estuary and upper D’Entrecasteaux Channel, and some offshore gyres (Herzfeld 2008). However, SB1 is subject to frequent storms and high wave action and is considered an exposed region, with TR more sheltered.

The EL site at the southern end of the D’Entrecasteaux is in a micro-tidal region with surface flows driven by the Huon River. Surface flow from the Huon tends to exit to the north of the channel, and as a result the flow is weaker toward the south where EL is located. In contrast, deeper oceanic water tends to enter the channel from the south, with average currents around the EL site ranging from 0.1 ms^{-1} at the surface to 0.01 ms^{-1} on the bottom (Herzfeld et al. 2010).

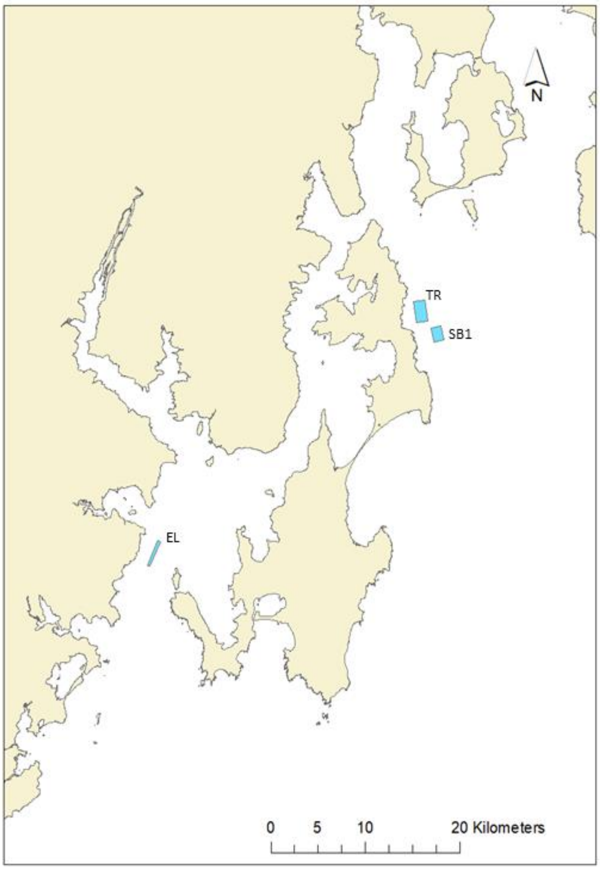

Figure 7.3-1Map showing the EL site (in blue) at the southern entrance of the D'Entrecasteaux Channel. The TR and SB1 sites (in blue) are located off the east coast of North Bruny Island in Storm Bay.

7.3.2.2 Description of models used

SETas

SETas was created using SHOC (Sparse Hydrodynamic Ocean Code, Herzfeld & Waring 2009), a model able to be applied from estuary to regional ocean domains and has been used extensively in studies around southeast Tasmania (Herzfeld et al. 2010; Jones et al. 2012). SETas, an intermediate scale model (Jones et al. 2012), is nested directly into a global model (OceanMAPS product from BlueLink), which resolves the Australasian region at 10 km, and supplies sea level, temperature, and salinity on the open boundaries. The model is forced with river flow from the Huon and Derwent rivers. Gauged river flow data is provided by the Department of Primary Industries, Parks, Water and Environment (DPIPWE). The model uses ACCESS atmospheric products for surface flux specification provided by the Bureau of Meteorology. The SETas model configuration is summarised in Table 7.3-1

CONNIE

CONNIE is an online particle tracking tool developed by CSIRO (Connie 2019) to model dispersion of material released from a point source around a model grid. The model has been used in multiple studies worldwide (e.g. Berry et al. 2012; Milton et al. 2014). The water current (eastern (u) and northern (v) components) field around southeast Tasmania is supplied by the Storm model, a hydrodynamic model with slightly higher resolution (but similar regional extent) than SETas. CONNIE can model passive transport (e.g. dissolved and particulate material), passive tracer decay and more complex particle behaviours (e.g.

swimming phytoplankton). CONNIE uses horizontal currents at fixed layer depths and does not include vertical transport. A model run may be made at one depth only, although there is an option to switch depths at night to simulate behaviour of more complex organisms. However, it is important to note in this study only the results for dissolved material were compared. The model configuration for this experiment is shown in Table 7.3-2.

Table 7.3-1 SETas model configuration

Horizontal grid	Orthogonal curvilinear: 175x120 cells
Horizontal resolution	2000 m (Open ocean), 400 m (upper estuary)
Vertical grid	Geopotential vertical coordinate system: 32 layers
Vertical Resolution	0.5 – 20 m
Minimum depth	2 m
Maximum depth	200 m
Atmospheric forcing	Access R provided by the Australian Bureau of Meteorology
Open ocean boundary forcing	OceanMAPS
Head of estuary forcing	Gauged river flow provided by the Tasmanian Department of Primary Industry, Parks, Water and Environment (DPIPWE).
Time step (dt)	24 s adaptive
Bathymetry	Margvelashvilli et al. (2009)
Vertical mixing	K – epsilon
Bottom friction	Combination of linear and quadratic (Herzfeld and Waring 2009)

Table 7.3-2 CONNIE – southeast Tasmania, model configuration

Horizontal grid	Rectilinear: 501x501 cells (southeast Tasmania grid)
Horizontal resolution	200 m
Vertical Layers	1, 2, 5, 10, 15, 28, 45, 72 m
Vertical Resolution	1 m
Time step (dt)	1 hour
Current field	SETas Model (SHOC)

7.3.2.3 Model scenarios

Both models were set up to simulate the distribution of dissolved nutrients (ammonium) released from open cage salmon aquaculture. Ammonia (NH_3) is created from salmon metabolic processes and released into surrounding waters where it is quickly protonated into a less toxic form ammonium (NH_4^+). Both forms of ammonia nitrogen exist in a dynamic ratio in the water. Salmon excrete NH_3 but once in the water we assume it is quickly converted into NH_4. In this study we use the term “ammonia” to reflect total ammonia nitrogen (i.e. both ammonia and ammonium) in the water. The amount of ammonia released depends on the amount of feed consumed. In this study, 5% of the total feed was assumed to be discharged by the fish as waste nitrogen, with 85% of this waste estimated to be in dissolved form (Wild-Allen et al. 2010). Monthly feed inputs were provided by salmon growers at each site for the periods over which the models were run; these feed rates were converted to rates of ammonia release (mg NH_3 s^{-1}) and used in both models (Table 7.3-3).

Table 7.3-3 Calculated ammonia production rates (mg NH_3 m^{-3}) for each salmon aquaculture operation (study site) for each time period outlined in model scenarios.

	Ammonia (mg NH_3 s^{-1})							
Site	Spring 2014	Summer 2014	Autumn 2015	Winter 2015	Annual 2014/15	August 2016	November 2017	January 2018
TR	13007	1522	0	11686	6554	2310		
SB1	23065	24043	21415	4289	18203	8665		
EL	10600	7408	8530	11210	9437		11249	2491

In CONNIE, 100 particles are released per day (per farm). This was then converted to meaningful ammonia concentrations (particles per cell) based on farm production levels (Table 7.3-3), cell dimensions and depth range at the relevant site. The two-week release period for CONNIE (Table 7.3-4) was timed to coincide with the actual observational sampling events. In SETas, ammonia was simulated using a tracer released at rates shown in Table 7.3-3. In SETas and CONNIE each salmon lease (study site) was represented by a point source at the locations shown in **Error! Reference source not found.**. Model run assumptions are shown in Table 7.3-4.

Table 7.3-4 Assumptions used in parameterisation of the following modelling to assess nutrient dispersion in Storm Bay (SB1 and TR sites) and East Lippies (EL site) using CONNIE and SETas.

Release Period	14 days
Dispersal Time	4-days^{+}
Decay Rate	4 days
Release Depth	SETas - CONNIE comparison 1 m, 15 m - CONNIE 1 – 15 m - SETas CONNIE - Obs. comparison 1 m, 2 m, 10 m
Modelled Period	SETas - CONNIE comparison* 1 – 14 September 2014 (Spring) 1 – 14 December 2014 (Summer) 1 – 14 March 2015 (Autumn) 1 – 14 June 2015 (Winter) Model Validation (Empirical Data – CONNIE) 29 July – 11 August 2016 (Winter) 1 – 14 November 2017 (Spring) 10 – 24 January 2018 (Summer)

$^{+}$ Applies to CONNIE only. *The SETas model was run for the period September 2014 – August 2015 with results for each 2-week period shown in the modelled period used for comparison with CONNIE .

A previous study of connectivity in Storm Bay using a passive tracer and the SETas model showed considerable seasonal variation in dispersal patterns (Herzfeld, 2008). For this reason four release periods (Table 7.3-4) were used in the model comparison scenarios, corresponding to each season; two weeks were assumed to be representative samples of current patterns for each season. The release period was the time over which ammonia was being put into the domain from a lease, which was naturally 2-weeks given the previous assumption. It should be noted CONNIE was run for exactly the time periods shown in Table 7.3-4, whereas SETas was run from 1/9/2014 – 31/8/2015, with results from the nominated two-week intervals used in the comparison. For validation of CONNIE at each site, the model was run for two weeks; one week either side of the date the observations were collected at that lease.

Dispersal time is a CONNIE specific parameter and represents the period ammonia was pushed around in the system after release. The decay-rate represented loss of ammonia over time due to chemical and biological processes. Modelling studies in the Derwent and Huon Estuaries and D'Entrecasteaux Channel have shown the greatest loss of nitrogen in this system is through denitrification (the process that converts nitrate to nitrogen gas). Results from modelling in these regions (Wild-Allen & Andrewartha 2016) suggest ammonia is converted to nitrate in the water column (via nitrification) at a constant rate of approximately 10% per day. The other major sink for ammonia is phytoplankton uptake, which is highly dynamic (both temporally and spatially). However, if it is assumed phytoplankton uptake is having a similar influence on ammonia level to that of nitrification, i.e. 10% each, then the cumulative loss would be 20% per day. It is also assumed 10% of farm derived ammonia (released in the top 15 m) is lost to the lower layers, giving a total loss of approximately 30%, which is broadly equivalent to a 4-day decay rate. It should be noted the dispersal time in CONNIE was chosen to match the decay-rate of ammonia as after 4-days of decay any remaining concentration should be negligible, i.e. nothing to disperse.

In the comparison between CONNIE and SETas, ammonia was assumed to be released between 1 m and 15 m to simulate the range across which it is actually released by fish in the pens. In CONNIE, which uses discrete depth ranges, ammonia is released at 1 m and 15 m respectively, whereas in SETas the release was continuous over the top 15 m. For the comparison between CONNIE and the observations, the model was run with the release at 1 m and 10 m with the results averaged and compared to empirical values collected at these depths. There was also a comparison made between CONNIE and high-resolution observations at EL collected at the surface (2 m), with the model also run at that depth. The release points of ammonia were from grid cells in CONNIE/SETas that contained the centroids of leases (Figure 7.3-2). In the comparison between observations and CONNIE, the release point for SB1 was from the grid cell containing the farm/cage zone (black square right, Figure 7.3-2). The release point for TR was the centroid of the lease. For EL the release point in November was from the cell containing the farm/cage zone (black square left, Figure 7.3-2). As two cage groups were active at EL in January 2018, there were two release points in grid cells containing farm/cage zone and the M3 point respectively (left Figure 7.3-2).

7.3.2.4 Collection of empirical data for model validation

Water samples were collected from around the salmon farms at intervals throughout the study to reflect high and low production levels. Samples were collected from 13 sites at varying distances from the farm at each location; EL was sampled on November 8th, 2017 (high production) and January 17th, 2018 (low production), and SB1 on 4th August 2016 (high production). At each sampling position duplicate samples were collected at depths of 2, 10 and 25 m over a 12-hour period using a Niskin bottle deployed from a boat. The sample sites were in designated zones: high, medium, low, farm and cage dependent on distance from source (Figure 7.3-2). All zones contained 3 points except for the low zone in both regions and the cage zone in SB1, which both contained 1. Sampling depths were designed to align with the model parameterisations as much as possible. All Niskin samples were transferred to sealed plastic containers and analysed for ammonia concentration in an onshore laboratory.

The observed values used in the comparison study with CONNIE, represented the depth averaged concentration (with standard error) calculated from several observations taken at each sampling station over a 12-hour period at 2 m and 10 m.

CONNIE outputs were also compared to empirical data collected at EL in November 2017 by CSIRO with an in-line system the Systea WIZ system. The Systea system allows for direct processing of samples on the vessel at intervals of 13 minutes for NO_x and 21 minutes for NH_4, with the capacity to take one sample every 3 minutes when fully operational. This provides the opportunity for much higher resolution mapping of the nutrient footprint. In this study the system was run over a 12-hour sampling period (the same period described above for the manual sampling with niskin bottles) with samples taken across two transects on each side of the farm; three locations near the cages were also sampled three times over the period (Figure 7.3-7.3-3). Samples were also collected at 500 m and 250 m from the farm on each cardinal direction. All sampling was from 2 m below the surface.

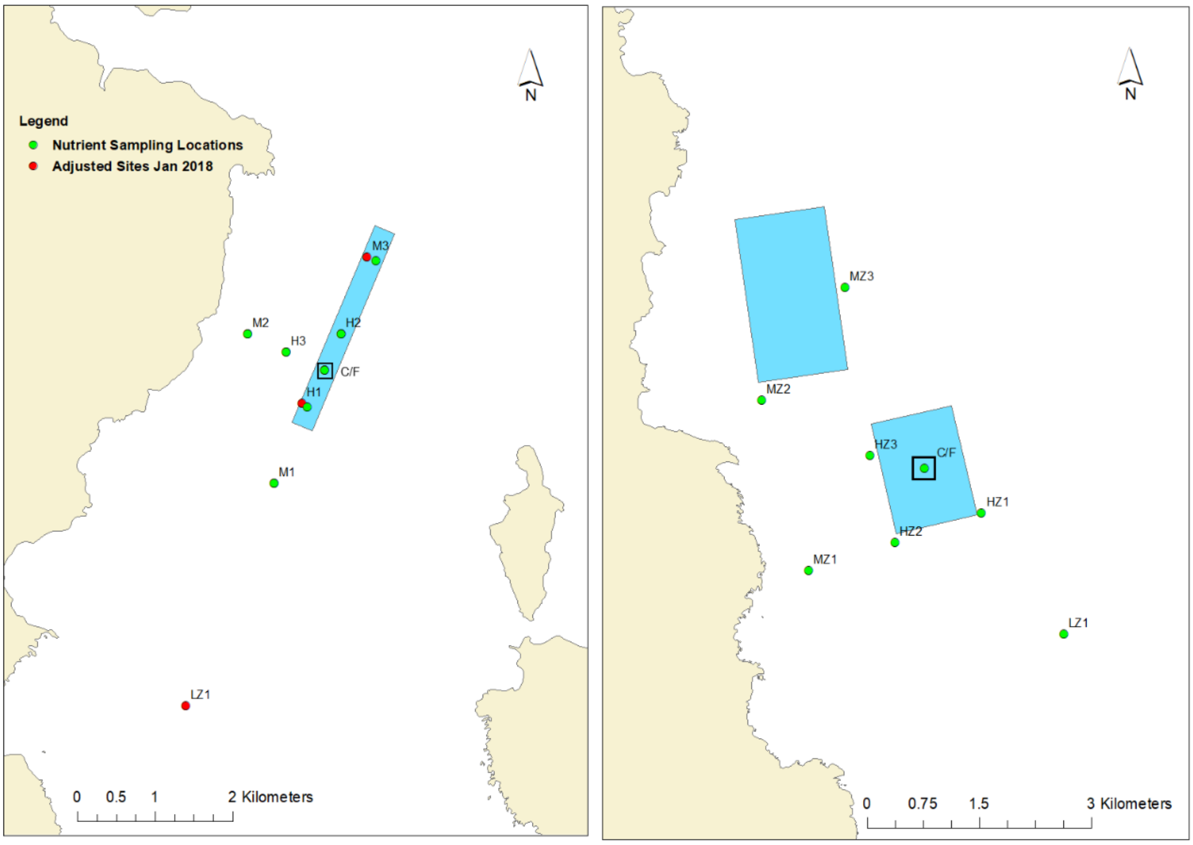

Figure 7.3-2 The sampling sites for EL (November 2017 and January 2018, left) and SB1 July 2016 (right). 'LZ' is low zone, 'MZ' medium zone, 'HZ' high zone and 'C/F' is cage/farm zone.

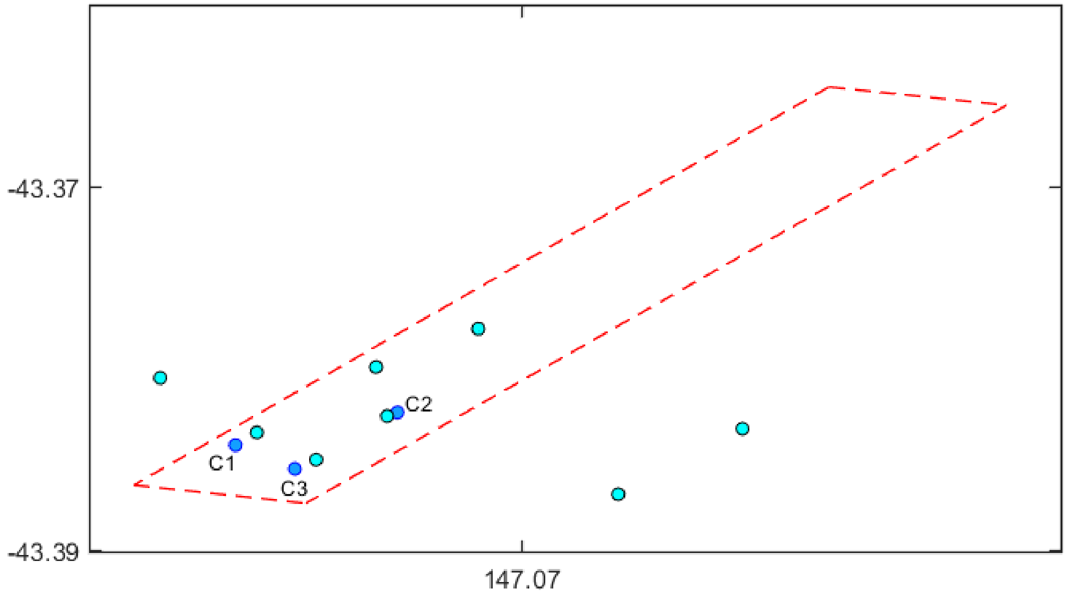

Figure 7.3-7.3-3 Sampling sites (2 m depth) for the Systea deployment (with CSIRO). C1, C2 and C3 (darker blue dots) are the cage locations (sampled three times in 12-hour period) with the other sites (light blue dots) sampled once along transect in alignment with cages grids.

It is important to note the concentrations returned by CONNIE are based on farm derived ammonia only and do not account for background levels, whereas observational data will contain both background and farm inputs. Therefore, an offset equivalent to the minimum observed value, was added to each modelled value to adjust for this whenever modelled and observed data were directly compared. The offset added to CONNIE values was the minimum observation (depth averaged using the 2 and 10 m samples) found from each sampling period, namely 5 and 0.84 (mg NH_4 m^{-3}) at EL in November and January respectively, and 0 mg NH_4 m^{-3} at SB1. For comparison with the CSIRO in-line sampling at EL in November, an offset of 1.26 mg NH_4 m^{-3} was used based on concentrations measured at 2 m. In this comparison study a release depth of 2 m was used in CONNIE and 1/10 of the total amount of farm waste (assuming farm waste is released evenly across top 10 m) to calculate nutrient footprint.

The root mean square error (RMSE) was used to quantify the fit of the model results (at sampling points) to the observations. RSME is an estimate of the mean distance between model and observation and is a method that is known to be sensitive to outliers by giving disproportionate weight to large differences (between observed and modelled values). For the comparison with the CSIRO in-line sampling a polynomial model was fit to modelled and observed ammonia and from this model, correlation between predicted and observed ammonia was determined.

7.3.3 Results

7.3.3.1 Comparison of SETas and CONNIE

Model outputs for ammonia (mg NH_4 m^{-3}) concentrations for the top 15 m of the water column around the SB1, TR and EL leases by both the SETas and CONNIE models are shown in Figure 7.3-7.3-3, Figure 7.3-4, Figure 7.3-5. SETas results for SB1 and TR show that in each of the modelled time periods there was a tendency for nutrients to be advected off the coast in an eastwards direction. There were temporal differences in both ammonia concentration and direction of distribution (Figure 7.3-4), with greater extension of the measurable nutrient plume eastwards and away from the Bruny Coast in March and June (C-Autumn/ D - Winter). The September (A - Spring) and December (B - Summer) data showed more constrained and localised dispersal. The annualised dispersal, calculated from the average of the seasonal footprints (Figure 7.3-4 E), showed the influence from both TR and SB1 leases, where at low concentrations the plumes from the two sites overlap. The SETas model predicted that a low level (< 5 mg NH_4 m^{-3}) of ammonia from the farms may extend inshore and reach the coast of Nth Bruny Island, and in spring and summer higher concentrations are predicted to extend to the coast.

The modelled footprints are a function not only of the prevailing environmental conditions (i.e., water currents) but also reflect the salmon production levels, which also varied over time (Table 7.3-3). The two leases in Storm Bay had quite different stocking regimes: in summer (December 2014), spring (September 2014) and autumn (March 2015) production was greater at the SB1 lease whereas in winter (June 2015) there was more production on the TR lease. In addition, annual average production at SB1 was approximately three times that of TR.

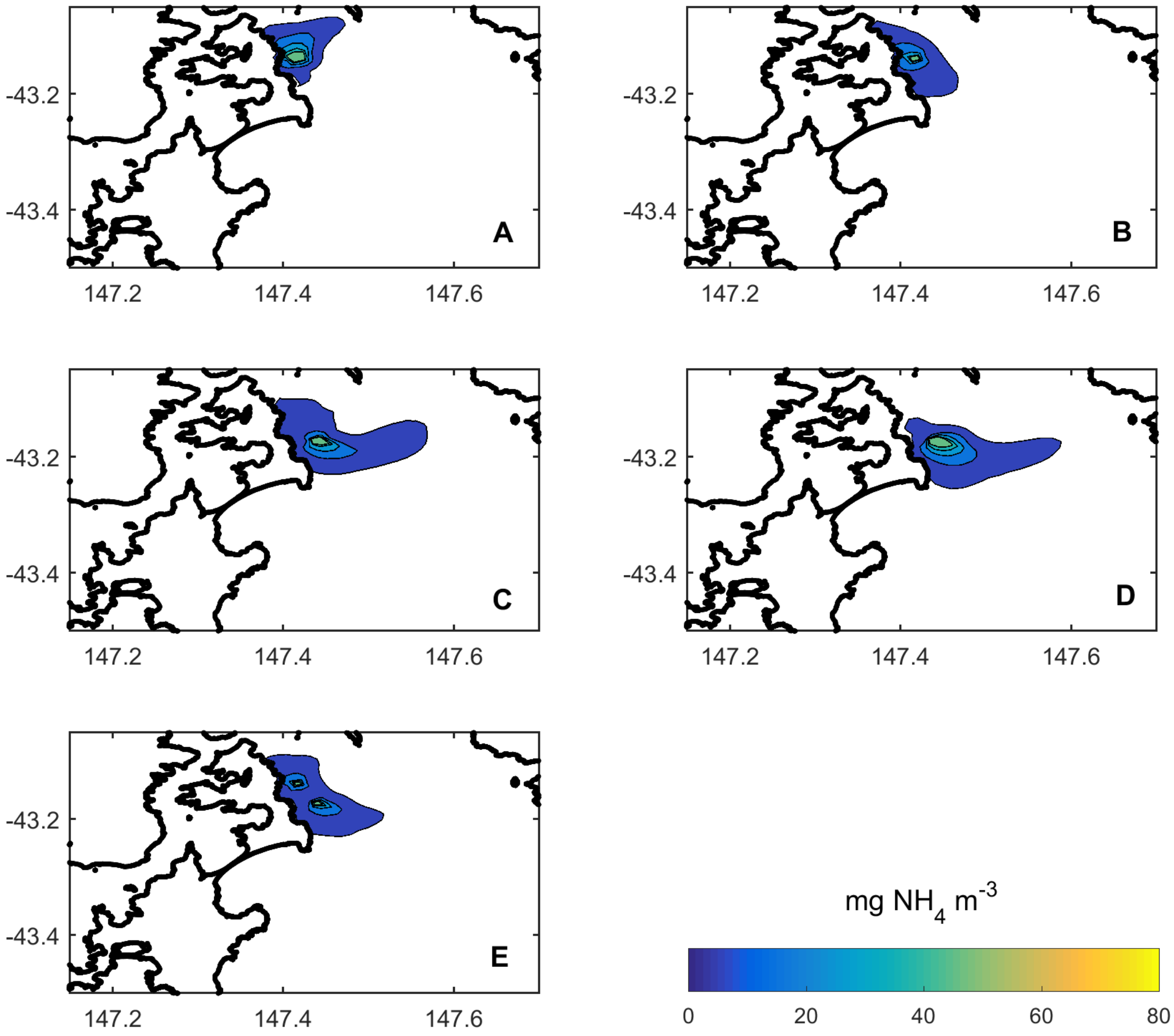

Figure 7.3-4 Average NH_4 concentrations, for the surface 10 m, due to emissions from SB1 and TR using SETas. A, B, C, D are the results from production between 1st – 14th September, 1st – 14th December (2014), 1st – 14th March and 1st – 14th June (2015) respectively. E is the averaged output from seasonal runs and represents annual concentration.

The model outputs from CONNIE for SB1 and TR suggested a similar temporal pattern of response (except perhaps for September) (Figure 7.3-5), but the overall concentrations and dispersion from the source were markedly reduced compared to the SETas results (although the low level nutrient concentrations in the modelled plumes in both model outputs are similar (< 5 mg NH_4 m^{-3})). The CONNIE outputs suggested that it was less likely that farm derived nutrients (ammonia) would extend inshore to the Nth Bruny coast at any time, but that the likelihood of interactions over the December period (B - Summer) were considerably less.

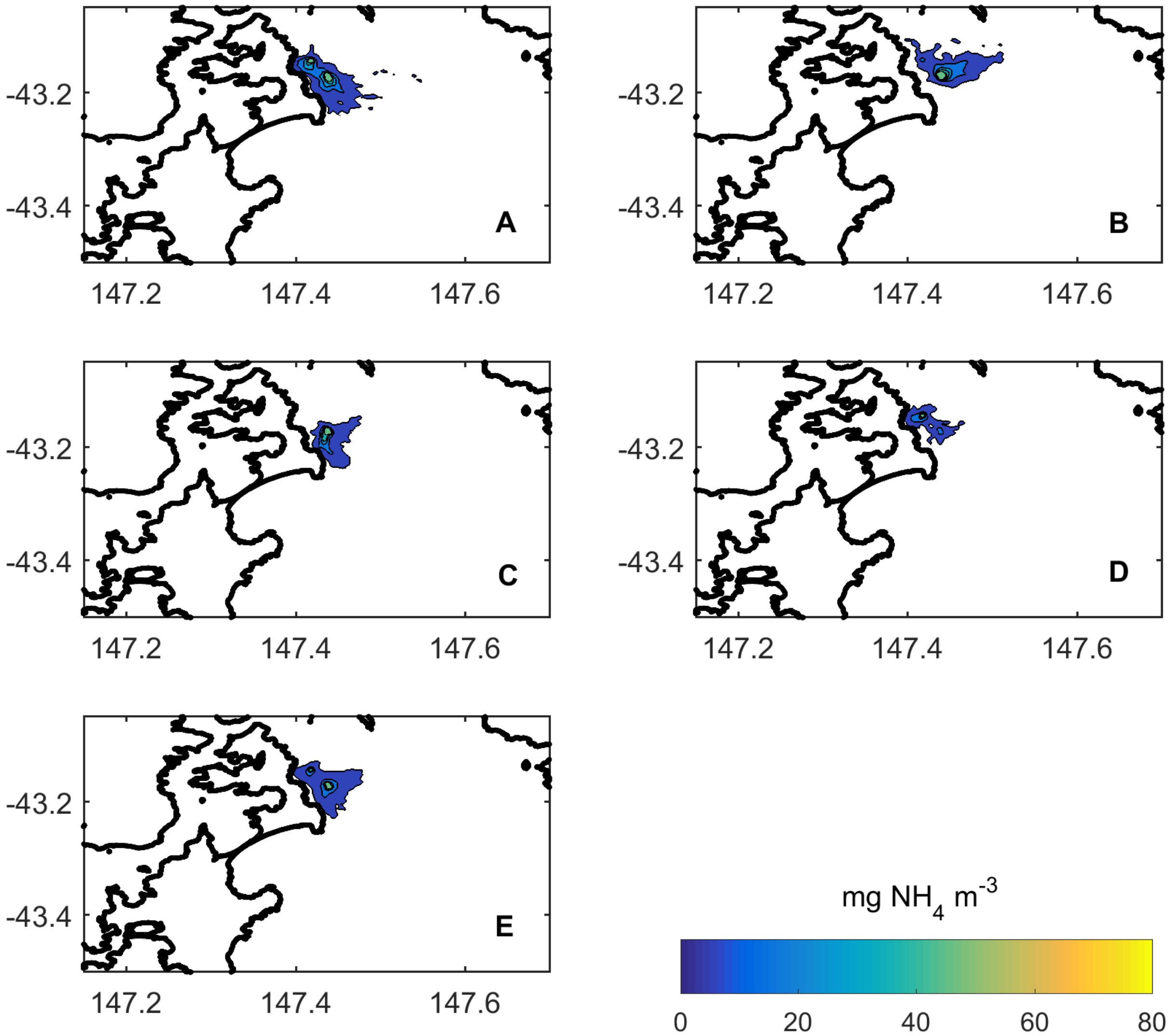

Figure 7.3-5 Average NH_4 concentrations, for the surface 10 m, due to emissions from the Storm Bay One and Trumpeter's leases using CONNIE. A, B, C, D are the results from production between 1st – 14th September, 1st – 14th December (2014), 1st – 14th March and 1st – 14th June (2015) respectively. E is the averaged output from other runs and represents annual concentration.

At East Lippies (EL) the SETas model suggested that farm derived nutrients would generally be advected north and mostly retained within the D'Entrecasteaux Channel (Figure 7.3-6). The spatial variation in this case was almost isometric. The modelled nutrient plume did extend to the coast closest to the farms in all study periods with highest concentrations (>15 mg NH_4 m^{-3}) predicted in September (Spring). As with Storm Bay, although the extent and direction of dispersion seemed to be temporally dictated, the concentration appeared to be more connected to changes in production levels (Table 7.3-3).

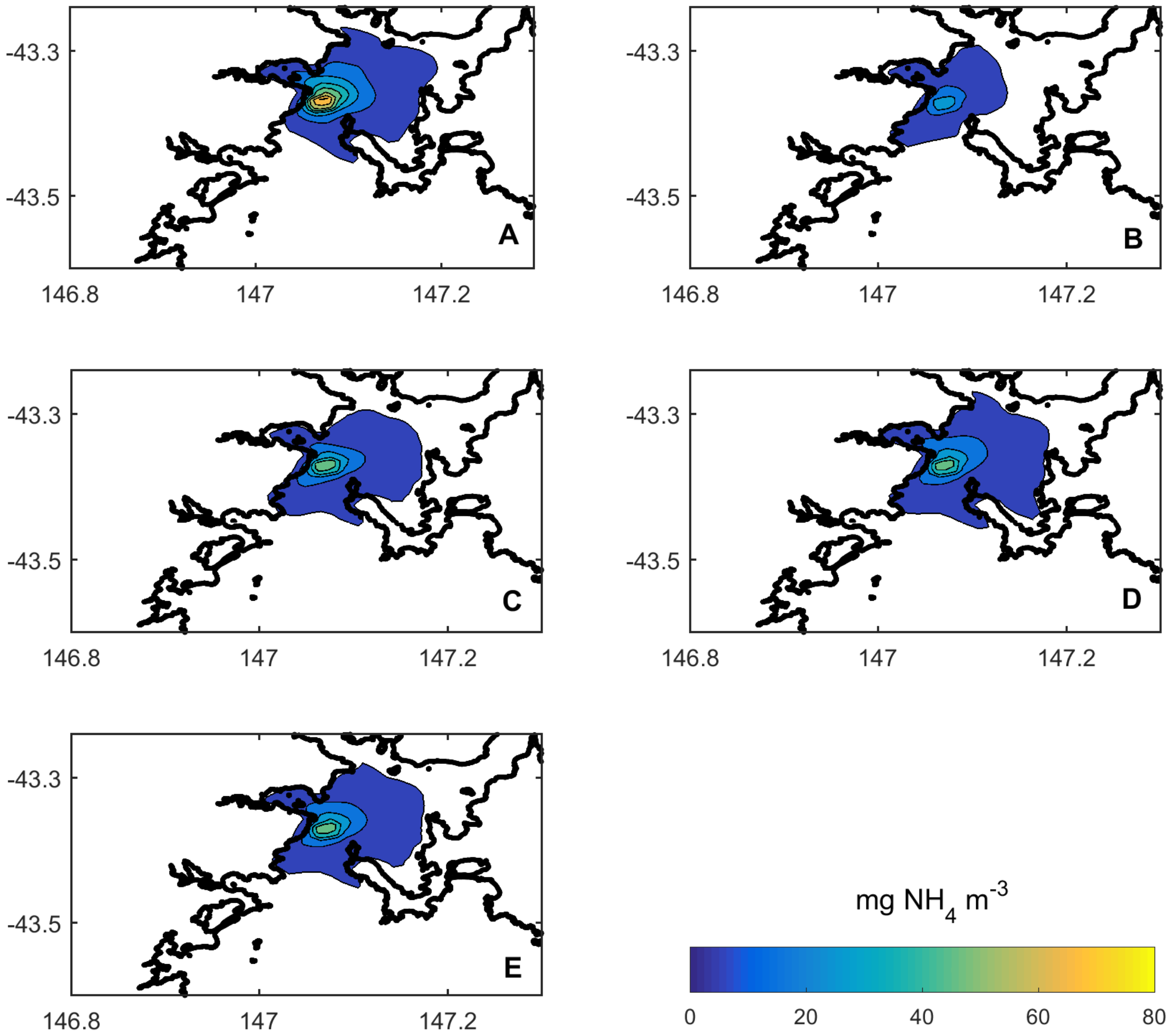

Figure 7.3-6 Average NH_4 concentrations, for the surface 10 m, due to emissions from the East Lippies lease using SETas. A, B, C, D are the results from production between 1st – 14th September, 1st – 14th December (2014), 1st – 14th March and 1st – 14th June (2015) respectively. E is the averaged output from other runs and represents annual concentration.

As for the Storm Bay simulations, the CONNIE model outputs for East Lippies suggested more constrained nutrient concentrations than the SETas model (Figure 7.3-7). Although farm produced ammonia did reach the coast, the concentrations were markedly reduced (< 5 mg NH_4 m^{-3}) in all cases except over the December (D- Summer) time period (Figure 7.3-7).

Comparing the predicated ammonia concentrations (annually integrated) for CONNIE and SETas at the sites used for sampling showed that the SETas model predicted consistently higher concentrations at both SB1 and EL (Figure 7.3-2).

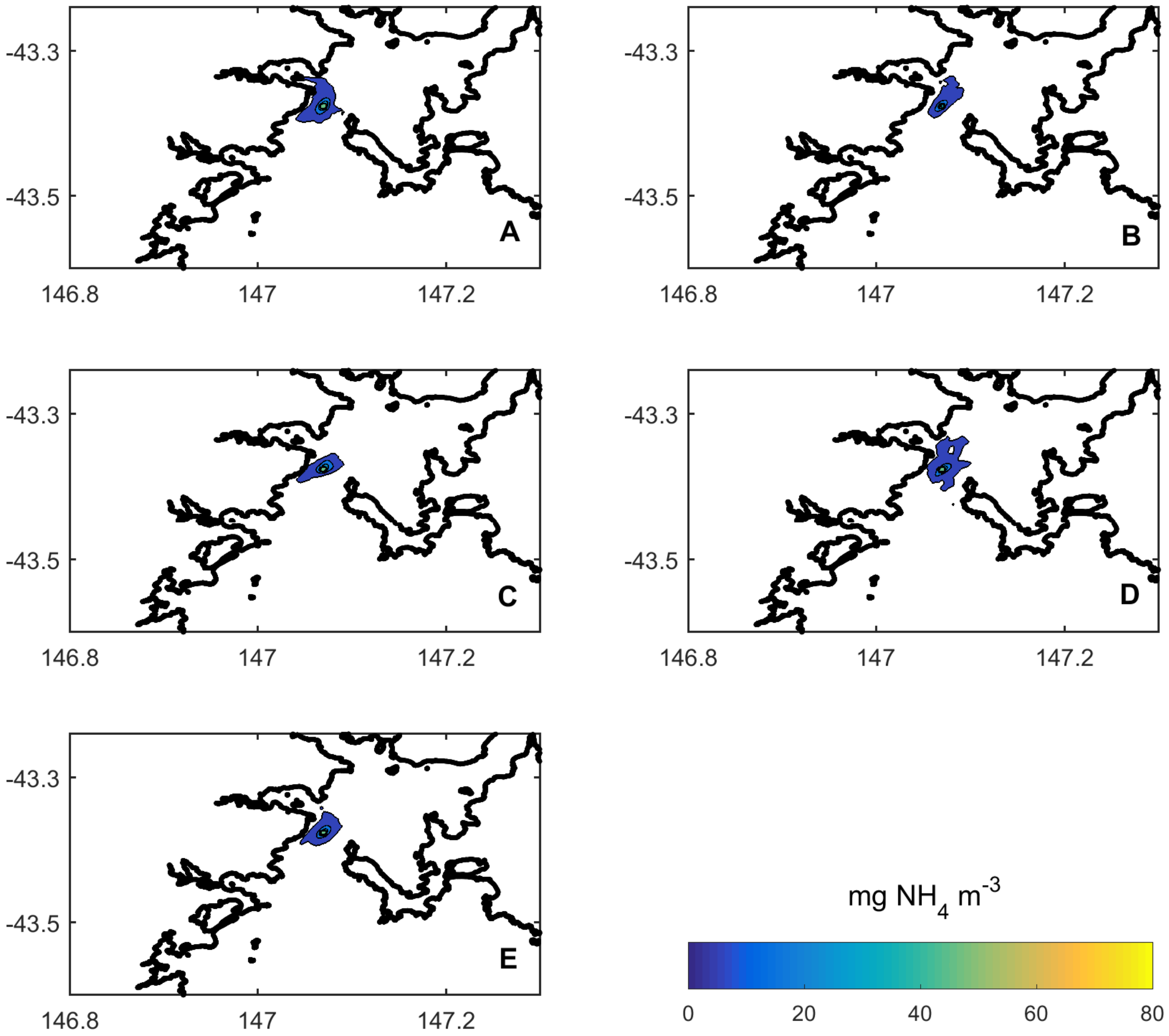

Figure 7.3-7 Average NH_4 concentrations, for the surface 10 m, due to emissions from the East Lippies lease using CONNIE. A, B, C, D are the results from production between 1st – 14th September, 1st – 14th December (2014), 1st – 14th March and 1st – 14th June (2015) respectively. E is the averaged output from other runs and represents annual concentration.

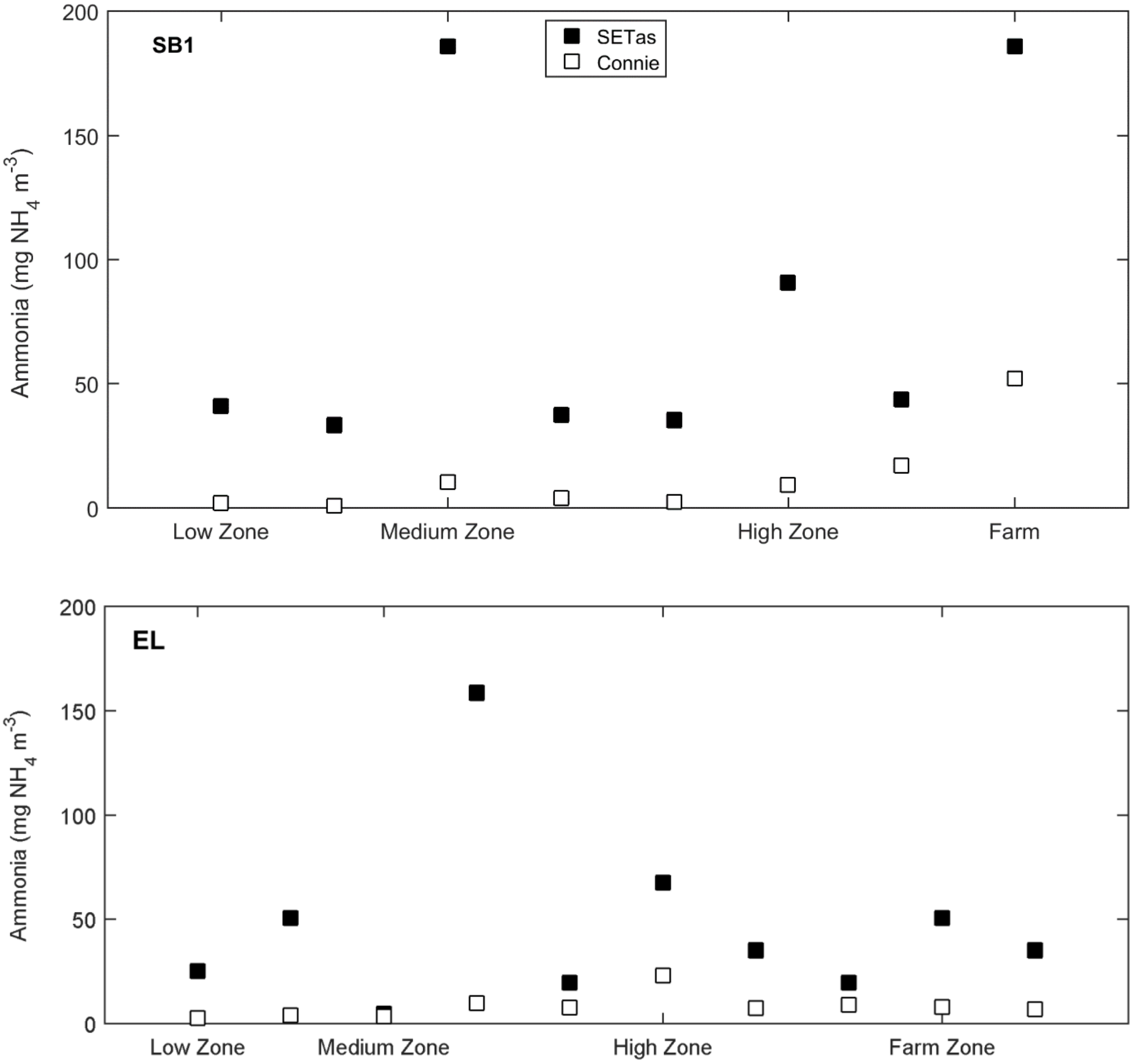

Figure 7.3-8 Comparison of NH_4 concentrations, for the surface 15 m, returned by CONNIE (green squares) and SETas (blue squares) at sampling sites around the SB1/TR (top) and EL sites (bottom).

7.3.3.2 Comparison of the CONNIE model results with empirical data

Ammonia concentrations (mg NH_4 m^{-3}) from samples collected around the SB1 lease on the 4th of August 2016 and the EL lease on both November 8th, 2017 and January 17th, 2018 were compared to CONNIE predictions in the in the model cells that corresponded to the sample sites.

The spatial gradient of ammonia (top 10 m) around EL was calculated based on the average of the model predictions for the period from 1st – 14th November 2017 (Figure 7.3-9 Model). The modelled gradient was relatively isometric around the point of release (Figure 7.3-9). The empirical data collected on the 8th of November 2017 was interpolated onto a 200 m resolution grid for comparison (right Figure 7.3-9).

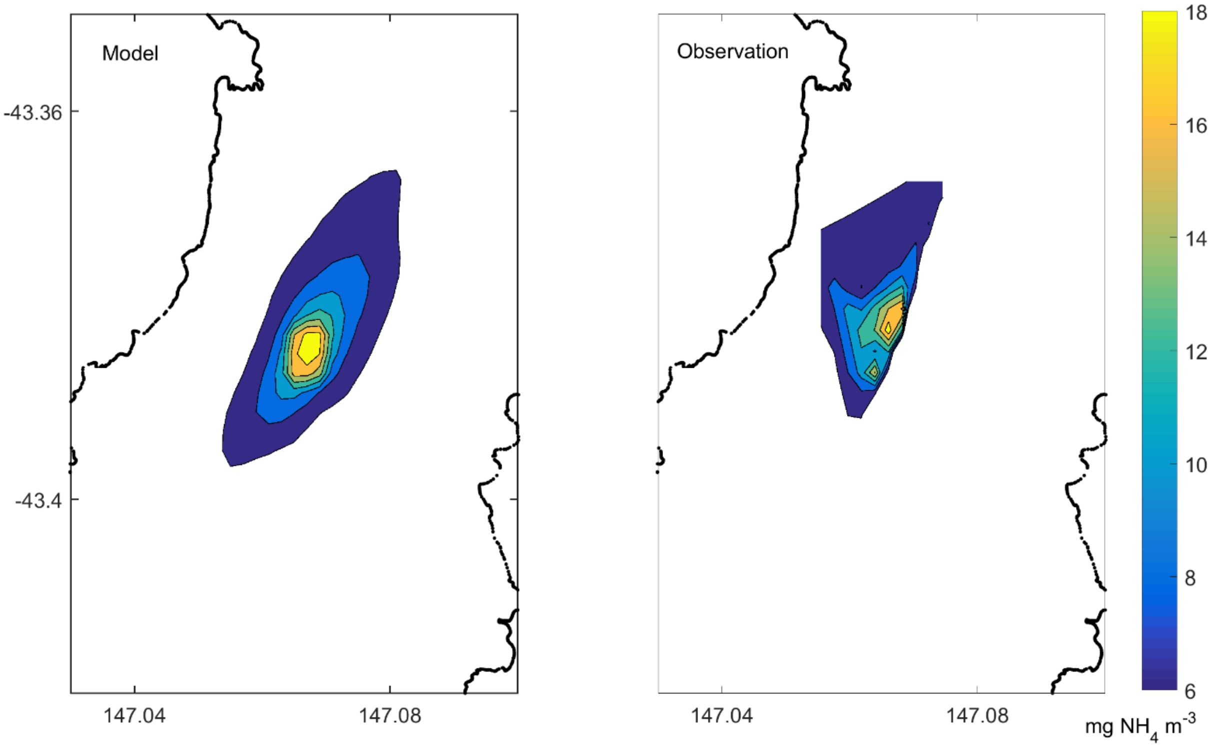

Figure 7.3-9 Spatial comparison between modelled (left) and observed (right) gradient of dissolved ammonia (top 10 m) at EL for November 2017. Model (CONNIE) results display the predicted flux for the period between 1st – 14th of November 2017, based on farm production levels and modelled water current data. The observed data was collected on the 8th of November 2017 at several points around the lease, the data was interpolated onto the same spatial grid as that used in CONNIE.

The observed and modelled patterns were broadly similar, with closer agreement in the higher concentration zones - noting that the observed pattern was an interpolation of 12 data points and the subsequent lack of smoothness accounted for much of the visual difference. Comparing the modelled and observed values directly at each sampling location showed a good match between results (Figure 7.3-10) with a root mean square error (RMSE) of 4. The greatest discrepancies were in the "farm zone".

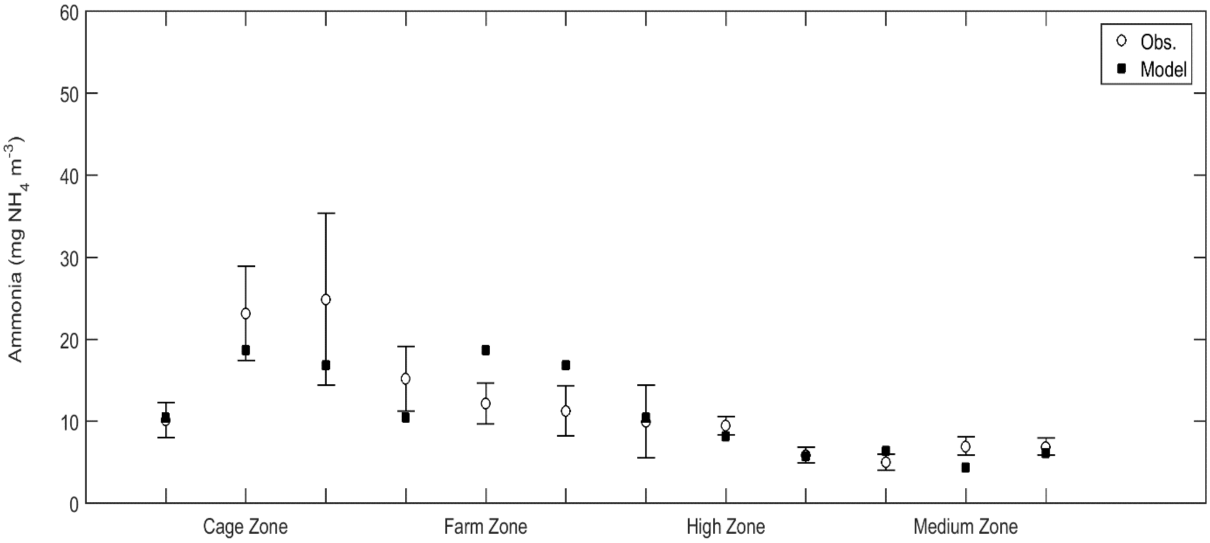

Figure 7.3-10 Mean observations (white circles) and standard error are compared to model values (black squares) at each sample location categorised in zones; cage, farm, high and medium dependent on distance to source (Figure 7.3-2) .

A similar comparison was performed at EL against the empirical data collected on the 17th of January 2018 (Figure 7.3-2 Observed). In this case modelled concentrations were the average of those predicted for the period 10th – 24th of January 2018 (Figure 7.3-11 Model). In this case, the modelled footprint for ammonia was significantly larger, but the concentrations across the footprint were notably lower compared to the November 2017 model run. The two centres of higher concentrations reflected the fact that at this time both southern and northern grids were operating at EL in contrast to November 2017 when only one grid was stocked. Because sampling was mainly around the southern part of the lease, the northern pattern was not as well resolved (right Figure 7.3-11). However, it was clear there were two centres of high concentration, like the pattern seen in the modelled results. A direct comparison between model and observations (Figure 7.3-12) again shows the similarity between them at each sampling station, with both following a similar trajectory although the predicted concentration was a little low at the cage. The RMSE between observations and model was 3.22.

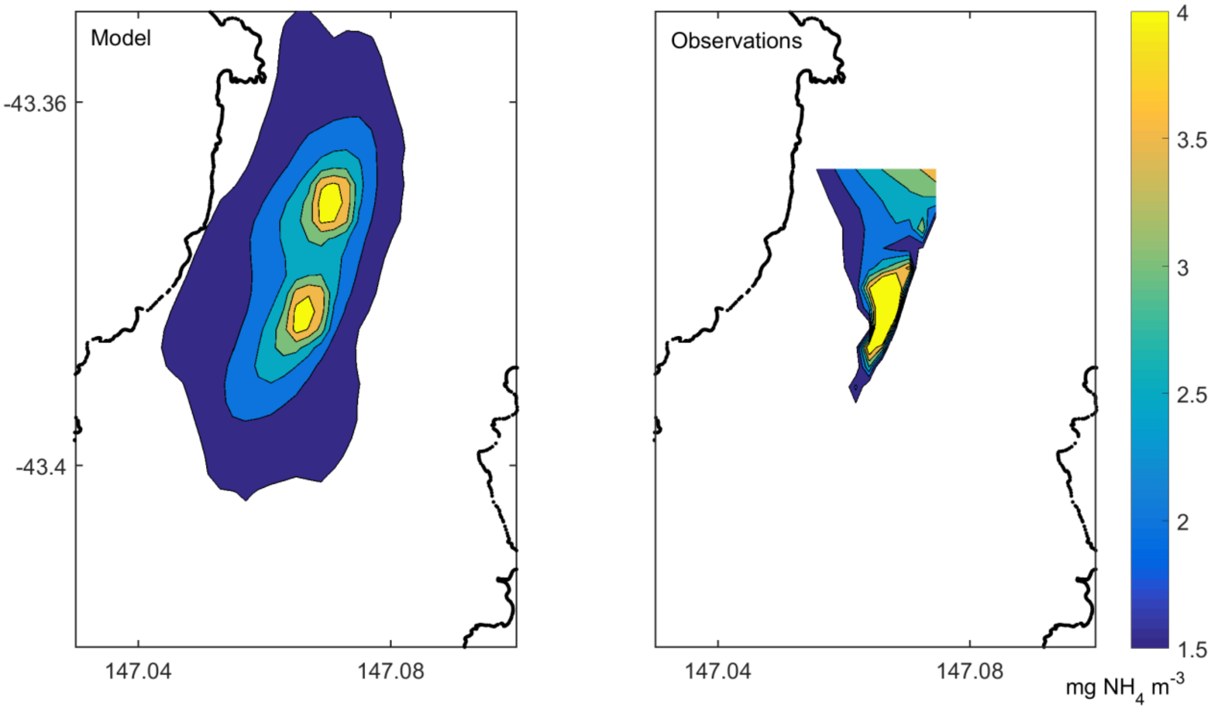

Figure 7.3-11 Spatial comparison between modelled (left) and observed (right) gradient of dissolved ammonia (top 10 m) at EL for January 2018. Model (CONNIE) results display the predicted flux for the period between 10th – 24th of January 2018, based on farm production levels and modelled water current data. The observed data was collected on the 17th of January 2018 at several points around the lease, the data was interpolated onto the same spatial grid as that used in CONNIE.

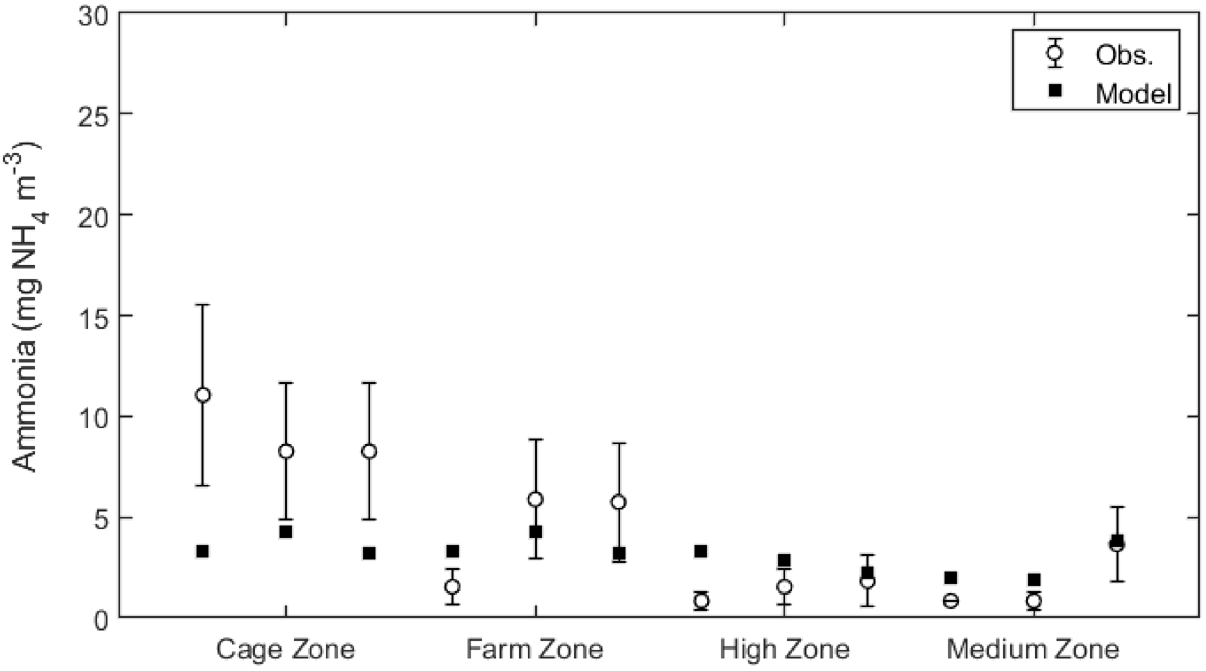

Figure 7.3-12 Mean observations (white circles) and standard error are compared to model values (black squares) at each sample location categorised in zones; cage, farm, high and medium dependent on distance to source (Figure 7.3-2) .

At EL, CONNIE predictions were also compared against the higher resolution empirical data collected at 2 m using CSIRO's inline Systea system on the 8th of November 2017. The empirical data were interpolated onto a 200 m resolution grid (right, Figure 7.3-7.3-13). For comparison, the CONNIE predictions are the average at 2 m for the period from 1st – 14th November 2017 (Figure 7.3-7.3-13 Model). The modelled gradient was relatively isometric around the point of release (Figure 7.3-7.3-13) and the pattern was broadly consistent with the empirical data, although the empirical data suggested more asymmetry. In this instance a polynomial was fitted to the modelled and observed data for analysis; the relationship was well represented by a 2nd degree polynomial (Figure 7.3-7.3-14). Observed ammonia correlated well with that produced by the model (one-way ANOVA, F = 7.71, $p < 0.004$) with R^2=0.48 and a RMSE of 5.05. It can also be seen that >75% of data points were within the 95% confidence interval. Correlation analysis of predicted (from the polynomial relationship in Figure 7.3-7.3-14) versus observed ammonia (Figure 7.3-7.3-15) showed them to be correlated (R^2 = 0.44; one-way ANOVA, F=21, $p<0.0001$; B1 slope, t-stat=4, $p< 0.0006$).

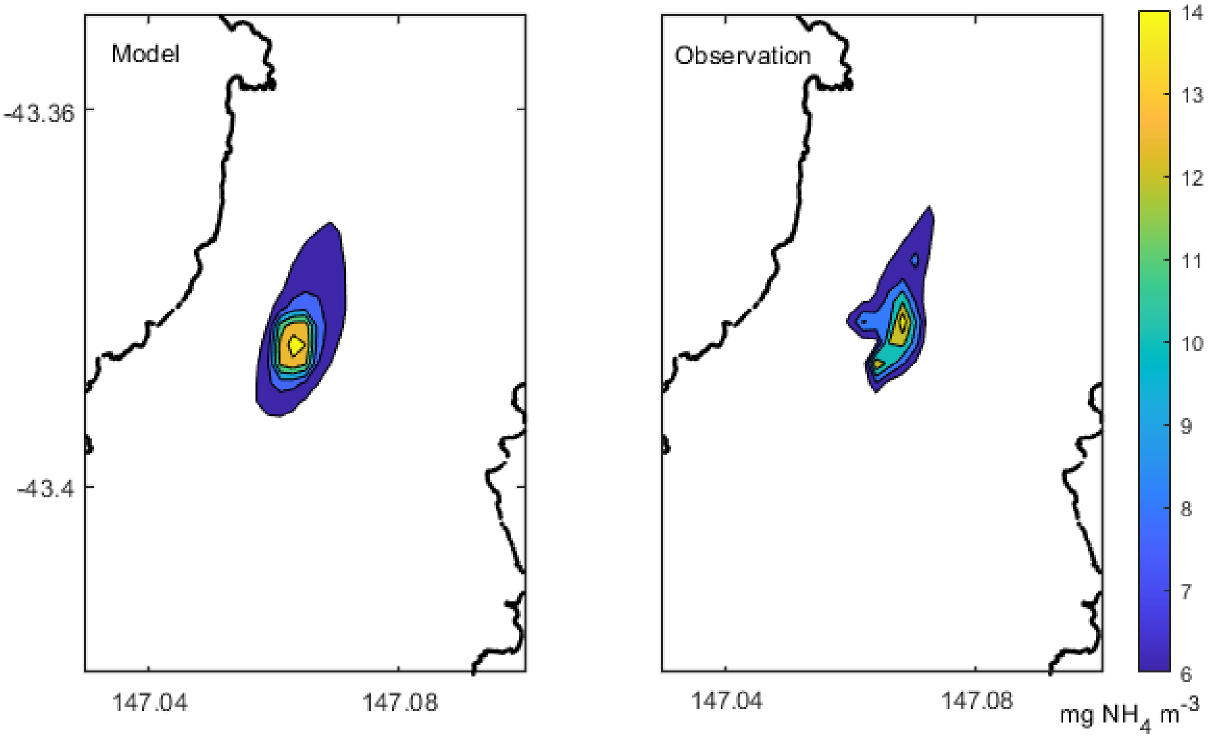

Figure 7.3-7.3-13 Spatial comparison between modelled (left) and observed (right) gradient of dissolved ammonia (2 m) at EL for November 2017. Model (CONNIE) results display the predicted flux for the period between 1st – 14th of November 2017, based on farm production levels and modelled water current data. The observed data was collected on the 8th of November 2017 at several points around the lease using the SYSTEA automated nutrient analyser combined with the manual samples taken at 2 m depth, the data was interpolated onto the same spatial grid as that used in CONNIE.

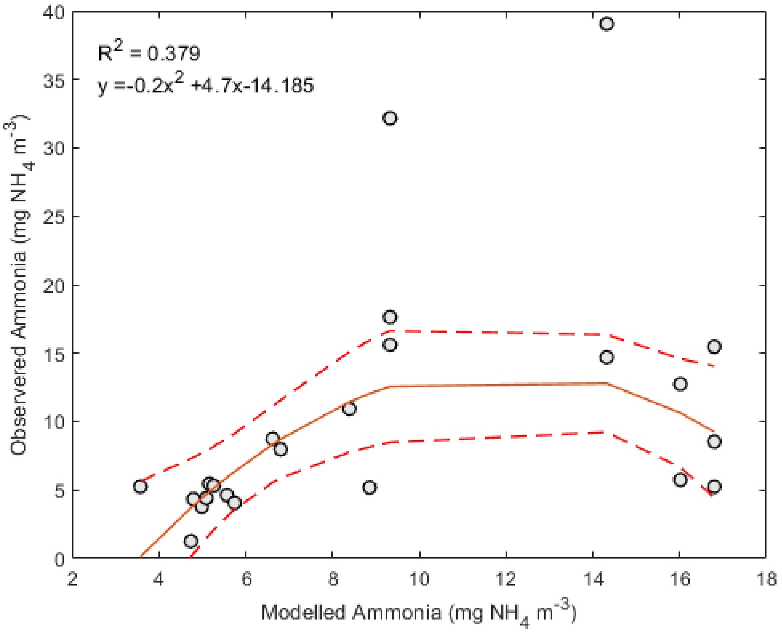

Figure 7.3-7.3-14 Scatter plot of modelled versus observed ammonia around EL (November) at 2 m depth. The regression line (orange) and 95% confidence intervals are also included.

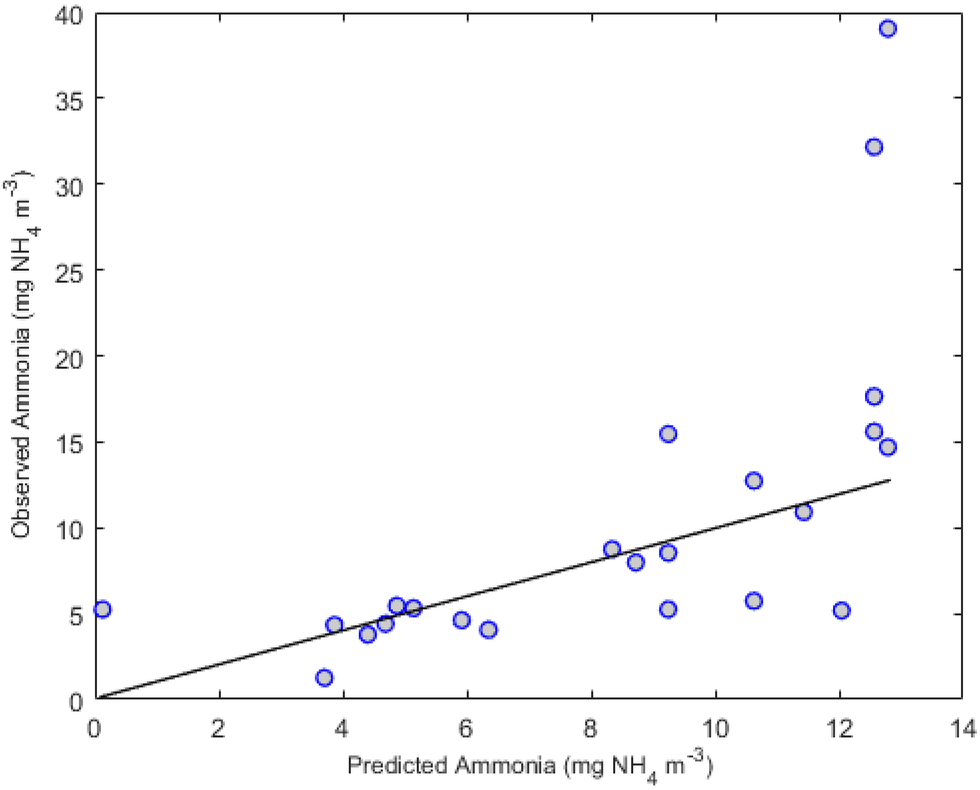

Figure 7.3-7.3-15 Scatter plots for predicted versus observed ammonia for EL (November) at 2 m depth, including regression (black) line.

Finally, CONNIE predictions at the Storm Bay sites (SB1 and TR) for the periods 29 July – 11 August 2016 (Figure 7.3-16 Model) were compared against empirical data collected 4 August 2016 (Figure 7.3-16 Observations).

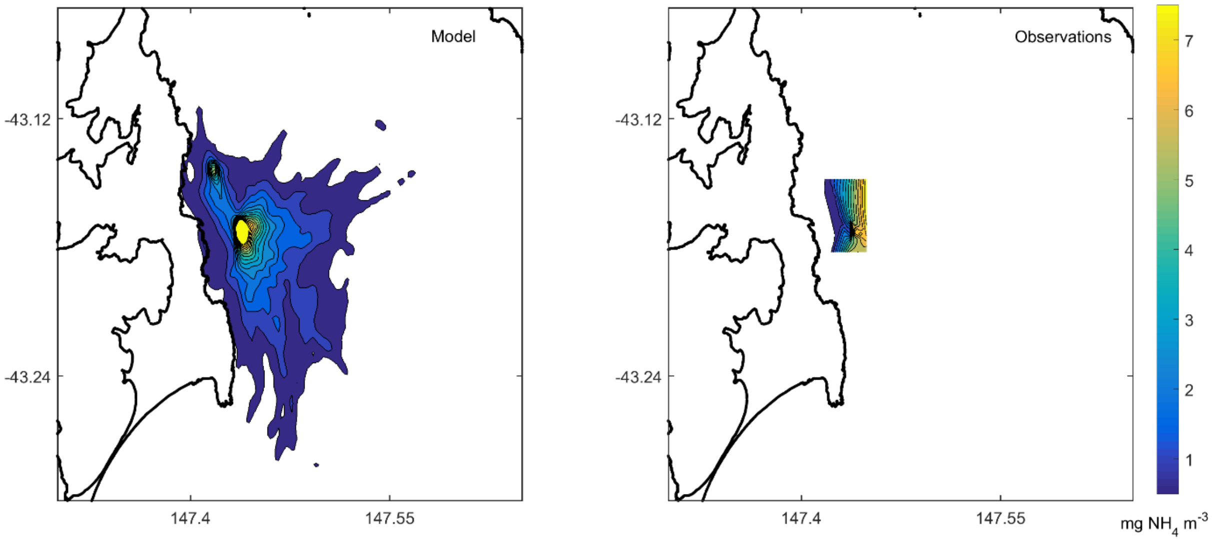

Figure 7.3-16 Spatial comparison between modelled (left) and observed (right) gradient of dissolved ammonia (top 10 m) at SB1/TR for January 2018. Model (CONNIE) results display the predicted flux for the period between 29 July – 11 August 2016, based on farm production levels and modelled water current data. The observed data was collected on the 4th August 2016 at several points around the lease, the data was interpolated onto the same spatial grid as that used in CONNIE.

The modelled output for this period (August 2016) was broadly consistent with that shown earlier (Figure 7.3-4) in that the direction of dispersion was predominately south to southeast. Farm production was low at this time, hence the overall concentrations predicted around the farms were relatively low. The observational data was limited in spatial extent relative to the modelled footprint limiting the scope of the comparison. That said, there was reasonable agreement when considering the steep gradient to the west. More formally, the direct comparison at each sampling site showed greater discrepancy in the cage/farm zone, with closer agreement beyond these points (Figure 7.3-17) and an overall RMSE of 5.9.

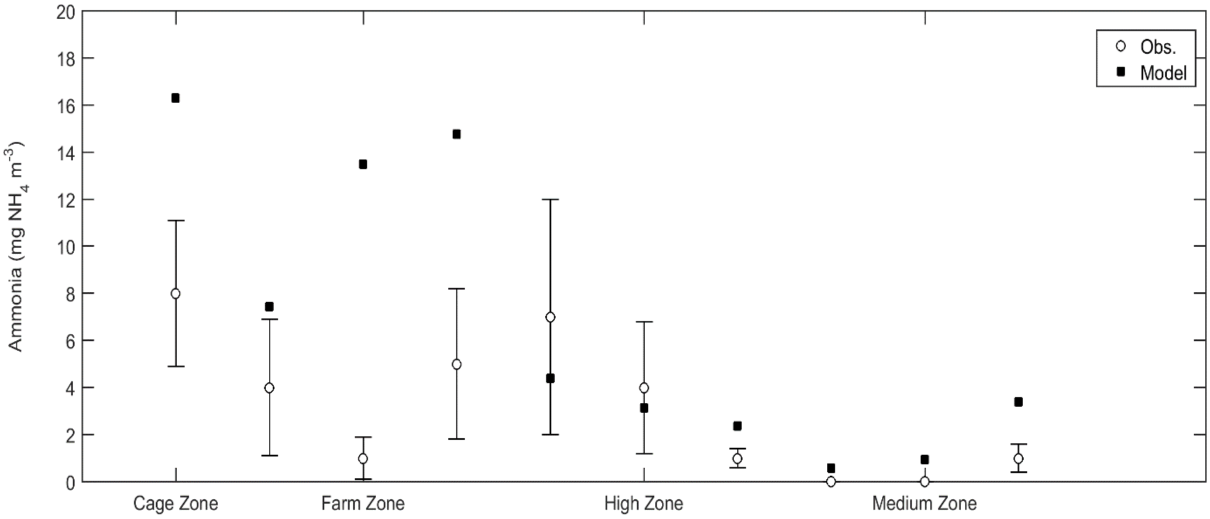

Figure 7.3-17 Mean observations (white circles) and standard error are compared to model values (black squares) at each sample location categorised in zones: cage, farm, high and medium dependent on distance to source (Figure 7.3-2) .

7.3.4 Discussion

7.3.4.1 Comparison of SETas and CONNIE

SETas consistently predicted a larger dispersal footprint with higher concentrations of ammonia than was found using CONNIE. This result was consistent across the two farming regions (El and SB1/TR) for all time periods modelled. As CONNIE uses currents supplied by SETas similar dispersal patterns might be expected. Some of the configuration differences between the models may help explain this result.

Firstly, CONNIE uses a 60-minute time-step in contrast to the 15 minutes used in SETas. If the currents shift appreciably within each hour, CONNIE will not include this information in the final footprint. Secondly, the ‘dispersal time’ parameter in CONNIE ensures all ammonia released in the grid circulates for a constant period from the time of release. Both models were compared over two-weeks of ammonia release; for CONNIE the dispersal time resulted in 18 days of movement of particles compared to the 14 days in SETas. Therefore, particles in SETas released in the last 4 days only decayed for between 0-4 days, whereas those in CONNIE decayed for the full 4 days. CONNIE also incorporates 4 days of current data that is not included in the SETas results. The extra decay time in CONNIE would have the effect of lowering the comparative concentration, whereas the extra flow time is likely to shift the dispersal pattern.

Another difference between the models is the limitation placed on CONNIE to use only discrete current profiles. Particles were released in CONNIE at two specific and isolated depths (1 m, 15 m) and dispersed horizontally by the current fields at each depth. In areas of the model grid where the bottom depth is less than 15 m (e.g. close to the shore), there is no current field and the particles effectively stop. As the final concentration is based on the average over the depth range (15 m), this would result in the concentration being underestimated at these points in the grid compared to SETas, which provides a more continuous range of current flow, including vertical transport between layers.

If we were to evaluate the two modelling approaches based solely on the prediction of nutrient exposure and potential impact, SETas predicted a far greater potential for interaction between the farms and the environment. However, the ammonia concentrations returned by SETas were typically higher than those observed historically in either region (Ross & Macleod 2012; Crawford et al. 2011). That said, it is important to acknowledge that these are relatively new leases, and as such, the period of ambient water quality monitoring is limited. Ongoing monitoring will provide greater insight into the nutrient dynamics around these leases. High resolution ammonia observations like those provided by CSIRO’s in-line Systea system will provide greater confidence in parameter values, such as decay-rate. Similarly, better understanding of the relationship between feed rate and ammonia production inside the salmon pens would potentially reduce this as a source of error within the model.

7.3.4.2 CONNIE validation

Generally, the modelled and observed values were reasonably similar with the RMSE at each site indicating a good fit between model and observation, and visual inspection of the observations versus model in each validation experiment supporting this. However, as mentioned in the previous section, collecting a longer time-series of observations would improve understanding of the model performance and allow for a more specific model validation. Data over a longer time frame would help better determine how the methods work temporally as well as spatially and could reduce the RSME, thus improving model performance and increasing certainty in results.

However, as the aim is to achieve a relatively simple modelling approach to aid management decisions about key environmental risks in a timely fashion, perhaps it is important to consider how much refinement the model needs and whether obtaining more data will actually improve the management outcomes. Whilst more data would certainly help to define and improve the variability in the model it is uncertain what “management” benefit would be obtained from resolving processes that occur on a diurnal scale (e.g. feeding times), for example. Ammonia output is a ‘first order’ impact associated with salmon farming. Consequently, the goal should be to reliably estimate the concentration of ammonia above a particular threshold value/ time frame. Second order information, such as where does the output ammonia go, is much more complex issue, potentially unnecessary for the key management application and more suited to the application of a biogeochemical model. The ANZECC (2000) guidelines, for example, provide threshold levels for ammonia in Australian and New Zealand estuaries which reflect ecological risk. It may be more constructive to frame the limitations and expectations of dispersal modelling around such constraints.

7.3.4.3 Other considerations

Several assumptions underpin this modelling approach. A two-week time-period was assumed to capture seasonal variation in hydrodynamic flow. Whilst this assumption considers the neap-spring tidal cycle observed around these sites, which occurs every two weeks, there may be significant temporal variation due asymmetric factors (e.g. wind, river flow) which may not be captured in the chosen period. If the purpose of the dispersal modelling was to rigorously examine seasonal difference in nutrient dispersal, then this assumption would require more thorough examination in conjunction with replicated seasonal sampling over a longer time period.

Another area where seasonal or temporal difference may be significant is their effect on the decay rate. Decay rate was assumed constant in this study, both over space and time. However, the processes it encapsulates (e.g. loss to phytoplankton and nitrification) are likely to vary temporally (seasonally) in the region (Ross & Macleod 2012; Crawford et al. 2011; Wild-Allen et al. 2010, 2016) and between sites. Targeted research to calibrate the decay rate at a range of sites and over longer or more meaningful timelines could improve how we represent this variable, but we may need to then consider how we incorporate that variability in the model - not a trivial task! Similarly, the offset value used as a proxy for background ammonia rates in the validation of CONNIE needs further verification as clearly this could be a significant source of variation in modelled outputs.

There are, of course, other tools with which to model nutrient dispersion. Biogeochemical (BGC) models for example, have been used in salmon aquaculture and nutrient dynamics research to quantify nutrification and assess impacts at a range of trophic levels. However, a comprehensive BGC model is costly to set up and validate, requires considerable expertise, and generally will take a long time to setup. Furthermore, it may not be necessary if the objective is to determine the risk associated with farm nutrient levels. Clarifying the nutrient level (quantum) that might reasonably be expected to affect a trophic shift and aligning that with an understanding of the inputs (quantum and distance) and likelihood (frequency) with which farm derived nutrients influence the adjacent environment may be all that is required.

Table 7.3-5 Comparing the management requirements and utility of SETas and CONNIE models.

Management Requirement	SETas	CONNIE	Comments (Limitations & Potential for Improvement)
MODEL ACCURACY			
Accurately reflect spatial distribution (dispersion/ dilution with distance from source),	moderate	good	Both models will capture dispersion pattern, CONNIE in current format is closer in concentration.
Accurately reflect temporal changes (natural influences on dispersion and dilution over time)	good	good	Both models capture variability inherent over time in region.
Accurately reflect quantum of change/ impact (the actual nutrient load in the system either as total amount or additional input).	moderate	good	SETas returned higher concentrations than traditionally observed. CONNIE was more accurate, under current configuration.
USEFULNESS			
Simplicity (level of information needed to run the model)	High	Low	CONNIE requires information easily gathered by a user, SETas requires environmental data that requires a level expertise.

Ease of use (ability for non-specialist to employ)	Expert	Standard	SETas requires expertise to setup and run. CONNIE can easily be run from web interface. However, post processing of data is required in both platforms which requires programming skills.
Speed (how quickly the analysis and output information can be generated).	days	minutes	

On the basis of Table 7.3-5, CONNIE is likely to provide a more cost-effective option, and may offer a better alternative for planners/regulators with respect to initial, short-term assessments for near field management decisions, than the alternative SETas model. However, the value of CONNIE would be reduced where assessment of long term, far-field or higher-level trophic interactions is the goal. It is also important to note the post processing of CONNIE data that generated the modelled outputs in this study is presently beyond the scope of the online tool and requires further expertise, but this could be relatively easily upgraded.

7.3.4.4 Conclusions

CONNIE has the potential to be a valuable management tool to help inform decisions regarding the dissolved nutrient footprint of salmon farming. CONNIE can provide valuable management information at a local scale, where first order effects dominate (i.e. farm derived ammonia gradients are detectable), but more complex relationships require more sophisticated and comprehensive approaches. Any model under consideration for the purpose of environmental compliance and regulation will require extensive calibration and validation with real data specifically aligned with the management objectives.

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7.4 Synthesis

7.4.1 Review of Study Background

Modelling environmental interactions with coastal salmon aquaculture in Tasmania has developed significantly over the past 15 years. CSIRO have built biogeochemical (BGC) models for Macquarie Harbour, D'Entrecasteaux Channel and Huon Estuary, and another BGC model is under development for Storm Bay to help assess system impacts from the proposed industry expansion. Furthermore, hydrodynamic models exist for the east coast of Tasmania (ETAS) and Storm Bay (SETas). All these models can be used to examine environmental changes and interactions at the system level over timescales from days to years.

These high-level research models have been developed on the back of major investment, research projects and system wide monitoring programs, and their implementation requires expertise in physical oceanography and/or biogeochemical modelling. As understanding of system response to aquaculture has developed, the knowledge necessary to build simpler decision support tools such as CONNIE and MAREE to help with aquaculture management has been acquired. These tools rely on the outputs and understanding generated by their more complex predecessors but require less expertise to implement and are far less costly to run.

Model development has been actively supported by the Tasmanian government agencies including DPIPWE which have sought to embed modelling approaches within aquaculture planning. However, using these models to plan and manage aquaculture developments relies on them being appropriately validated and calibrated. The commercially available model, DEPOMOD, is used regularly for lease scale depositional modelling to determine the benthic footprint from solid waste exiting fish pens. The developers of DEPOMOD have recently released a new version, NewDEPOMOD which offers improved modelling of resuspension and sediment transport processes. The benthic footprint is required in the Environmental Impact Statement (EIS) when applying for new leases or changes to existing leases, but neither the original or new version of DEPOMOD has been validated for Tasmanian sites. BGC models are generally designed to be most effective at the estuary scale when the dominant biogeochemical and hydrodynamic processes are well resolved. Intermediate scale (sub-grid) interactions are not commonly the focus of these models so these interactions may not be well represented or even included in the models. Simpler approaches, based on the more complex BGC model outputs, have been proposed as useful tools to support management decisions but these have not been validated or calibrated.

7.4.2 Review of Results

In this study, the focus was to validate DEPOMOD/NewDEPOMOD to quantify depositional footprints for Tasmanian salmon aquaculture. This tool provided a good to very good fit with observed deposition rates in two out of the three sites where model validation was applied. The site that returned a poor fit was much more dispersive, with higher average current speeds and deeper bathymetry than the other two. With the exception of sulphides at Storm Bay One (SB1) the model outputs correlated well with measured changes in the benthic data indicating modelled deposition could be used to make predictions on benthic impacts in the regions studied. However, there was no universal threshold value that could be applied across all sites.

The potential for dispersal modelling to provide meaningful information on near-scale interactions and the potential effects of dissolved waste outputs from Tasmanian marine salmon aquaculture was assessed. It was found that dispersal modelling can indeed be a useful tool for predicting the near-field nutrient concentrations from salmon cages. Nutrient concentrations of farm derived ammonia predicted using the particle tracking model CONNIE correlated well to observed data collected around two sites in southeast Tasmania.

Sedimentation rates were measured empirically using sediment traps and the observed deposition data was compared to modelled outputs at the SB1 lease and at the Franklin lease in Macquarie Harbour. The results suggested a ‘very good’ classification for model fit using NewDEPOMOD, and a ‘good’ fit using DEPOMOD. The results at East Lippies (EL) in the D’Entrecasteaux Channel were not as good. At this site, the modelled outputs suggested a ‘poor’ fit compared to sediment trap data, with the model predicting higher deposition close to the cage compared with that seen at distance. The observations, although showing slightly elevated loads at the cage, suggested a more constant pattern of deposition with distance.

East Lippies (EL) was characterised as a dispersive site with higher mean current flow and deeper bathymetry than either SB1 or Franklin. This may have resulted in poor model performance when using default parameter values; however, it is also possible that the experimental design used to collect observations was at fault. The sediment trap transects were deployed in an east west direction whereas the direction of the mean current flow was southerly. Thus, the traps may not have been positioned to most effectively describe the gradient of deposition from the pens. This design could be improved by increasing the number of traps to improve the spatial resolution of observations along each transect and by having additional transects perpendicular to the original direction (i.e., north and south). These measures would help to ensure an accurate characterisation of the gradient of deposition around the pens which could be used to definitively assess model performance.

There are other measures that could be implemented to assist in validating measurements of deposition. For example, identification of quantitative markers specific to farm derived organic matter (i.e., using a marker with a known concentration in feed/ waste material) would provide more accurate observations. This does not just apply to material collected in sediment traps but would also allow direct comparison between organic content in the sediments beneath the farm and that predicted by the model.

Correlation of modelled deposition with benthic indices and empirical measures of benthic condition provides a means to link modelled outputs directly to management objectives and condition measures. The results suggest both sulphides and infaunal abundance were correlated with modelled deposition rates at all sites except for SB1 where the relationship with sulphides was not clear.

This result infers that modelled deposition could be a good predictor of benthic change. However, the benthic validation showed that while benthic enrichment clearly increased as sediment deposition intensified there was no evidence of a threshold deposition level able to be used to universally infer benthic change based on model predictions. Consequently, this relationship would need to be independently determined for each new location.

At SB1, there was considerable variability in sulphide levels with respect to modelled deposition between surveys. The first survey was undertaken when farming had only just begun at this site, consequently dissolved oxygen (DO) levels in the sediment may still have been high as there would be a lag time between deposition of organic matter and increased sulphide production. Sulphide concentrations were very low across all transects in the first survey showing no relationship with deposition. The second survey showed a clearer relationship, and one more typical of a gradient of enrichment. Using a longer time series of sulphide level observations around a lease would help to establish a more robust relationship with deposition.

All measures of organic enrichment will exhibit some variation within sites due to seasonal and inter-annual environmental variability. Similarly, there will also be environmental variability between sites, such as the low DO typically found in Macquarie Harbour relative to southeast Tasmania. This study suggests that any threshold value intended to characterise ecological shifts needs to address this potential variability by employing conservative scenarios aimed at worst case conditions, and the value needs to be location specific. For example, ‘total abundance’ is a key indicator of benthic condition in the current benthic assessments (Ross & Macleod 2012; Ross & Macleod 2017) but varies considerably regionally; Macquarie Harbour is naturally depauperate compared with other farming locations in southern Tasmania. Consequently, the sediments’ capacity to assimilate organic material and respond to changes in organic flux rates would be different. However, it is not just absolute differences in sedimentation that need to be considered; the sediment assimilation processes themselves will be inherently different. For example, DO is fundamental to the assimilation of organic matter in Macquarie Harbour, and this is not only naturally low in Macquarie Harbour but varies considerably interannually which will affect the threshold level at which sedimentation might initiate an ecological (and management) response for this region. The results suggest modelling could potentially be used to infer local scale ecological responses based on depositional inputs which in turn can be used to inform decisions on management of depositional waste. However, these responses varied both between and within locations and as such the modelling would need to be independently validated for each region, and perhaps even site within region. Consequently, further analysis and validation would be required before threshold levels could be reliably proposed.

Default parameter settings for both DEPOMOD and NewDEPOMOD were used in all model runs except when comparing with the sediment trap data (total deposition) where resuspension was turned off. For the most part, the default settings characterised the depositional conditions quite well. The modelled fluxes at Franklins and SB1 provided a very good fit with observed fluxes, with no need for extensive calibration of depositional processes within NewDEPOMOD. The D’Entrecasteaux Channel (EL) site was the only lease where modelled fluxes were not well correlated with observations. As previously stated, that result may have been due to how the observations were collected. However, NewDEPOMOD has been shown to perform poorly in high flow environments using the default values for the sediment transport processes as those values are generally not representative of these conditions. These environments require a calibration of parameter values to tune the model output to fit with observed data. A similar result was achieved by using a shorter time series of current data (hydrodynamics) at EL, where the mean current speed was more in line with that of a depositional site. In this way it was possible to achieve a good correlation between the model output and benthic indices. However, to assess model accuracy in dispersive conditions a calibration/validation exercise is necessary. This would involve tuning those parameters responsible for sediment transport including resuspension; however, this should not be done arbitrarily as each parameter represents an actual physical process. For example, critical shear stress is an important factor used to parameterise resuspension in NewDEPOMOD. This parameter determines the velocity at which particles will resuspend and is very much dependent on sediment composition. A better understanding of this factor for the sediment types found around aquaculture sites in Tasmania would potentially reduce uncertainty within the model. It should be noted, there is work presently underway to validate NewDEPOMOD in high flow areas (Rebecca Weeks, pers comm) in

Scotland and these results can be used to develop a default parameter set for use in high flow environments.

Given the observations above regarding the sensitivity of the models to the underpinning hydrodynamics, it is clear reliable current data is critical for depositional modelling. The developers of DEPOMOD propose a current record of 3-6 months to adequately capture variability in the system. In the absence of reliable empirical data, it is possible to use current fields provided by calibrated hydrodynamic models in place of the spatially invariant current field used in NewDEPOMOD. Modelled current fields may be particularly appropriate at sites like Macquarie Harbour where water flow can vary markedly across relatively short distances and timeframes due to bathymetry, river flow and oceanic inputs, but less important at locations (e.g., Storm Bay) where the currents are typically more uniform and predictable.

In general, the modelled dissolved nutrient (in this case ammonia) concentrations showed strong correlation with observations. Storm Bay was not as well correlated as the other sites, but it was the site at which the technique was first tested; the approach was improved over the latter two surveys at East Lippies. The increased spatial replication in subsequent empirical sampling provided a more accurate representation of concentrations within the lease. Consequently, if the Storm Bay validation was repeated, using the improved sampling techniques, it is likely to show greater correlation. It is proposed dispersal modelling can potentially be used in a similar manner to depositional modelling to inform both site selection and location of environmental monitoring sites, and that water quality data currently collected across various monitoring programs could be used to calibrate and validate the modelling.

CONNIE was assessed as being more accurate than hydrodynamic modelling alone in reproducing near scale nutrient concentrations around salmon farm leases. CONNIE offers a relatively low-cost method to assess dispersal of dissolved waste, potential for interaction, and management at distances of 1-2 km, which is a scale not as well understood using existing BGC models. Correlating the concentrations predicted through dispersal modelling with observed changes in the near-field ecology could increase the value of this approach as an ecological management tool. The method could be used to predict footprints of dissolved nutrient enrichment around a farm, using concentration levels as guides for ecological impact, similar to the way depositional modelling is used for solid material. This approach could be used to explore production scenarios. Furthermore, the ‘decay rate’ could be investigated to produce a more accurate measure of the variability in rates of biochemical loss both spatially and temporally across marine sites. Scenarios that use ‘decay’ can be contrasted with those using ‘no-decay’ to examine areas where farm waste(s) might potentially accumulate. This could then be used to better inform management decisions with respect to the potential for adverse interactions and remedial actions.

Dispersal modelling can clearly offer a low-cost alternative to BGC modelling, which is often more suited to larger scales of interaction and may not always contain the key processes of interest to management (e.g., threshold outputs and potential for reef interactions). However, establishing the relationship between dissolved nutrient concentration levels and ecological impact would be key to using dispersal modelling as a reliable management tool. This study has demonstrated the potential of this method to quantify dissolved nutrient footprints around salmon farms and establishing a benthic relationship. The approach now needs refining on a case-by-case basis.

7.4.3 Summary of Recommendations

- Dispersal modelling can indeed be a useful tool for predicting the near-field nutrient concentrations from salmon cages. Nutrient concentrations of farm derived ammonia predicted using the particle tracking model CONNIE correlated well to observed data collected around two sites in southeast Tasmania.
- We propose that dispersal modelling can potentially be used in a similar manner to depositional modelling to inform both site selection and location of environmental monitoring sites, and that water quality data currently collected across various monitoring programs could be used to calibrate and validate the modelling. However, the approach can be improved by constraining the decay rate, a proxy for nutrient loss.
- Modelled deposition is a good predictor of benthic change. However, the benthic validation showed that while benthic enrichment clearly increased as sediment deposition intensified there was no evidence of a threshold deposition level be used to universally infer benthic change based on model predictions. Consequently, this relationship would need to be independently determined for each new location.
- Depositional modelling can be refined further with a more suitable choice of default parameter values, particularly in dispersive sites where resuspension needs to be properly calibrated.

8 Overarching Conclusions, Recommendations & Implications

As conclusions and recommendations have been included at the end of each of the individual sections (Local Scale Monitoring, Reef Interactions, Modelling) we have sought to provide a more overarching view of the key project findings and outcomes in this section.

In that context the overarching conclusion with respect to local scale monitoring would be that, for the most part, the existing monitoring guidelines specified in Macleod and Forbes (2004) do still provide a reasonable understanding of the local scale environmental conditions (impact and recovery), associated with current farming practices based on substrate type. Both the biotic and abiotic parameters currently used for environmental monitoring of salmon farms remain sensitive and fit for purpose. However, there were some areas where it was proposed that improvements could be made given the inherent variability in environmental conditions and response to enrichment in both space (within and between regions) and time (surveys).

Key amongst these is that environmental standards should focus on change relative to baseline and reference conditions rather than rely on fixed parameter ranges. As such, the study reinforces the importance of establishing robust baseline conditions and ongoing monitoring of reference conditions. We also recommend that reference locations should be further from the farms than previously considered given the larger footprints evident from this study and that further consideration be given to the level of change that defines ‘unacceptable’ when measured at compliance locations 35 m from the lease boundary.

The visual assessment methodology frequently used to inform management was found to still be a robust measure of sediment health. However, evaluating the full suite of criteria proposed by Macleod and Forbes (2004) provides a more holistic and informative measure of sediment health than the reduced set of criteria currently used. were used. There were also clear differences between regions in the visual assessment condition criteria such that some of the newer sites/ regions, such as Macquarie Harbour, required adaptation of the guidelines.

From a farm management perspective, it was clear that fallowing remains important in facilitating the recovery of the system and that where cage footprints are extensive or particularly pronounced, that consideration should be given to fallowing at the scale of the entire cage grid rather than pen by pen, particularly in a system like Macquarie Harbour due the influence of variable water dissolved oxygen concentrations.

The most significant conclusion with respect to the reef assessment is that that there is potential for interactions between salmon farming and adjacent reefs. Importantly, the study identified indicators that can alert us to adverse changes in reef biodiversity and function, and which provide an early warning of deterioration or loss of resilience in reef systems.

This study recommends adopting a monitoring program that combines regular but less frequent (e.g. every 5-7 years) Edgar-Barrett biodiversity surveys with more frequent (e.g. biannual) RVA assessments to target specific measures of reef function. Surveys should be undertaken using both methods before the commencement of farming to establish baseline conditions. Given the inherent variability in the system, baseline as well as ongoing surveys should be conducted at multiple sites at varying distances from farms.

It is important to note that determining causality, even with the aforementioned monitoring approach remains challenging, and will ultimately rely on a weight of evidence approach based on a detailed understanding of the local environment, the broader regional and global pressures, and the inherent characteristics of the reef community itself.

Modelling environmental interactions with coastal salmon aquaculture in Tasmania has developed significantly over the past 15 years. This study clearly demonstrated and validated the value of dispersal modelling as a means to predict near-field nutrient concentrations resulting from salmon farming and showed that dispersal modelling can be used in a similar manner to depositional modelling to inform both site selection and location of environmental monitoring sites. The study showed that modelled deposition is a good predictor of benthic change but was unable to identify a threshold deposition level that could be universally applied to infer a specific level of benthic change, noting that this relationship would need to be independently determined for each location. The study also noted the need to be mindful of default parameter values, particularly in dispersive sites where resuspension can have a marked effect on the depositional footprint and therefore needs to be carefully calibrated.

The implications of this research are many; but critically the findings provide both reassurance that the benthic impacts of salmon farming can be reliably monitored and measured in all farming regions, and guidance on how best to use this understanding to manage those interactions sustainably. Whilst there is still more research needed to fully understand the interaction between reef ecosystems and salmon farming, this study does provide a means to monitor for critical changes in reef condition in areas where salmon farming co-occurs. We hope that this study will provide the knowledge necessary to support any changes to both farm and compliance monitoring and help to ensure sustainable management of salmon farming in Tasmania.

As noted above, please refer to the individual sections for detailed recommendations.

9 Extension and Adoption

This has been very broad uptake of the outcomes from this project, with wide extension and adoption in several key areas.

There have been numerous extension and adoption activities associated with FRDC 2015-24. Outcomes of the local scale and reef monitoring components of the project have been widely adopted by government. All new lease areas now include reef monitoring in their Broadscale Environmental Monitoring Program requirements, with the Rapid Visual Assessment (RVA) technique developed through this project incorporated into Environmental Licenses for Storm Bay. IMAS has also been actively involved in providing advice to the EPA for the drafting of the Environmental Standards for monitoring of finfish aquaculture in Tasmania; this includes the drafting and inclusion of methods for both biodiversity and RVA reef surveys and a review and update to the methods used to assess sediment conditions.

RVA survey techniques have also been adopted by other stakeholder groups. This includes the Derwent Estuary Program (DEP, see report) and more recently the Tamar Estuary & Esk Rivers Program (TEER) are looking to adopt remote techniques for monitoring reefs in the Tamar estuary. The Tasmanian Abalone Council (TACL) also motivated for a follow-up study in the D'Entrecasteaux Channel that was subsequently commissioned to a) build a longer time series of data for RVA monitoring and b) assess the broadscale health of reefs in the southern D'Entrecasteaux Channel (see report).

In Macquarie Harbour the observations from this study were pivotal in driving significant change to management and industry practice in Macquarie Harbour.

Further extension and adoption of the outcomes from this project include Advice Notes to Marine Farming Branch (MFB) on depositional and dispersion modelling e.g. a review evaluating the latest version of the depositional software NewDEPOMOD and a CONNIE based evaluation of the nutrient dispersion footprint under different biomass scenarios in Storm Bay. Activities undertaken as part of FRDC 2015-024 have also been extended to the Storm Bay environmental licenses, with methods for depositional modelling adopted into the Environmental Licence requirements for leases in Storm Bay. We have had two Honours projects focused on improving the utility of depositional modelling in Macquarie Harbour and another Honours project examining critical sheer stress in higher flow environments, which have all been built on the outputs from this project.

We have also extended the use of CONNIE from FRDC 2015-024 into exploratory leases in Storm Bay and broader functions such as invasive clam dispersal or seaweed dispersal from farming activities. The process has also been employed in several subsequent advice notes to DPIPWE for modelling risk of disease transfer between salmon leases in Storm Bay. The applications for depositional and dispersal monitoring are wide-ranging and we expect the use of these tools to continue to extend into the future.

More broadly, IMAS has made submissions to both Federal senate and Tasmanian legislative council enquiries regarding the environmental sustainability of finfish aquaculture, where scientific outcomes from FRDC 2015-24 were drawn on to provide input. A submission was also made to the Marine Farming Planning review panel for the independent assessment of salmon farming at Okehampton Bay to inform management and monitoring of salmon farming, particularly regarding environmental baseline data available for reef habitats.

9.1 Project coverage

We have regularly contributed to media, industry and government articles based on the progress and outcomes of this project and have participated in scientific and community discussions on the interactions of salmon farming.

Most significantly, we have developed the Salmon Interactions Team website (https://salmoninteractionsteam.org/), which has become a platform for dissemination of scientific information, including outcomes from this project.

We have also contributed to broader IMAS communications using outcomes from this project, including FAQs/Research Insights (https://salmoninteractionsteam.org/research-insights-faq), TSIN articles ("Methods for monitoring reefs") as well as social media posts through IMAS.

Perhaps most notably, the findings of this project in Macquarie harbour where pivotal in driving change in management and industry practice in Macquarie Harbour. Through this project and its successor in Macquarie Harbour (FRDC 2016-067) there have been multiple public reports, presentations at public forums in Strahan and elsewhere, and numerous interviews for the print media, televisions, and radio, including several live interviews on ABC Mornings with Leon Compton.

Above all, we have been in constant communication with industry and government, disseminating information through presentations of data and discussion around implications of outputs.

10 Appendices

10.1 List of Researchers and Project Staff

IMAS staff for field sample collection, lab analysis and data preparation

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