Assessing the risk of marine invasive species to nearshore marine ecosystems of Australia's Antarctic research stations and subantarctic islands

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Abstract

The oceans surrounding Antarctica represent one of the world's remaining near-pristine environments. Despite this, the pressures on these regions are growing. Climate change is expected to negatively impact many endemic Antarctic marine species, with as many as 79% of species expected to lose at least part of their habitat due to rising temperatures. These are species which have been near-isolated for more than 20 million years in harsh, yet stable, environments. They are ill-equipped to cope with even minor changes to their environments. A secondary effect, which is of growing concern to Antarctic marine scientists, is that these changes to the climate will make the region more hospitable to marine invasive species. Coupled with this is the increasing human, and ship presence in the Southern Ocean, creating more opportunities for non-native species to be carried to the region. Until now, we have had very little understanding of the potential impacts that these non-native species may have to endemic ecosystems and species.

There has been considerable work done in the region for terrestrial invasions; however, there has been a paucity of such work in the marine realm. There are currently no known established populations of marine invasive species in the region, but the list of observations of marine invasive species is growing. A window of opportunity exists for the Antarctic region to predict, develop plans for, and implement policy instruments to get on the front foot of marine invasions. The adage 'prevention is better than cure' is also a mantra in the field of invasion biology; often by the time we notice invasions they are difficult, and expensive to act upon. Given the remoteness and hostility of the Southern Ocean environment, we need to be able to predict invasions before they occur and take policy and legislative steps to reduce the transfer of and subsequent establishment of these species.

Australia currently manages three continental research stations in East Antarctica, as well as two subantarctic islands in the Southern Ocean. This thesis explores the risk of marine invasive species reaching and establishing in shallow coastal marine ecosystems near these regions, explores potential impacts that invasive species could have on native ecosystems, and interrogates the policy instruments designed to protect the region from marine invasive species to identify any gaps that currently exist.

There are four key stages of species invasion. These are introduction to a new region, establishment in the new region, spread beyond the initial establishment site, and impact on native ecosystems. Two of these stages, introduction and spread, were explored using Lagrangian particle tracking (Chapter 3). To assess the introduction stage, I ran particle tracks in reverse to find potential source locations of planktonically-dispersed species that are carried to each of the five Australian Antarctic sites. The subantarctic was connected to substantially more potential source locations, extending as far as the South American coastline. In contrast there were very few connections to potential source locations for Davis and Mawson stations. Casey was connected to both coastal Antarctic sites and the subantarctic and could provide a stepping-stone for subantarctic species, or non-native species in the subantarctic to enter the coastal ecosystems of the Antarctic Continent. To explore the ability of species with planktonic larvae to disperse around the Southern Ocean from the Australian Antarctic sites, I ran particle tracking simulations forward in time with the Australian Antarctic sites as the point of origin.

The second stage for a species to overcome on the journey to becoming invasive is establishing in the new region. I used an extreme gradient boosting machine learning algorithm to predict which known invasive hull fouling species could survive in the waters near the Australian Antarctic research bases and subantarctic bases currently, and in the future under two climate change scenarios (Chapter 4). I found that most species would be unable to survive the hostile Southern Ocean environment: however, four species were identified that could inhabit the region currently and five that could survive the conditions in the future. At all five sites, the predatory asteroid *Asterias amurensis* was identified as a threat. Although these five species would struggle to establish under Antarctic conditions, being able to survive in the region is one less barrier for these highly adaptable species to overcome before becoming established.

The final stage to becoming invasive is having an impact on the native ecosystem. This is explored in Chapter 5 of this thesis. I collated a list of high-risk marine invasive species that have been identified in this thesis and by other expert working groups. I used an ensemble-ecosystem model to simulate a range of potential interactions between native species and an introduced species to explore the range of plausible impacts following a successful invasion. I found that there were no species for which an extinction-level decline was certain, though every species did experience significant population declines in at least some of the simulations with all the different invasive species. The worst affected native species group was the predatory gastropods when the system was invaded by the ascidian *Botryllus schlosseri*, where around 40% of simulations resulted in the extinction of the native species. On average, native species experienced a decline in abundance following the invasion of species. The invasive species, on average, also remained at quite low abundance, and in some cases declined in abundance. In keeping with the precautionary principle, a very high significance of impact was assigned.

Chapter 6 synthesizes the existing literature with my findings to provide an overall risk assessment of the marine invasive species threat to the nearshore marine environments of Australia's Antarctic research stations and subantarctic islands. I have shown that the overall risk is high at the Australian Antarctic sites, due to the potential for invasive species to plausibly cause native species extinctions. Whilst the risks to this region are substantially lower than other regions of the Antarctic, particularly the west Antarctic Peninsula region, there is still a plausible risk of marine invasive species being introduced to, establishing in, spreading from, and impacting the native nearshore coastal ecosystems adjacent to the Australian Antarctic research stations. The subantarctic islands are more difficult to quantify, given the lack of species community information of nearshore benthic ecosystems in these regions, making predictions around invasive species impacts impossible. Macquarie Island is at higher risk, due to the increased opportunities for introduction and the milder environmental conditions making establishment more likely. However, without understanding the range of potential impacts from invasive species I provided a range of risk, which was low to severe overall. Similarly, for Heard Island I could not predict the impacts of marine invasive species and have provided a range of risk. The risk of introduction and establishment is lower overall than Macquarie Island, so the overall risk is between low and high for Heard Island.

Chapter 7 of this thesis explores how well current Australian policy instruments protect these Antarctic and subantarctic regions from marine invasive species, through all stages of an invasion process. I constructed a social-ecological model of the system using the drivers-pressures-state-impact-response (DPSIR) framework to identify causal links between the social and ecological components of the system and framed the responses around the four key stages of an invasion. Policy responses include monitoring and surveillance for invasive species on pathways and at recipient locations, increasing standards of hull cleaning and the development of rapid management action plans to deal promptly with non-native species. The policy responses were well covered in the policy instruments; however, they were largely general in nature rather than specifically aimed at MIS.

This is the first comprehensive study to assess the risk of marine invasive species to all the nearshore coastal environments of Australia's Antarctic research bases and subantarctic islands. I have shown that the overall risk is high, due to the potential for invasive species to plausibly cause native species extinctions. Whilst the risks to this region are substantially lower than other regions of the Antarctic, particularly the west Antarctic Peninsula region, there is still a plausible risk of marine invasive species being introduced to, establishing in, spreading from, and impacting the native nearshore coastal ecosystems adjacent to the Australian Antarctic research stations and subantarctic islands.

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Keywords

Antarctica, anthropogenic impacts, ecosystem modelling, food web impact, invasion ecology, machine learning, marine ecology, marine invasive species, particle tracking, polar, policy analysis, risk assessment, Southern Ocean, subantarctic

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Abbreviations

CanESM2	Canadian earth system second generation
CMIP5	Coupled model intercomparison project phase 5
DPSIR	Driver, pressure, state, impact, response framework
EEM	Ensemble ecosystem modelling
GBIF	Global Biodiversity Information Facility
GISD	Global invasive species database
HYCOM	Hybrid Coordinate Ocean Model
IMO	International Maritime Organisation
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
IUU	Illegal, unreported, and unregulated (fishing)
MIS	Marine invasive species
NCEP	National Centres for Environmental Prediction
NNS	Non-native species
OBIS	Ocean biogeographic information system
QUT	Queensland University of Technology
RCP	Representative concentration pathway
ROSE	Random over-sampling examples
SSS	Sea surface salinity
SST	Sea surface temperature

Contents

Chapter 1: Background

1.1 Motivation

Sitting in a lecture theatre at The University of Queensland in 2015, as a Master of Conservation Science student, I awaited the conservation problem paper I was to be randomly assigned, which I was to use to create a conservation action. I had been at home the previous day with a sick child, so I and one other student were to receive the final two papers. We were handed the papers by Professor Hugh Possingham and began to look over them. The other student was bitterly disappointed in her paper, because it was based in the Arctic, and she was from snowy Canada. I told her that I would happily swap my paper (not that I can remember what mine was now) so that I could work on the Arctic paper. It was that simple, uneventful exchange of papers that led me here.

The paper in question was *Climate change, non-indigenous species and shipping: assessing the risk of species introduction to a high-Arctic archipelago* by Ware et al (2014). The aim of this study was to identify areas in Svalbard, Norway, that were at high risk of being environmentally suitable for marine invasive species (MIS) because of climate change. The main conclusion of this study was that the risk of MIS introduction and establishment would increase over the coming decades in Svalbard, and indeed, the entire Arctic region, unless concerted preventative management measures were put in place. Up until this point, I had not thought much about invasive species and climate change in polar regions. Something in this paper though clicked with me, and I was interested to know, *if this is a likely threat in the Arctic – what threat is there to the Antarctic?* Now, thanks to an incredibly supportive supervisory team, I can, at least part way, answer that question.

1.2 Context

The Southern Ocean represents one of the last near-pristine environments on the planet. Whilst it has not been spared from human influences, such as marine plastic pollution and climate change, it has, so far, been protected from the establishment of MIS. There are a number of factors which afford this protection, such as: hostile environmental conditions; isolation and detachment from other continents; and limited human presence (Peck et al., 2005). However, we are already seeing the erosion of some of these environmental barriers as a result of climate change affecting the Southern Ocean and Antarctica (Duffy et al., 2017). The Western Antarctica Peninsula region is one of the fastest-warming places on the planet, where the sea surface temperature has increased by 1˚C between 1951 and 2005 (Meredith and King, 2005).

So far, East Antarctica, in which the Australian Antarctic bases and subantarctic islands that are the focus of this thesis are located, has been spared these dramatic temperature increases, with research showing that the ozone hole may be buffering these warming effects (Perlwitz et al., 2008). This, however, is likely to change in the future as the ozone hole repairs itself over the next half century (Turner et al., 2014). Nevertheless, a myriad of environmental changes are predicted to occur, including increasing sea temperatures decreasing salinity; increasing acidification, retreating sea ice, strengthening of winds and currents, and a loss of fitness in endemic biodiversity (Bergstrom, 2022; Griffiths et al., 2017; Smetacek and Nicol, 2005), though the total extent of these changes is uncertain due to the difficulty in modelling ice dynamics (IPCC, 2019). These changes could create an environment that is suitable for MIS.

MIS have caused irreparable damage to marine ecosystems worldwide (Ruiz et al., 1997). There are currently more than 2,100 species of introduced or invasive marine species known globally, with roughly 12% of those having a negative impact in their introduced range (International Union for the Conservation of Nature: Invasive Species Specialist Group, 2017). Ideally, species introductions are prevented from occurring; however this has not occurred for much of the world, and economically and ecologically catastrophic invasions have taken place (Bax et al., 2003; Chan and Briski, 2017). For example, the comb jelly (*Mnemiopsis leidyi*) that has invaded the Black Sea has contributed to the destruction of the local anchovy fishery by becoming the dominant planktonic organism (Ivanov et al., 2000). Similarly, the zebra mussel (*Dreissena polymorpha*) has caused billions of dollars in damage in the United States alone since being first detected in the Great Lakes in 1988 (Roberts, 1990). However, one region of the world, the Southern Ocean, has remained free of MIS owing to its remoteness and harsh environment.

To date, there are no recorded non-native species (NNS) having established in the Southern Ocean, though the literature indicates several occurrences where NNS have been discovered in Antarctic waters. First, a red macroalgae species (*Porphyra linearis*) native to the North Atlantic and Mediterranean (Varela-Álvarez et al., 2017), was found in a survey of marine benthic macroalgae in the Antarctic Peninsula region in 1994 (Clayton et al., 1997). It was stressed, however, that only one specimen of *P. linearis* was found during the survey and that more evidence was needed to confirm the presence of the species in the region. Second, a qualitative plankton haul in the Antarctic Peninsula in 2002 included two decapod larvae of the genus *Emerita* and the genus *Pinnotheres* (Thatje and Fuentes, 2003). Both these genera are found in South America, and it was concluded that these larvae were likely of South American origin. Finally, a species of shell-crushing crab, *Hyas araneus*, was recorded during an oceanographic survey of the Antarctic Peninsula region on the *RV Prof W. Besnard* in 1986 (Tavares and De Melo, 2004); however there has been no further evidence that this species has established a population in the Southern Ocean (Barnes et al., 2006).

The Southern Ocean's biota is comprised primarily of endemic species, which will be amongst the worst affected by climate change (Griffiths et al., 2017). Along with species in the Arctic and high alpine regions, Antarctic species do not have the capacity to move poleward in response to warming conditions. Often, the endemic species are cold-stenotherms, and increased ocean temperatures could push these species to their upper thermal thresholds (Clarke et al., 2007; Peck, 2002). Adding invasive species to an already stressed environment, through climate-change mediation range expansions (Walther et al., 2009), natural range expansions, or human-facilitated transfers (Lewis et al., 2005), could create a combination of stressors that has the potential to drive the endemic species to extinction with unknown flow-on consequences for the greater ecosystem.

Whilst it is concerning that we already have examples of incursions by NNS, it should be noted that these have occurred in the Antarctic Peninsula region. This is unsurprising as it is relatively close to the South American continent, the environmental conditions are comparatively mild compared to East Antarctica, it receives much more shipping activity; and there is more research conducted there than in East Antarctica. By contrast, the East Antarctic region has received very little attention in terms of NNS and potential MIS.

1.3 Purpose

The purpose of this thesis is to detail the risk of MIS reaching shallow ocean benthic habitats near Australia's Antarctic continental stations and subantarctic islands. It is structured in a way that covers the key components of a risk assessment (Figure 1.1):

1. Likelihood of the hazard occurring, and
2. Consequences of the hazard occurring.

This will then allow me to give an indication of overall risk of MIS in these regions. Finally, I will elucidate policy responses currently in place that are designed to prevent and/or manage MIS transfer to these regions.

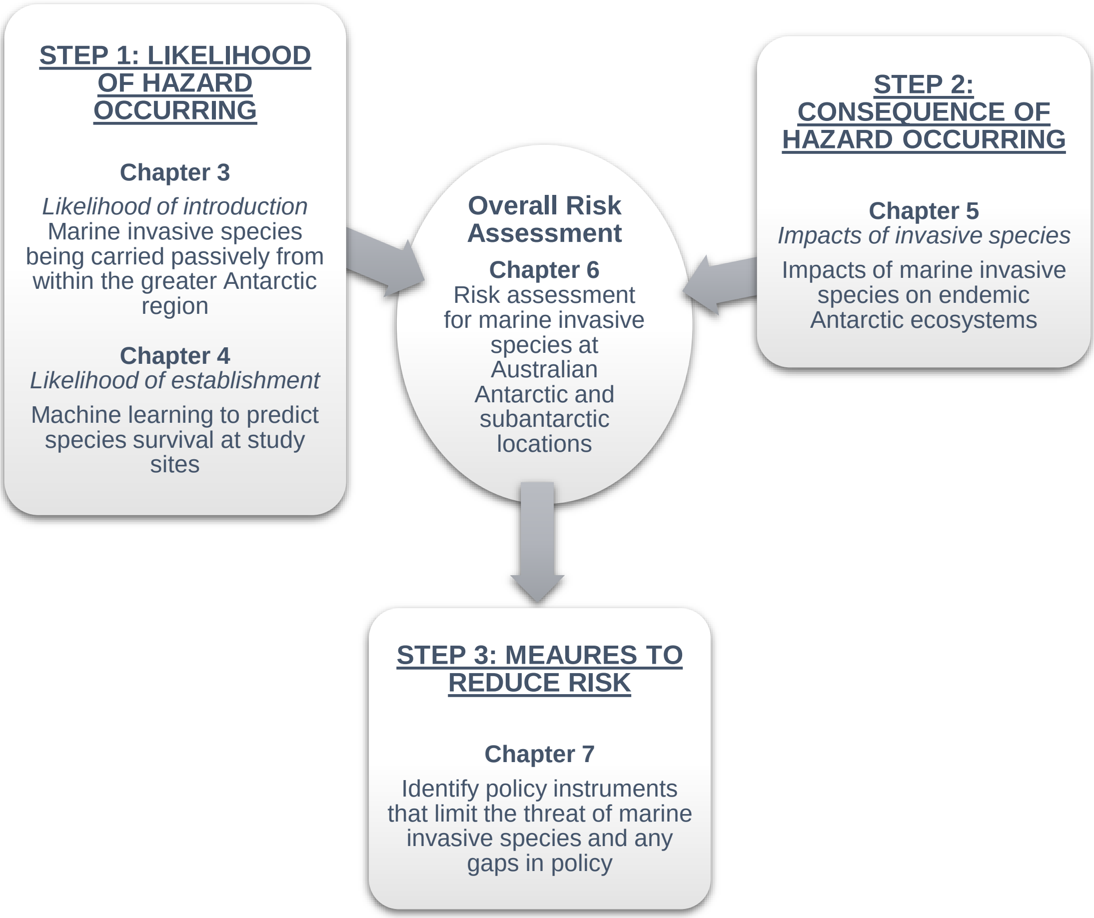

Figure 1.1 Components of the thesis structure, which in combination provides a risk assessment of the threat of marine invasive species to nearshore environments adjacent to Australia's Antarctic research stations and subantarctic islands.

1.4 Thesis Outline

Chapter 1 provides the context for this thesis, details how the different chapters come together to form an overall risk assessment, a summary of each chapter, and the significance of the work.

Chapter 2 critically examines the body of work surrounding the theme of MIS in the Antarctic, and particularly, the Australian Antarctic region. This chapter also provides an overview of the five key study regions: Casey Station, Mawson Station, Davis Station, Macquarie Island, and Heard Island, which are all managed by the Australian Antarctic Division.

Chapter 3 details the use of particle tracking software to determine potential sources of MIS from within the greater Antarctic region. Back-tracking models are used to identify which areas of the Antarctic and subantarctic are oceanographically connected to the Australian Antarctic research bases and subantarctic islands. Forward tracking of particles from those source points will then give an idea of the hazard that an invasion at that location would place on the Australian Antarctic and subantarctic locations.

In Chapter 4 an extreme gradient boosting machine learning algorithm is used to predict which known MIS that have a hull fouling association could survive in the shallow marine benthic habitats adjacent to Australia's Antarctic research stations and subantarctic islands. Whilst this chapter alone does not provide a total assessment of 'risk', it does allow us to focus surveillance and quarantine measures to prevent the accidental transfer of those species found to be a potential threat to the Australian Antarctic Territory and subantarctic islands.

Chapter 5 details the use of a species interaction network analysis to understand the potential impact that known MIS identified in Chapter 2, as well as other high-risk MIS identified in the literature, could have on benthic ecosystems of the Antarctic and subantarctic.

Chapter 6 synthesizes the outcomes from Chapters 2 – 5 to form an overall risk assessment of the risk of MIS in nearshore ecosystems adjacent to Australia's Antarctic research bases and subantarctic islands.

Chapter 7 examines the current Australian policy that relates to non-native species transfer. I use the drivers-pressure-state-impact-response (DPSIR) framework to conceptualize the social-ecological model of the system in order to identify potential policy responses that could be used to stop or limit MIS transfer to the Antarctic and subantarctic. I then use these responses to review current Australian-applicable policy instruments and identify gaps in policy.

Chapter 8 is the overall discussion of the thesis, summarizing the results and providing suggestions for future work.

1.5 Significance

To date there have been no risk assessments of MIS conducted for the Australian Antarctic and subantarctic regions, except for the Macquarie Island Marine Park to the east and south of Macquarie Island (Summerson et al., 2006), despite growing concern about MIS in Antarctic waters (Hughes et al., 2020; McCarthy et al., 2019). These areas house unique and vulnerable ecosystems and species which are already under stress from climate change (Bergstrom, 2022; Ingels et al., 2012). We have a unique opportunity to prevent marine invasions from occurring or to implement surveillance and management policies to identify and deal with incursions early in the invasion process.

If non-native species are allowed to establish in the Southern Ocean and become invasive, it will be incredibly difficult to manage, let alone eradicate them from these ecosystems (Hughes et al., 2020). The problem is likely to be compounded by the fact that the Southern Ocean is an internationally managed region which relies on consensus decision making, and responses will be subsequently slower than a response implemented by an individual sovereign state (Hughes and Pertierra, 2016).

Southern Ocean coastal marine organisms, particularly those around the continent are particularly at risk of being impacted by multiple stressors. To cope with warming waters, many species globally will extend their poleward distribution or move to deeper waters. For species living near Antarctica, there is no cooler refugia for them to migrate to. At least 79% of known native species will lose at least part of their thermally suitable habitat before the end of the century (Griffiths et al., 2017). These native ecosystems are already under the stress of climate change and adding invasive species to the mix could result in the loss of more species than would be expected from either stress individually.

This thesis is designed so that the methods can be readily applied to other regions of the greater Antarctic and Southern Ocean region, to create a holistic risk assessment of the MIS threat to one of the last remaining near-pristine environments left on Earth.

Chapter 2: Introduction and Literature Review

2.1 Study sites

Australia currently manages three continental research stations (Casey, Davis, and Mawson) and two subantarctic island groups (Macquarie Island, and the Heard and McDonald Island group) in the East Antarctic region, and these are the focus of this thesis (Figure 2.1). For the purpose of this thesis, the Heard and McDonald Island group will be simplified to Heard Island, except where it has appeared as Heard and McDonald Islands in the published Chapter 4.

2.1.1 Australian Continental Research Stations

The three Australian Antarctic continental stations are staffed year-round; with summer populations exceeding winter populations. These stations are all located next to the ocean with facilities for large ships to dock. Shipping to these stations only occurs around the Austral summer, as winter sea-ice (also known as fast-ice) prohibits the arrival of even ice-breaker ships to these locations. International air travel to this region is possible during the summer, with Casey Station being located approximately 65 km from the Wilkins Aerodrome. Intracontinental flights can be made on smaller ski-ways which are located near all three stations.

Compared to the Antarctic Peninsula region, East Antarctica receives very little shipping traffic (McCarthy et al., 2019). Shipping is largely confined to national research vessels which enter the region from Hobart, Tasmania, Australia. There are occasions when ships from other regions, such as New Zealand, South Africa, and some northern hemisphere ports, dock at the Australian Antarctic locations, but they represent a small fraction of ship visits (Australian Antarctic Division, 2021a). Also, unlike the Antarctic Peninsula, tourism is effectively non-existent in East Antarctica due to the harsher conditions and much greater distances from neighbouring continents.

Figure 2.1 Location of Australia's continental Antarctic research stations in East Antarctica and Australia's subantarctic islands. Also shown are the Subantarctic Front, Polar Front, and Southern Antarctic Circumpolar Current Front. Adapted from the Quantarctica 3 dataset (available from https://www.npolar.no/quantarctica).

Casey Station is the eastern-most of the Australian continental research stations (66˚ 16′ 55″ S, 110˚ 31′ 39″ E) and is closest to the Australian continent (Australian Antarctic Division, 2019a). It is located on the Bailey Peninsula region of Wilkes Land, East Antarctica. Davis Station is the most southerly of the continental research stations (68˚ 34′ 36″ S, 77˚ 58′ 03″ E) and is in the Vestfold Hills region in Princess Elizabeth Land (Australian Antarctic Division, 2019b). It is the most southerly of the Australian Antarctic research stations. Mawson Station is the western-most of the continental research stations

(67° 36′ 10″ S, 62° 52′ 26″ E) and is in the Holme Bay region in Mac Robertson Land (Australian Antarctic Division, 2019c). Compared to Australia's other continental stations, Mawson station has fewer shallow coastal environments. The bathymetry sharply drops off to over 100 m depth. Mawson station holds the record for the longest continual operation in the Antarctic.

2.1.1.1 Antarctic nearshore marine ecosystems

Shallow marine habitats are rare in Antarctica and limited to regions where the land is seasonally ice free (Clark et al., 2015). Consequently, they generally coincide with regions with a human presence. Shallow coastal environments display incredible heterogeneity in the communities they support (Stark et al., 2019). Sea ice is the biggest driver of ecological spatial heterogeneity in the region, due the role it plays in regulating available light to the benthos (Clark et al., 2017; Johnston et al., 2007). Within these habitats there is considerable spatial heterogeneity, with some areas experiencing year-round sea-ice cover and others that have varying levels of sea ice cover that break out over the Austral summer. As a result, the amount of photosynthetically available light varies throughout these habitats. Regions that are perpetually covered in ice are free of macroalgae, and instead are dominated by marine invertebrates, like polychaetes, echinoderms, and molluscs (Johnston et al., 2007). Conversely, areas which receive a relatively larger light budget are often macroalgae dominated.

There is also a suite of environments which experience periods of sea ice cover in the colder months and then are seasonally ice free. Regions that are ice-free (seasonally or perpetually) are vulnerable to iceberg groundings (Constable et al., 2014). These icebergs scour benthic organisms as they come aground, and ecosystems can take many years to recover, and could create areas for introduced species to inhabit (Gutt et al., 2015; McCarthy et al., 2019). These ice-free habitats are also more vulnerable to disturbance by wind and wave action (Stark et al., 2019).

Ocean temperatures remain below freezing, with many endemic species having physiological adaptations to cope with sub-zero life (Peck, 2002). These endemic species are typically stenothermic and can only tolerate very little change in environmental temperatures. Antarctic marine species also have reduced growth rates and extended generation times, making them particularly unsuited to abrupt changes in the environment (Peck, 2005). Productivity is also highly seasonal in the Antarctic region, resulting in many endemic species having alternating feeding regimes and the region displaying a high degree of omnivory (Peck, 2018).

2.1.2 Australian Subantarctic Islands

The two subantarctic island groups managed by Australia are Macquarie Island and Heard Island.

Macquarie Island is located halfway between Australia and the Antarctic continent (54˚ 37′ 12″ S, 158˚ 51′ 40″ E). It contains a permanent research facility and expeditioners are taken ashore in tenders from larger ships. Though the research facility and expeditions are managed through the Australian Antarctic Division, its jurisdiction lies with the Australian state of Tasmania. It is located on the Macquarie Ridge where the Pacific and Australian tectonic plates meet. Due to its unique geology, the island was given a World Heritage Site status in 1997.

Heard Island (53˚ 3' 0"S, 72˚ 37' 12"E) is the only location studied in this thesis without a permanent human presence or research facilities. Unlike Macquarie Island, Heard Island is under federal jurisdiction and, is again, managed by the Australian Antarctic Division. The islands sit atop the Kerguelen Plateau in the Southern Indian Ocean, which is situated on the Polar Front, with Heard Island sitting to the south of the Polar Front. Similar to Macquarie Island, Heard Island as World Heritage Listed in 1997 due to its outstanding universal natural values.

2.1.2.1 Subantarctic nearshore marine ecosystems

The nearshore marine environments of Australia’s subantarctic islands are biogeographically similar (Thackway et al., 1998). The nearshore benthic habitats of Heard Island and Macquarie Island vary depending on topography and oceanography of the region (Meyer et al., 2000). Both island groups sit in close proximity to the Polar Front, with Macquarie Island sitting slightly north, and Heard Island sitting slightly south. The islands’ maritime settings and position within the range of the world's strongest westerly winds make them susceptible to strong wave action in shallow waters. Like many regions of the subantarctic, echinoderms are prevalent and there is an abundance of slow-growing species.

2.2 Risk assessment framework

The purpose of this thesis is to provide a risk assessment of the threat of MIS to nearshore coastal environments of Australian Antarctic stations and subantarctic islands. This thesis will enhance understanding of the risk posed by MIS in an Australian context. The framework I use here (Figure 2.2) is adapted from the initial environmental evaluation for the operations of the new Australian icebreaker ship, *RSV Nuyina* (Australian Antarctic Division, 2021b). Three chapters of this thesis contribute to the risk assessment (Chapter 3, 4, and 5).

	Consequence of successful invasion				
Likelihood of successful establishment	**Insignificant**	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Medium	Medium	High	Severe	Severe
Likely	Low	Medium	High	High	Severe
Possible	Low	Low	Medium	High	High
Unlikely	Low	Low	Medium	Medium	High
Remote	Low	Low	Low	Medium	Medium

Figure 2.2 Risk assessment framework adapted from the *RSV Nuyina* risk assessment matrix (Australian Antarctic Division, 2021b).

The literature review synthesizes the current knowledge of non-native species being carried on ships to the Antarctic region, while Chapter 3 examines the potential passive pathways from the greater Antarctic region to the study sites (Figure 1.1). These two chapters therefore encapsulate the risk of a species being introduced to the locations studied in this thesis (Table 2.1).

Table 2.1 Definitions for the likelihood of introduction, adapted from the *RSV Nuyina* risk assessment matrix (Australian Antarctic Division, 2021b).

Likelihood of introduction	
Almost certain	Is expected to occur. Has occurred in the Antarctic region or subantarctic region in the past year.
Likely	Will probably occur. Has occurred in the Antarctic region or subantarctic region in the past two years.
Possible	Might occur at some time in the future. Has occurred in the Antarctic region or subantarctic region in the past five years.
Unlikely	Could occur but considered unlikely or doubtful. Has occurred in the Antarctic region or subantarctic region in the past ten years.
Remote	May occur in exceptional circumstances. Has not occurred in the Antarctic region or subantarctic region in the past ten years.

Then, in Chapter 4, I model the potential for known non-native hull fouling marine species to survive the present environmental conditions at each of the five study sites, and at points of time in the future based on two scenarios of climate change, to give an indication of the potential for the species' successful establishment (Figure 1.1 and Table 2.2).

Table 2.2 Definitions for the likelihood of successful establishment used in the risk assessment matrix, adapted from the *RSV Nuyina* risk assessment matrix (Australian Antarctic Division, 2021b).

Likelihood of successful establishment	
Almost certain	Is expected to occur in most circumstances. Has occurred in the Antarctic region or subantarctic region in the past year.
Likely	Will probably occur. Has occurred in the Antarctic region or subantarctic region in the past two years.
Possible	Might occur at some time in the future. Has occurred in the Antarctic region or subantarctic region in the past five years.
Unlikely	Could occur but considered unlikely or doubtful. Has occurred in the Antarctic region or subantarctic region in the past ten years.
Remote	May occur in exceptional circumstances. Has not occurred in the Antarctic region or subantarctic region in the past ten years.

The next step then, is to determine the impact of a potential invasive species (Table 2.3), that is a non-native species which has successfully established and spread beyond the site of establishment. In Chapter 5 I explore the range of potential impacts from a number of high-risk non-native species assuming they can successfully establish into a native Antarctic food web using ensemble ecosystem modelling (Figure 1.1). This modelling will give an indication of the potential consequences of a successful invasion.

Table 2.3 Definitions for the consequence of successful invasion used in the risk assessment matrix, adapted from the *RSV Nuyina* risk assessment matrix (Australian Antarctic Division, 2021b) and updated to include the IUCN Red List Categories definitions (International Union for the Conservation of Nature, 2012).

Consequence of successful invasion	
Catastrophic	Severe, widespread, and irreversible environmental damage. Local extinction of species OR loss of genetic diversity OR impact to one or more threatened species. Native species decline to Critically Endangered levels (population decline of 80% or more).
Major	Major environmental damage with ongoing impacts. Substantial loss of population OR potential loss of genetic diversity OR loss of individuals of threatened species. Native species decline to Endangered levels (population decline of 50% or more).
Moderate	Significant environmental damage with the potential for reversal with intensive interventions. Minimal impact on populations OR some impacts on individuals of threatened species. Native species decline to Vulnerable levels (population decline of 30% or more).
Minor	Isolated, but significant environmental damage with the potential for reversal with intensive interventions. Some individuals are impacted. No population impacts and no impact to threatened species.
Insignificant	Minor environmental damage that can be reversed. No observable change to native species.

2.3 The path to invasion – an overview

There are several hurdles for a species to overcome before it is considered ‘invasive’ and I summarise these in Figure 2.3 (Australian Government, 2017; Canning-Clode, 2015; Lodge et al., 2016). First, a species must be introduced to a new region. This means the species needs to be connected by some pathway to the recipient region, either geographically or anthropogenically, or a combination of both. Geographically connected regions can facilitate passive movement of species around a region, for example by way of ocean currents. Anthropogenically connected regions are where people travel between regions, such as travelling on a ship from one place to another. Further, the potential invasive species must be present on the pathway between the donor and recipient regions. This could be via plankton in a current, in the case of geographically connected regions, or by entrainment of a potential invasive species on a person, vehicle, or vessel, in the case of anthropogenically connected regions. In the Antarctic, hull fouling is the likely anthropogenic pathway for non-native species (Lewis et al., 2005). There are also passive oceanic connections between the Antarctic, the subantarctic, and South America which could contribute to the transfer of non-native species to the region (Fraser et al., 2018).

Secondly, the recipient region must provide a suitable habitat for the invasive species to survive and become established. This would require that the recipient region is environmentally suitable, that there is a suitable food source in the recipient ecosystem, that there is a competitive advantage by the potential invader to exploit, and that the region is free of predators that could remove an establishing population (Cárdenas et al., 2020). Finally, the species must be able to complete its life cycle in the recipient region and grow in population. This is when a species is deemed to have become established in the recipient region. If a species becomes established in the Antarctic it means that it has overcome the hostile environmental conditions and may have a competitive advantage over native species.

Third, the species must spread beyond its initial establishment site (Lange and Marshall, 2016). In the context of the Southern Ocean, this spread could occur by passive oceanic connections between the establishment site and other areas of suitable habitat. This is also where intra-Antarctic spread could occur from other regions around the continent. This is a concern for the Australian Antarctic and subantarctic sites, as it is more likely that an invasion would first occur in the Antarctic Peninsula region, where there are closer connections to the South American continent, and there is significantly more shipping traffic (McCarthy et al., 2019). A non-native establishing on the Antarctic Peninsula should serve as a warning to the greater Antarctic region, as it shows the species can survive in polar conditions and has undergone successful reproduction.

Finally, to be deemed invasive, a species must have a negative impact on endemic ecosystems or have a negative economic impact. This complex multi-dimensional biosecurity problem can be a tricky area to navigate as in other regions of the world there may be considerable economic gain from bringing in a non-native species (e.g. for food crops) even if there is a negative outcome for natural environments (Lodge et al., 2016). In contrast, there are limited, if any, economic benefits to be gained from introducing a non-native species to the Antarctic region and this practice is banned under the current Treaty arrangements.

There is an inverse relationship between the effectiveness of interventions aimed to control invasive species and the cost of the intervention (Figure 2.3) (Leung et al., 2002), particularly in the marine realm (Hughes and Pertierra, 2016) where eradication is a rare outcome and involves early identification of a MIS and a rapid response, generally in a localised region (Giakoumi et al., 2019). Therefore, being able to predict and then prevent introductions from occurring is a priority in MIS management (Hulme, 2009).

2.3.1 Barriers to invasion in Antarctica – a summary

It was long believed that the Polar Front created a physical barrier between the Southern Ocean and the other global oceans; however, this barrier has been shown to be permeable (Fraser et al., 2018). There are other conditions, however, that make the Southern Ocean inhospitable to marine species from other regions of the world. Near the Antarctic continent, ocean temperatures are often below freezing. Species in these regions have specially adapted to life in subfreezing temperatures, such as producing proteins that act as antifreeze in the blood of polar fish species (Lane et al., 2000). They have also adapted to highly seasonal productivity, which has led to an abundance of omnivory and prey switching practices.

Many invasive species are limited to the shallow coastal shelf regions of the world. Another feature of the Antarctic region which precludes many coastal MIS is that there is a paucity of ice-free environments, and that the Antarctic iceshelf extends out past the coastline in most places, abruptly facing deep ocean, and creating very little shallow environment for MIS to establish in the region (Stark et al., 2019). Those areas that are ice-free are subject to iceberg scour, sometimes completely scouring the benthic ecosystem from the ocean floor (Constable et al., 2014). Further, the presence of sea ice limits the ability of species to disperse around the coastal regions of the continent and limits the available light to benthic ecosystems. There are also no areas of coastline which are free from ice during the winter months (Peck, 2018), creating incredibly harsh environments for species trying to establish in these regions.

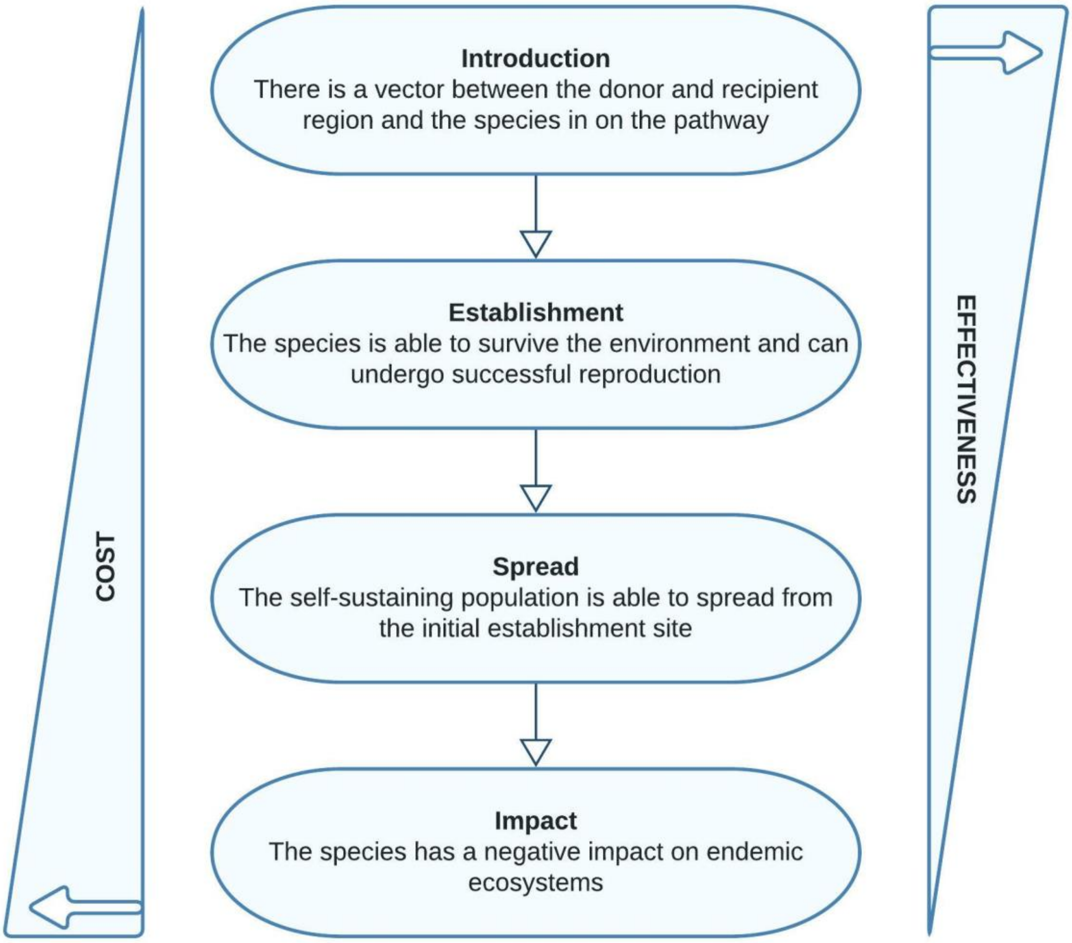

Figure 2.3 The invasion pathway. A species must successfully complete each step to be deemed invasive. The cost effectiveness of interventions is highest at the earliest stages of the invasion process, as is the effectiveness of management measures.

A key feature of the preclusion of MIS from the Antarctic region is its isolation from other continents (Stark et al., 2019). For example, it is located more than 3,000 km away from southern Australia. Human presence in the region is limited to the established research bases on the continent and subantarctic islands, and shipping traffic is significantly less than in other regions of the world (McCarthy et al., 2019). This means that the propagule pressure exerted by any one ship is low and that the chance that the species being carried by that ship being able to survive in the Southern Ocean waters is even less (Canning-Clode, 2015). Combined, these barriers could create a situation where genetic bottlenecking precludes a species from successfully establishing, due to low founding individuals and a lack of genetic flow between invaded and native ecosystems (Canning-Clode, 2015).

2.4 Assessment of the likelihood of successful establishment and spread of marine invasive species

2.4.1 Stage One: Introduction

The first stage of the invasion process is being introduced to a new area. To do this, there must be connectivity between a source population and the receiving region. The Australian Antarctic research stations and subantarctic islands are amongst the most isolated places on the planet. Australia's stations and islands in the East Antarctic region receive considerably less shipping traffic and human presence when compared to the West Antarctic Peninsula region (McCarthy et al., 2019).

2.4.1.1 Shipping to the study regions

Macquarie Island is the most visited of the Australian sites, as it is the only location visited by tourist vessels on a regular basis, as well as by research resupply vessels (Australian Antarctic Division, 2021a; McCarthy et al., 2019). There are also tourist ship voyages to Macquarie Island with itineraries which include the New Zealand subantarctic islands. Conversely, Heard Island is the least visited, sometimes with no ship visits within a year (Australian Antarctic Division, 2021a). The Antarctic stations are largely connected by research and/or resupply vessels and on average receive less than 15 ship visits per annum, compared to regions of the West Antarctic Peninsula which can receive more than 150 ships per annum (McCarthy et al., 2019). There are very few tourist ships that visit East Antarctica, especially when compared to the West Antarctic Peninsula. Shipping between the five Australian Antarctic and subantarctic sites occurs regularly, with one ship visiting two or more of these sites in the same voyage (Australian Antarctic Division, 2021a). There are fishing vessels which operate in the region of the subantarctic, with growing concern over the presence of illegal, unreported, and unregulated (IUU) fishing in the region (McCarthy et al., 2019). It is currently unknown if shipping traffic will increase in the East Antarctic but increased interest in Antarctica and the subantarctic as a tourist destination may see more ships visiting the region in the future (McCarthy et al., 2019).

2.4.1.2 Gateway cities as potential sources of marine invasive species

There are five Antarctic gateway cities that straddle the Southern Ocean: Hobart (Australia); Cape Town (South Africa); Christchurch (New Zealand); Ushuaia (Argentina); and Punta Arenas (Chile). Of these, Hobart is the key gateway city to the Australian Antarctic research stations and subantarctic islands. For tourist ships that visit Macquarie Island, the point of origin is most often from a South Island location in New Zealand, such as Bluff. Much less frequently, ships from other regions may visit the Australian Antarctic and subantarctic regions (Australian Antarctic Division, 2021a; Bergstrom, 2022; McCarthy et al., 2022).

Hobart has recorded over 70 introduced species in the Derwent Estuary, including nationally recognized pest species, such as *Asterias amurensis* (northern Pacific sea star), *Undaria pinnatifida* (Japanese kelp) and *Carcinus maenas* (European green shore crab) (Aquenal Pty Ltd, 2008a, 2008b, 2008c; Whitehead, 2008). Similarly, ports in New Zealand have a high load of MIS, including *U. pinnatifida*, as well as other species not currently found in the Derwent Estuary, such as *Charybdis japonica* (Asian shore crab) (Biosecurity New Zealand, 2021).

2.4.1.3 Human-mediated dispersal

Historically, the dumping of ballast water has been the key vector of species transfer around the globe. This led the International Maritime Organization (IMO) to develop the *International Convention for the Control and Management of Ships' Ballast Water and Sediments*. This convention outlines requirements for the safe management of ballast water so that it poses a much lesser threat of marine species transfer than in the past (International Maritime Organisation, 2004). Ballast water is taken on board ships to compensate for the weight of goods that have been removed. In respect to the Antarctic, discharge of ballast generally does not occur in the region as ships are moving goods from north to south (Barnes et al., 2006; Lewis et al., 2003). This means that ballast water is taken on in the Antarctic region, following the unloading of supplies at research bases. This presents an opportunity for Antarctic marine species to be carried to other ports around the world, but this phenomenon is outside the scope of this study.

Since the implementation of the *International Convention for the Control and Management of Ships' Ballast Water and Sediments* (International Maritime Organisation, 2004), it is now recognized that hull fouling presents the most likely vector by which marine species will be transferred around the world and to the greater Antarctic region (Drake and Lodge, 2007; Lewis et al., 2005; McCarthy et al., 2019). Currently, there are no similar conventions with respect to the management of hull fouling, though international guidelines have been issued by the IMO (International Maritime Organisation, 2011) which work in tandem with the *Polar Code* (International Maritime Organisation, 2014). Similarly, many countries have adopted national-level guidelines, such as the Australian series of voluntary guidelines for industry-specific management of hull fouling (Marine Pest Sectoral Committee, 2018). Likewise, New Zealand has formalized the standard for management of *Biofouling on Vessels Arriving to New Zealand* (New Zealand Ministry for Primary Industries, 2014).

Hull fouling is of particular concern in the Antarctic region for several key reasons. First, icebreaker ships are often laid up in gateway ports over the winter, where they can accumulate a suite of fouling organisms (Lee and Chown, 2007; Lewis et al., 2003; Summerson et al., 2006). These ships are not antifouled annually, meaning that significant populations of potential MIS could be inhabiting the hull and other submerged surfaces of the vessel. Second, any antifouling that is applied to the vessels is subject to scouring when the vessel travels through areas of sea ice (Lee and Chown, 2009). Whilst this situation has the added benefit of scouring off any attached fouling species, it should be noted that in the interests of safety, vessels will avoid sea ice where possible. Also, the first port of call following an over-wintering period may not be surrounded by sea ice, such as the subantarctic islands. Thirdly, not all submerged surfaces are subject to antifouling. Antifouling is applied in a dry dock, and the ship is held by wide sections of strapping, effectively excluding those areas from receiving antifouling. Finally, there are niche regions on a ship's submerged surfaces, such as sea chests, where significant populations of fouling organisms can establish and be carried in relative safety to the Antarctic and subantarctic regions (Hughes and Ashton, 2017; Lee and Chown, 2009, 2007; Lewis et al., 2005, 2004, 2003).

2.4.1.4 Hull fouling assemblages in the Southern Ocean

Recognition of the threat of hull fouling species in the Antarctic region has only recently gained traction. To date, there have been six studies which have examined the hull fouling assemblages of ships that have travelled to the Southern Ocean (Table 2.4) (Hughes and Ashton, 2017; Lee and Chown, 2009, 2007; Lewis et al., 2005, 2004, 2003). A common theme throughout these studies is that hull surfaces are almost entirely scoured of fouling organisms (and of antifouling coatings) after traversal through sea ice and that niche areas of a ships submerged surfaces, like sea chests, were found to harbour sometimes significant communities of fouling organisms. The studies have included surveys of national research and resupply vessels, *RV Aurora Australis* of Australia, *SA 'Agulhas'* of South Africa, *L'Astrolabe* of France, and *RSS James Clark Ross* of Britain. Research support vessels, *Sir Hubert Wilkins*, *RV Southern Supporter,* and the private yacht *Tiama* were also surveyed. Finally, there were two illegal, unreported, and unregulated (IUU) vessels which were confiscated by authorities which have also had their fouling communities surveyed, the *Viarsa* and the *Volga*.

Despite the paucity in the number of studies describing hull fouling assemblages on Antarctic vessels, nearly 80 species have been identified from 11 phyla, spanning three kingdoms (Tab 2.4). The *RV Aurora Australis* displayed the highest number of species (n = 38), though it should be noted that this vessel was surveyed more than any other vessel and at different time periods. The IUU vessel *Volga* also held a considerable variety of species (n = 17) and highlights the potential for these vessels to carry significant propagule loads in Southern Ocean waters. Some of the species found on these vessels are known to be invasive globally, such as the bivalve, *Mytilus galloprovincialis* and the colonial ascidian, *Ciona intestinalis* (Lee and Chown, 2007; Lewis et al., 2005).

Table 2.4. Summary of species found on hull fouling surveys of ships that operate in the Southern Ocean. Some species names have been updated since the surveys, referred to above, and names in the table reflect the currently accepted name for each species. National research and resupply vessels: *RV Aurora Australis*, *L'Astrolabe*, *RSS James Clark Ross*, and *SA 'Agulhas'*. Research support vessels: *RV Southern Supporter*, *Sir Hubert Wilkins*, and the private yacht, *Tiama*. IUU vessels: *Viarsa* and *Volga*. Species include those attached to submerged surfaces as well as species found within attached species (e.g., within algal matts on the hull surface).

Taxa	*RV Aurora Australis*	*L'Astrolabe*	*RV Southern Supporter*	*RRS James Clark Ross*	*SA 'Agulhas'*	*Sir Hubert Wilkins*	*Tiama*	*Viarsa*	*Volga*
PLANTAE									
CHLOROPHYTA									
Bryopsidales									
Bryopsis sp.			X						
Cladophorales									
Cladophora sp.						X			
Cladophoropsis sp.			X						
Ulvales									
Ulva compressa	X		X						
Ulva intestinalis			X		X	X		X	X
Ulva rigida	X	X				X			
Ulva sp.	X	X			X	X			
RHODOPHYTA									
Rhodophyta sp.						X			
Unspecified macroalgae		X							
Ceramiales									
Ceramium sp.					X				
Halymeniales									
Grateloupia filicina					X				
TRACHEOPHYTA									
Magnoliopsida									
Ruppia megacarpa	X								
ANIMALIA									
ANNELIDA									
Polychaeta									
Demonax leucaspis									X
Harmothoe sp.	X								
Hydroides elegans									X
Hydroides ezoensis			X						

Taxa	RV Aurora Australis	L'Astrolabe	RV Southern Supporter	RRS James Clark Ross	SA 'Agulhas'	Sir Hubert Wilkins	Tiama	Viarsa	Volga
Polydora sp.	X								
Sabella spallanzanii									X
Spirobranchus taeniatus	X					X			
Unspecified polychaetes	X	X							
ARTHROPODA									
Amphipoda									
Caprella sp.			X			X			
Corophiidae sp.	X								
Hyalidae sp.	X								
Hyperiid sp.			X						
Ischyroceridae sp.	X								
Jassa falcata			X						
Jassa herdmani			X						
Jassa sp.	X								
Monocorophium acherusicum	X		X						
Monocorophium insidiosum								X	X
Unspecified amphipods		X				X			
Cirripedia									
Balanus amphitrite			X					X	X
Balanus variegatus			X						
Conchoderma aurita			X						
Conchoderma auritum				X					
Elminius modestus	X	X				X	X		
Lepas antifera			X						
Lepas sp.	X				X				
Unspecified cirripedia	X	X							
Decapoda									
Halicarcinus quoyi	X					X			
Palaemon serenus						X			
Unspecified decapoda	X	X							
Isopoda									
Unspecified isopoda	X	X							
BRYOZOA									
Bugulidae									
Bugula flabellata	X						X	X	
Bugula neritina			X					X	X
Bugula stolonifera									X

Taxa	*RV Aurora Australis*	*L'Astrolabe*	*RV Southern Supporter*	*RRS James Clark Ross*	*SA 'Agulhas'*	*Sir Hubert Wilkins*	*Tiama*	*Viarsa*	*Volga*
Tricellaria occidentalis	X						X	X	
Celleporoidea									
Celleporina sp.			X						
Electridae									
Electra sp.	X								
Membraniporina									
Membranipora membranacea	X								
Schizoporellidae									
Schizoporella unicornis									X
Watersiporidae									
Watersipora subtorquata	X		X			X		X	X
CHORDATA									
Actinopterygii									
Unspecified actinopterygii	X								
Ascidiacea									
Ascidiella aspersa									X
Botrylloides leachi						X			
Botryllus schlosseri									X
Ciona intestinalis	X				X	X			
Didemnum sp.						X			
Styela conopsis						X			
Styella plicata									X
Unspecified ascidians	X		X						
CNIDARIA									
Hydrozoa									
Bougainvillia muscus	X								
Campanulariidae sp.	X		X						
Clytia hemispherica			X						X
Ectopleura crocea	X						X		X
Obelia dichotoma	X				X		X		
Obelia sp.	X								
Plumulariidae sp.			X						
Ralpharia magnifica	X					X			
Unspecified hydroids	X	X							
MOLLUSCA									
Bivalva									
Crassostrea gigas									X

Taxa	RV Aurora Australis	L'Astrolabe	RV Southern Supporter	RRS James Clark Ross	SA 'Agulhas'	Sir Hubert Wilkins	Tiama	Viarsa	Volga
Mytilus galloprovincialis	X		X		X	X		X	X
Gastropoda									
Nudibranchia sp.	X								
Unspecified gastropods	X								
PORIFERA									
Unspecified poriferans		X							
CHROMISTA									
OCHROPHYTA									
Phaeophyceae									
Dictyotales sp.						X			
Ectocarpales sp.	X	X				X			
Ectocarpus siliculosus					X				
OTHER									
Microbial/algal biofilms				X	X				

To date, there is no confirmation of established populations of MIS found in the greater Antarctic region, although a number of individual occurrences of MIS have been documented (Cárdenas et al., 2020; McCarthy et al., 2019). Some represent passive dispersals from within the Southern Ocean or adjacent regions, such as *Ulva intestinalis* (Clayton et al., 1997; McCarthy et al., 2019), *Pinnotheres sp.* (Thatje and Fuentes, 2003), and *Emerita sp.* (Thatje and Fuentes, 2003). The dispersal mechanism is unclear for *Halicarcinus planatus* (Aronson et al., 2015a; Griffiths et al., 2013) and *Rochinia gracilipes* (Griffiths et al., 2013). The remaining species, however, are likely to have been introduced to the region by anthropogenic means, either in ballast water or as fouling on hulls (Table 2.1). This includes the mussel species, *Mytilus* cf. *platensis*, for which a significant, but non-self-sustaining, population was discovered in the South Shetland Islands in 2019 (Cárdenas et al., 2020).

Table 2.5 Non-native marine species that have been observed within the Southern Ocean that were likely to have arrived by anthropogenic pathways (i.e. ballast water and/or fouling).

Species	Location	Year	Native range	Introduced range	Reference(s)
Bugula neritina (Linnaeus, 1758)	Dronning Maud Land, East Antarctica (70.3°S, 24.2°E)	1960	Unknown, global distribution recognized since late 18th century	North America, South America, Europe, Middle East, Asia, Australia, New Zealand, Pacific Islands, Atlantic Islands	(Winston and Woollacott, 2007) (Global Invasive Species Database, 2008) (Griffiths et al., 2003)
Ciona intestinalis (Linnaeus, 1767)	Dronning Maud Land, East Antarctica (71.1°S, 11.5°W)	1996	Believed to be northeast Atlantic	North America, South America, Australia, Asia, New Zealand, South Africa	(Gutt et al., 2000; Nydam and Harrison, 2007) (Global Invasive Species Database, 2007)
*Ectopleura crocea** (Agassiz, 1862)	Dronning Maud Land, East Antarctica (71.1°S, 11.5°W) and Queen Mary Land (approx. 66.0°S, 92.0°-96.0°E)	1966 & 1998	Northwest Atlantic	Northeast Pacific, America, South Africa, Australia, New Zealand, Mediterranean	(Griffiths et al., 2003; Hewitt, 2002)(Ocean Biogeographic Information System, 2020)
Hyas araneus (Linnaeus, 1758)	Elephant Island (61.1°S, 55.8°W)	1986	North Atlantic, Arctic Sea	No invaded range	(Tavares and De Melo, 2004)
Mytilus cf. *platensis* (d'Orbigny, 1842)	South Shetland Islands, West Antarctica Peninsula (62.0°S, 58.0°W)	2019	South America	No invaded range	(Cárdenas et al., 2020)

* This species appears as *Ectopleura ralphi* in the 'SOMBASE/TOTAL – Bioconstructors' dataset, however the two species have become synonyms as a result of a more recent study, with *E. crocea* taking name precedence (Imazu et al., 2014).

2.4.1.5 Oceanic connectivity

The Antarctic Circumpolar current encircles the Antarctic continent and flows in a continuous west to east direction (Figure 2.1). This current and its associated fronts have essentially created a physical barrier to invasion, though it is not completely impermeable (Fraser et al., 2018). South of the Southern Antarctic Circumpolar Current Front the direction of flow reverses (east to west) to form the Antarctic Coastal Current. These features have given rise to the circumpolar distribution of marine species such as krill (Griffiths, 2010). It also raises the possibility of MIS being spread around the continent if they were to establish in the Southern Ocean or adjacent regions.

Historical models of Southern Ocean connectivity have focussed on the flow of currents, and until recently, these models indicated that there was no exchange between regions north of the Polar Front and south of the Polar Front. However, recent modelling has included additional, finer scale oceanic processes, such as the Stokes drift which shows that this barrier between north and south of the Polar Front is more permeable than expected (Fraser et al., 2018).

2.4.1.6 Natural range expansions

East Antarctica is a considerable distance from adjoining continents; however there is a much narrower distance between the West Antarctic Peninsula and the southern tip of South America. The separation of these continents occurred roughly 25 million years ago, creating the Drake Passage and establishing the Antarctic Circumpolar Current (Tavares and De Melo, 2004). This delineation of ocean temperatures means that marine temperatures are still significantly higher in South America, than on the Antarctic coastal shelf (McCarthy et al., 2019). The ability of species to naturally expand their ranges from South America to the Antarctic coast is therefore limited.

However, there is current conjecture that durophagous crabs are poised to re-establish in the Antarctic Peninsula region, after being absent from the region for millions of years (Aronson et al., 2015a). This includes the subantarctic native *Halicarcinus planatus* and the globally invasive *Carcinus maenas*, whose ability to downregulate magnesium ions in their blood could allow them to survive in subfreezing temperatures. Whether this group represents a natural range expansion, or a reestablishment of species which were historically found in the region is under debate (Griffiths et al., 2013). Regardless, the arrival of this group to the Antarctic coastal region could have devastating effects for extant marine ecosystems (Aronson et al., 2015a, 2015b; Griffiths et al., 2013). The return of species which were previously precluded by the cold temperatures will likely see extinction of some extant and common species groups which have evolved in an environment free of these predators (Aronson et al., 2009).

2.4.1.7 Non-facilitated dispersal

As described previously, the Southern Ocean is more connected to the global oceans than previously realised. This could lead to a situation where marine species can be passively spread around the continent as plankton or as fouling on natural (e.g. kelp) or anthropogenic (e.g. plastic) rafts (Lewis et al., 2005). Further, the origin for this pathway could reside outside of the Southern Ocean, such as southern South America, or from subantarctic islands that are north of the polar front, like Macquarie Island. This has already been shown to occur with the non-facilitated dispersal of a subantarctic kelp, *Druvillaea antarctica*, from subantarctic South Georgia and Kerguelen (Fraser et al., 2018). Similarly, plastic debris has been identified as a potential vector for marine invasive species (Barnes, 2002; Lewis et al., 2005). However, plastic debris entering the Southern Ocean has either, come from within the Southern Ocean, or has originated from mid-ocean dumping, and is more likely to carry a suite of pelagic, rather than coastal invasive species (Lewis et al., 2005).

2.4.1.8 A back and forward Lagrangian particle model to identify sources

Lagrangian trajectory models have been used to model various particle drift tracks in the oceans and in the atmosphere, from oil (Jones et al., 2016) to plankton (Melsom et al., 2021) to drifting vessels for search and rescue purposes (Licer et al., 2020) to atmospheric radioactive drift (Bartnicki et al., 2016). There are a wide range of trajectory models available with varying computational requirements and applied for different domains (i.e., ocean and atmosphere). Regardless, they are run in either online or offline. The online models have the benefit of not needing separate models for particle trajectories and velocity fields (Dagestad et al., 2018; van Sebille et al., 2018). Offline models however have the benefit of being able to include multiple velocity fields that cover the different types of forcing, and can be run backward in time to determine potential origin sites of particles (van Sebille et al., 2018). These offline trajectory models require input from external forcing models, which are velocity variables for processes like ocean currents, Stokes drift and wind (Dagestad et al., 2018).

The Polar Front has been historically regarded as an impenetrable barrier for the southward dispersal of marine species, and indeed, ocean current models support this assumption. It is only when smaller scale oceanic processes are incorporated, by way of higher-resolution ocean current models that can resolve eddies as well as the inclusion of Stokes drift (wave direction propagation), that models show that the Polar Front is permeable (Fraser et al., 2018).

This then poses the question, could species be carried from either within the Southern Ocean or outside the Southern Ocean to the five study sites? And to that end, how do we determine where in the many thousands of kilometres of Antarctic, Australian, African, and South American coastlines do we disperse our particles from? This is where the backtrack feature could be useful. Particles can be released at key study sites and the particle tracking model can be run back in time, still using important ocean current, Stokes drift, and wind models. This can then be used to identify potential source locations for those particles. This though, is only half the story. Once the potential source locations are identified, forward running models could show us all the trajectories of particles release from the source site. This would facilitate quantification of the risk coming from each potential source location, and to identify potential source locations that may pose a high risk (i.e., those sites where a large portion of the particles released from the site reach the study sites within a biologically relevant time period). These sentinel sites can act as an early warning that invasion may be imminent. Chapter 3 develops such models for each of the five study sites explored in this thesis.

2.4.2 Stage Two: Establishment

To become established a species must be introduced into an area with environmental conditions suitable for the potential invader. The harsh environmental conditions combined with the remoteness of Antarctica have long been heralded as key reasons that MIS have not established in the region (Chown et al., 2015; Clarke et al., 2005), and that there have been few occurrences of non-native species in the Southern Ocean (Hughes et al., 2020; McCarthy et al., 2019). However, climate change is already starting to affect the region and increasing interest from national Antarctic programs, tourism operators, and legal and illegal fishing mean that the Southern Ocean is becoming more suitable and accessible for non-native species (Duffy et al., 2017; Hughes et al., 2020; McCarthy et al., 2019).

2.4.2.1 Predicting habitat suitability

The prediction of habitat suitability has generally taken two forms in the realm of invasive species: environmental matching and species distribution modelling. Environmental matching has been used to identify potential source locations of MIS based on the similarity of certain environmental variables (often sea temperature and salinity) between the area of concern and other areas that it is connected to (Ware et al., 2014). From this, researchers can identify environmentally similar areas and infer that marine species found within the matched sites could become an issue at the area of concern.

Species distribution models take the known occurrences (and in some cases known absences) of a species and overlay it with environmental variables which are known to drive the distribution of species (such as temperature, rainfall, etc). There are several statistical methods that are used to undertake these analyses, from generalised linear modelling, to maximum entropy models, to newer machine learning techniques such as neural networks, random forests, and gradient boosting (Elith and Leathwick, 2009; Früh et al., 2018; Shiferaw et al., 2019).

The emergence of the machine learning algorithms and their ability to make predictions with high accuracy in a multitude of fields has provided a new tool that can be used to make species distribution models. Currently, extreme gradient boosting (e.g. XGBoost), based on decision-tree methods, has been identified by the data science community as the leading algorithm for making predictions (Chen and Guestrin, 2016). This method creates an ensemble of decision trees to create a strong classification based on a set of weak classifiers. It is robust against overfitting, has customizable hyper-parameters, includes cross-validation, and its non-parametric nature makes it useful when working with correlated predictor variables (Shi et al., 2019). Chapter 4 uses this method to predict if there are any known hull fouling MIS that could currently, or in the near future, be capable of surviving the conditions at the three Antarctic sites or two subantarctic islands.

2.4.2.2 Climate change

The Antarctic Peninsula region is experiencing some of the fastest warming on the planet (Meredith and King, 2005). In contrast, East Antarctic has remained relatively stable, with research showing that the ozone hole may be buffering the warming effects of climate change (Perlwitz et al., 2008). However, this is likely to change in the future as the ozone hole repairs itself over the next half century (Turner et al., 2014). A myriad of predicted changes are expected to occur in the environment in East Antarctica, such as increasing sea temperatures, decreased salinity, increased acidification, retreating ice sheets, strengthening of winds and currents, and, a loss of fitness in endemic biodiversity (Griffiths et al., 2017; Ingels et al., 2012; Smetacek and Nicol, 2005). The result will be an environment that is becoming less suitable for endemic species, whilst simultaneously becoming more suitable for non-native species, as shown in Figure 2.4.

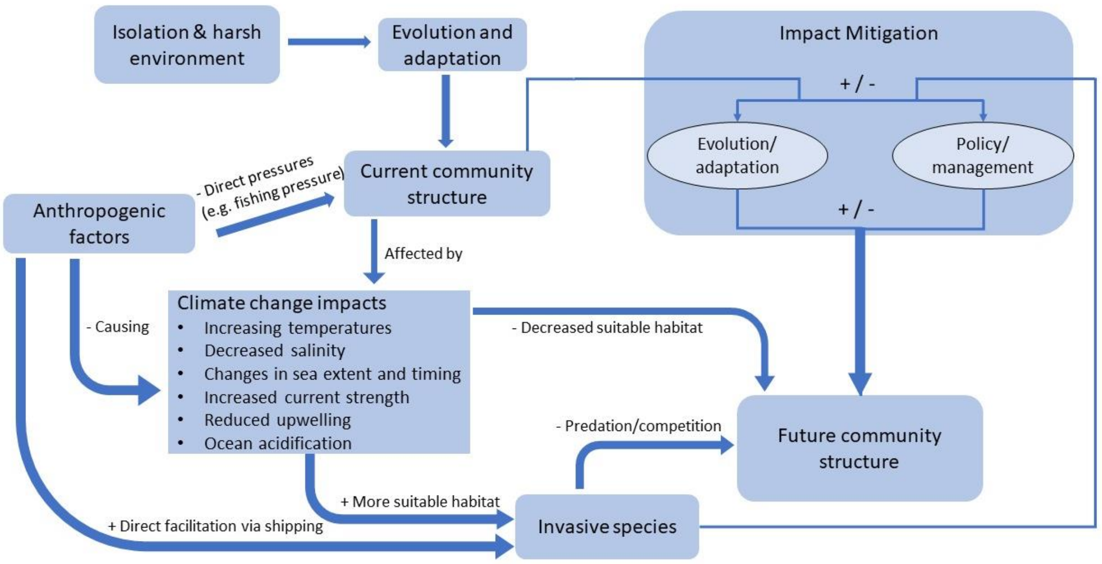

Figure 2.4 Conceptual diagram of the interactions and possible influence of stressors on the future benthic community structure in the Southern Ocean region.

2.4.3 Stage Three: Spread

As I am concerned with a marine environment, much of this stage mirrors stage one – introduction. There is a general consensus that marine invasions are likely to begin in the Antarctic Peninsula region by way of high traffic shipping to the region or by close proximity to the southern tip of South America (Hughes et al., 2020; McCarthy et al., 2019). The Antarctic Peninsula region is also undergoing more significant climatic changes, recording some of the fastest warming temperatures on the planet (Meredith and King, 2005). Even so, there are no known established non-native species in this region, indicating that environmental barriers to invasion are holding, that the species that have arrived have had an environmental or competitive disadvantage, or that we simply have not found them yet. There is also concern that the subantarctic islands could act as stepping stones for invasions (Byrne et al., 2016). Chapter 3 will identify any links between the Australian Antarctic stations and subantarctic islands and the Antarctic Peninsula, East Antarctica, the subantarctic islands, and nearby continents. Further, I will identify paths of spread from each of the Australian Antarctic sites and subantarctic islands to other regions within the Southern Ocean.

2.5 Assessment of significance of impact

2.5.1 *Stage Four: Impacts*

Although introductions of species should be avoided where at all possible, the impacts of introducing non-native species to a new region range from benign to catastrophic. Understanding the potential impacts that a species could have on a recipient ecosystem can help managers prioritize decisions around the actions to take in respect to each invasive species.

2.5.1.1 Predicted impacts of marine invasive species in the Southern Ocean

The impacts of marine invasive species (MIS) are largely unknown for the Southern Ocean. The most recent work in this field used expert working groups to identify high risk species and outlined the potential impact of each species if it were to establish in the Antarctic Peninsula region (Hughes et al., 2020). The result was a list of general statements that cover impacts, such as changes to community competition, competition with native species, and the assumption that the absence of a group in the current endemic community would lead to major impacts following the establishment of that new group. The authors do conclude, however, that there is considerable uncertainty in predicting these impacts.

2.5.1.2 How could we better predict the impact of marine invasive species on native ecosystems near Australia's subantarctic islands and Antarctic research stations?

There has been recent interest in developing tools to help understand ecosystem dynamics and how they can change following a perturbation. Qualitative modelling has been used to predict the changes that would occur to a community following the removal of one or more invasive species (Raymond et al., 2011) and to understand community structure and dynamics (Alexandridis et al., 2017; Dambacher et al., 2002; Mouillot et al., 2013). A common assumption of these methods is that the ecosystem is in equilibrium. While this assumption can be applicable in the context of understanding community dynamics, it is less suited to situations where there is a movement away from an equilibrium, such as when a species is being introduced (or indeed, reintroduced) to an ecosystem.

Newer quantitative methods, such as ensemble ecosystem modelling, have been developed to capture realistic and plausible changes to an ecosystem following a perturbation (Baker et al., 2017). By incorporating species growth rates this method can model outcomes where the final community structure is significantly different from the original stable community, and it has the capacity to model species extinctions. Chapter 5 uses this method to create an ensemble of ecosystem models following the introduction of 'high-risk' MIS into a known food web from Casey station, East Antarctica.

2.6 Policy and regulation of marine invasive species in the Antarctic and subantarctic region

No single government has control over the Antarctic, nor do any countries retain sovereign rights over any part of the region. Historically, Argentina, Australia, Chile, France, New Zealand, Norway, and the United Kingdom have laid territorial claims in Antarctica. However, these territorial claims were never legally recognised and the Antarctic Treaty does not recognise (nor remove) those claims, and as such, there are no sovereign rights conferred with those claims (Secretariat of the Antarctic Treaty, 2011).

The Antarctic Treaty (adopted 1959, entered into force 1961) was originally signed in Washington, USA, by twelve countries which were active in the Antarctic during the 1957-58 International Geophysical Year. These countries were Argentina, Australia, Belgium, Chile, France, Japan, New Zealand, Norway, Russian Federation, South Africa, United Kingdom, and United States of America. The overarching goal of the Antarctic Treaty is to ensure "*that Antarctica shall continue for ever to be used exclusively for peaceful purposes and shall not become the scene or object of international discord.*", whilst also "*promoting international cooperation in scientific investigation."* (Secretariat of the Antarctic Treaty, 2011).

The Antarctic Treaty applies to all areas south of 60˚S but does not affect the rights of any State with regards to the high seas within that area. Therefore, the Treaty does not cover either Macquarie Island (54˚S) or Heard Island (53˚S). Since the Antarctic Treaty came into force there have been two conventions and one protocol established as means to protect and manage the living resources of the Antarctic: the Convention for the Conservation of Antarctic Seals (adopted 1972, entered into force 1978), the Convention on the Conservation of Antarctic Marine Living Resources (adopted 1980, entered into force 1982), and the Protocol on Environmental Protection to the Antarctic Treaty (also known as the Madrid Protocol, adopted 1991, entered into force 1998).

Australia implements the Antarctic Treaty and its conventions and protocols through national legislation. This includes the *Australia Antarctic Treaty Act 1960 (Cth)*, the *Antarctic Treaty (Environmental Protection) Act 1980 (Cth)*, and the *Antarctic Marine Living Resources Conservation Act 1981 (Cth)*. In the marine realm, Australia is also signatory to the International Maritime Organization's Convention for the Control and Management of Ships' Ballast Water and Sediments (adopted 2004, entered into force 2017), the Convention on the Control of Harmful Anti-fouling Systems on Ships (adopted 2001, entered into force 2008), and the International Code for Ships Operating in Polar Waters (adopted 2014, entered into force 2017) (known as the Polar Code).

Heard Island is an Australian Territory and is managed by the Australian Antarctic Division. Macquarie Island forms part of the state of Tasmania and is managed under the Tasmanian Parks and Wildlife Service, although logistics and management of the station are managed through the Australian Antarctic Division. As these islands are not covered under the Antarctic Treaty, they fall under the guise of the United Nations Environmental Programme which includes the Convention on Biological Diversity (adopted 1992, entered into force 1993). Also, both these subantarctic island groups are listed on the World Heritage List in 1997 under the World Heritage Convention (adopted 1972, entered into force 1975) by the United Nations Educational, Scientific and Cultural Organization and the Australian National Heritage List 2007 under the Environmental Protection and Biodiversity Conservation Act 1999. Heard Island is covered by the *Heard Island and McDonald Islands Act 1953 (Cth)* and the subordinate Environment Protection and

Management Ordinance 1987. As Macquarie Island falls under the jurisdiction of the state of Tasmania, it is covered by the Tasmanian *National Parks and Reserves Management Act 2002 (Tas)* and the subordinate *National Parks and Reserves Management Regulations 2019 (Cth)*, and the *Threatened Species Protection Act 1995(Tas)* and the subordinate *Threatened Species Protection Regulations 2016 (Tas)*.

2.6.1.1 What are the gaps in policy to the threat of marine invasive species in the subantarctic and Antarctic regions?

MIS have only recently been identified as a matter of concern in Antarctica, and this is reflected in policies protecting for the greater Antarctic region. Indeed, an evaluation of policy surrounding non-native species explicitly states that '*The extent of non-native species in the marine realm is largely unknown; hence, in this work we focus predominantly on Antarctic terrestrial communities*' (Hughes and Pertierra, 2016). Since that evaluation, the Committee for Environmental Protection under the Madrid Protocol, has updated its Non-native Species Manual (Committee for Environmental Protection, 2019) to highlight the emerging threat of MIS. Still, it identifies marine invasions as a theme requiring additional research and suggests that the development of monitoring framework is still required. To date, there are no monitoring or surveillance techniques in use for MIS in the Antarctic region. Non-native species that have made it to the Antarctic region are rather found by chance.

Although it is clear that MIS are on the agenda for the Antarctic region, this has not necessarily translated into policies at the international or national level designed to prevent non-native species from being introduced or managed once they have been introduced. Although the premise of preventing invasive species seems to be uncontroversial, the reality is that implementation of policy and practices is up to the individual Consultative Parties within their own countries. This practice can therefore lead to inconsistent interpretations and varying levels of implementation and compliance measures between the various parties (Convey et al., 2012; Tin et al., 2014).

2.6.1.2 Evaluating institutional responses to the threat of marine invasive species in Antarctica in an Australian case study

Gap analysis is a common approach to examine whether there are sufficient regulatory responses to prevent or manage a threat. To do this a conceptual model of the socio-ecological system can be created to define the scope of the regulatory responses examined. One method to create this model is to use the Drivers-Pressure-State-Impact-Responses (DPSIR) framework (Smeets and Weterings, 1999) (Figure 2.5). The DPSIR framework has been used extensively as a decision-making tool in the environmental realm, and is used by national level assessments such as the Australian State of the Environment Reports (Jackson, 2017; Majorošová, 2016; Panov et al., 2009; Smeets and Weterings, 1999). The strength of this framework is that it explicitly captures the relationships between environmental, social, cultural, and economic components of a system (Bradley and Yee, 2015).

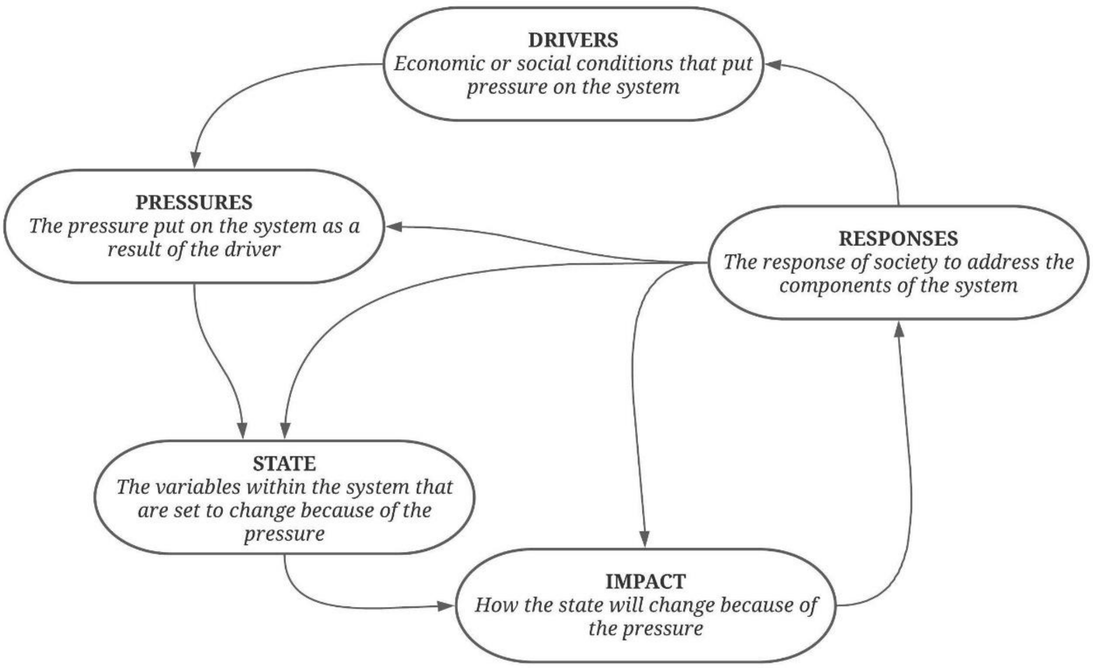

Figure 2.5 The DPSIR framework with interactions between the different components of the model.

Identifying potential institutional responses to the issue of the threat of MIS required the identification of existing policy instruments that are relevant to the region of interest. Policy instruments include all international policy instruments, Australian national and state legislation, regulations, policies, plans and other formal and informal instruments that relate to different components of the problem. Moreover, the flexibility of the DPSIR framework means that potential institutional responses can be identified at varying stages of an invasion – from pre-introduction to managing for impacts. Each policy instrument will be examined in the gap assessment to ascertain the level at which the policy instrument covers the indicated policy response. Chapter 7 uses the DPSIR model to describe the system of the MIS threat in an Australian context and undertakes a policy gap analysis.

Chapter 3: Where do they come from? Where do they go? Identifying potential source locations of planktonic marine invasive species arriving at Australia’s Antarctic research bases and subantarctic islands

3.1 Preamble

This chapter explores the passive oceanic connectivity in the Antarctic and Southern Ocean region. Specifically, this chapter aims to understand the capacity of MIS to arrive within biologically relevant time periods at five Australian sites, from other regions of the Antarctic and Southern Ocean. This analysis is based on present-day oceanic current and wind data and does not capture the effect of future climate change.

3.2 Abstract

The Antarctic Circumpolar Current has been heralded as a physical barrier to southward dispersing marine species, as well as enabling the circumpolar distribution of Antarctic marine species. However, this barrier is not as impermeable as originally thought. Ocean models that incorporate fine-scale processes, such as eddies and wave motion, show that there is dispersal from lower latitudes into the higher latitudes of the Southern Ocean. Another barrier to southward marine dispersion is the harsh conditions of the polar marine environment, though as the climate changes these barriers are breaking down. The west Antarctic Peninsula region is one of the fastest warming places on the planet and is likely to be the first place a marine invasion takes place in the Antarctic. Once species have arrived, they may be able to use the Antarctic Circumpolar Current to spread around the continent. Here I use Lagrangian particle tracking to identify potential source locations of planktonic MIS, that could arrive at Australia’s Antarctic research bases and subantarctic islands passively as a result of oceanic processes such as currents, wave action, and wind drift. I found that the subantarctic islands are most at risk from the west Antarctic Peninsula

region, as well as the southern reaches of South America and South Africa; however the number of particles that were carried from each of these source locations to the islands was very low. Conversely, the Australian Antarctic research sites received the same number of particles but from a much-constrained spatial range owing to the finer scale oceanic processes observed along the Antarctic coastline. The results demonstrate that tools such as particle tracking can reveal potential sources and pathways for invasive species. Though the risk of invasive species successfully establishing in the Antarctic region is currently very low, the results show that it is not implausible.

3.3 Introduction

The Antarctic Circumpolar Current and its associated fronts have long been regarded as a key means of biologically isolating the Antarctic continental shelf from adjacent ocean systems (Chown et al., 2015; Clarke et al., 2005). Simultaneously, it has been credited as the reason that some endemic species have distributions around the entire Antarctic continent (Galaska et al., 2017; Norkko et al., 2007). Whilst its role as a biological isolator has been recently refuted (Fraser et al., 2018), this demarcation of warmer temperate waters from cold polar waters does contribute in limiting dispersion into the higher latitudes (Aronson et al., 2011, 2009; Barnes et al., 2006). There is growing concern, however, that new non-native species may arrive on the Antarctic shelf region via either human-mediated transfer (e.g. hull fouling on ships) or climate-mediated range expansions from the Western Antarctic Peninsula region (Hughes et al., 2020; McCarthy et al., 2019).

Changes to the climate are likely to create conditions that are suitable for some MIS (Chapter 4; McCarthy et al., 2019). The West Antarctic Peninsula is one of the fastest warming places on the planet, and though warming in East Antarctica has lagged it is also expected to warm over the coming years (Meredith and King, 2005). This will lead to both a reduction in thermally suitable habitats for many Antarctic native marine species (Griffiths et al., 2017; Ingels et al., 2012) and the creation of more thermally suitable habitats for non-native species (Duffy et al., 2017; Hughes et al., 2020). Coupled with this is the growing number of ships accessing the Southern Ocean, particularly in the Antarctic Peninsula region (McCarthy et al., 2019). These changes combine to increase the likelihood of non-native species being introduced to the Antarctic region.

Examples of MIS have already been recorded within the Antarctic Peninsula region, as well as in East Antarctica (Cárdenas et al., 2020; McCarthy et al., 2019). Despite these occurrences, there is no current evidence of an established population of MIS anywhere within the greater Antarctic region. Recent work has highlighted the growing threat of MIS in the Antarctic (Chapter 4), particularly in the Antarctic Peninsula region (Hughes et al., 2020). The Antarctic Peninsula receives the most ships of any Antarctic region, with a high number of national Antarctic bases in the region, as well as a burgeoning tourism industry (McCarthy et al., 2022, 2019). In contrast, East Antarctica receives limited shipping traffic and tourist voyages.

If we take the view that the Antarctic circumpolar current provides a possible freeway for species to travel around the continent, we might expect that species that are brought into other regions of Antarctica could disperse around to East Antarctica. Indeed, circumpolar distribution is a feature shared by many endemic Antarctic marine species (Brasier et al., 2017), though recent genetic work has revealed that there have been groups which were previously identified as a single species, which are better classified as species complexes (Leese et al., 2015). Despite this, there are examples of Antarctic benthic species with distributions around the continent, such as the echinoid, *Sterechinus neumayeri* (Brockington et al., 2007) and two bivalve genera, *Gaimardia* and *Hiatella* (Güller et al., 2020).

Coastal benthic species often have a planktonic larval phase, with the time spent in the larvae (the pelagic larval duration) and the distance the larvae is carried varying considerably between species (Assis et al., 2015). The pelagic larval duration of a species is linked to ocean temperature, whereby cooler ocean temperatures lead to longer durations in the plankton (O’Connor et al., 2007). Examples of this include common Antarctic echinoderm species, such as *Acondontaster hodgsoni* and *Psilaster charcoti*, which can survive in the plankton for more than a year (Shilling and Manahan, 1994). This pattern of extended pelagic larval duration is also observed in known invasive species, such as the echinoderm *Asterias amurensis*, where larval duration exceeds 100 days at temperatures below 9˚C (Dunstan and Bax, 2007).

Modelling of species dispersal in the marine environment is often in the form of Lagrangian particle tracking models (Chapter 2, p. 33), where individual particles mimic the movement of larvae in the water column (Brasier et al., 2017; Dagestad et al., 2018; Fraser et al., 2018). To ensure realistic results, multiple geophysical forcing data sets are required, such as eddy-resolving ocean circulation models, Stokes drift models that capture the motion of wave propagation, and surface wind data (Dagestad and Röhrs, 2019). The importance of high-resolution data and the incorporation of Stokes drift data into Southern Ocean particle tracking models was highlighted recently to show that connectivity exists across the Polar Front between the warmer waters of the north and the cold polar waters; a feature that was not captured by ocean current data alone (Fraser et al., 2018). While there are models that incorporate larval biology and behaviour (Brasier et al., 2017) advection-only models can be used when there is insufficient biological information. Such models can be used to map theoretical dispersals of larvae over large distances through modelling a large number of particles to mimic the many millions of gametes released by marine species during broadcast spawning events (Brasier et al., 2017; Bruce, 1998).

Here, I use a Lagrangian particle tracking Python package, OpenDrift (Dagestad et al., 2018), to explore how marine environments near the Australian Antarctic bases and subantarctic islands are oceanographically connected to other regions around Antarctica. The aims of this study are to:

1. Identify potential source locations of planktonic marine species within and outside of the Southern Ocean by running back-tracking trajectory models.
2.
    a) Quantify how many particles that leave those potential source locations come within a 100km range of the Australian Antarctic bases and subantarctic islands.

    b) Identify sites that can be part of an early warning system to detect MIS at Australia's Antarctic bases and subantarctic islands.
3. Explore the potential for the Australian Antarctic bases and subantarctic islands to act as an early warning signal for other study sites.

3.4 Methods

3.4.1 Overview

The movement of larvae in the plankton was simulated using particle tracking models that incorporated current, wave, and wind ocean variables over the study period. For Aim 1 I released particles at each of the five study sites and tracked them back through time to identify potential source locations. For Aim 2a I used these potential source locations as release locations for a new set of particles that were run forward in time to quantify the proportion of particles that reached the five study sites. Aim 2b used these results to determine if there are any locations around the Antarctic, subantarctic, or beyond, that could act as an early warning signal of the MIS threat to the study sites. In Aim 3 particles were released at the five study sites and trajectories were run forwards in time to explore the potential for the five study sites to act as an early warning signal to the other study sites of the threat of MIS.

3.4.2 Study sites

Australia hosts three continental Antarctic stations and two subantarctic island locations (Chapter 2.1). The continental stations are Davis (68˚ 34′ 36″ S, 77˚ 58′ 03″ E); Mawson (67˚ 36′ 10″ S, 62˚ 52′ 26″ E); and Casey (66˚ 16′ 55″ S, 110˚ 31′ 39″ E). The stations are all located in East Antarctica and the major oceanographic force in the region is the Antarctic Coastal Current which runs counterclockwise close to the edge of the Antarctic continent. The two subantarctic islands are Heard and MacDonald Islands (herein referred to as Heard Island) group (53˚ 3' 0"S, 72˚ 37' 12"E); and Macquarie Island (54˚ 37′ 12″ S, 158˚ 51′ 40″ E). These subantarctic islands are situated in the Antarctic Circumpolar Current which runs in a clockwise direction around the continent.

3.4.3 Lagrangian Trajectory Model

Lagrangian trajectory models are used to simulate trajectories of objects or organisms given prevailing oceanic or atmospheric processes (Chapter 2, p. 33). Here, I used the offline open source OpenDrift Python framework to simulate the movement of pelagic larvae in the Southern Ocean (available from https://github.com/OpenDrift/opendrift/) (Dagestad et al., 2018). This package is user-friendly and does not require an expert knowledge of Python. It is also highly customizable allowing for user-specified parameters for advections schemes, coastline interactions, and vertical mixing to name a few. The trajectory models used are a 4th order Runge-Kutta scheme to approximate the temporal derivative of the trajectory, where interpolation of the trajectory of a particle between time *n* and time *n*+1 in the velocity field of net ocean movement is estimated four times per timestep, leading to increased accuracy of projected trajectories (van Sebille et al., 2018). When particles hit the coastline, they were moved to their previous location to avoid stranding of particles near the site of release. This allowed particles to leave the Antarctic coastline and provide a much broader suite of locations that the particle could travel through. I used the generic 'OpenOcean' advection only model which does not include parameters for larval behaviour and physiology, to obtain a generalised output of the movement of planktonic larvae that could apply to many species. This approach has been used before where there has been limited information on a species larval biology (Brasier et al., 2017). I constructed a layer of hexbins (diameter = 100 km) for the spatial extent of the model to capture instances where particles passed close to either the five Australian study sites or other Antarctic and subantarctic locations.

3.4.4 Data Sources

I incorporated three ocean forcing datasets to run the trajectory models: ocean currents, wind velocity, and Stoke drift (Table 3.1). Ocean current data was from the HYCOM (Hybrid Coordinate Ocean Model) global circulation model. Wind drift is incorporated in the OceanDrift module by default with a value of 2% of wind velocity, however as wind drift acts variably on ocean particles depending on the depth of the particle and is influenced by Stokes drift, I randomly assigned wind drift to be a proportion of wind velocity between 1% and 3% (Dagestad and Röhrs, 2019). Wind velocity data was from the NCEP (National Centres for Environmental Prediction) Global Atmospheric Model. Stokes drift data from the Copernicus global analysis forecast model was incorporated as it has been shown that including Stokes drift data increases accuracy in particle trajectory studies (Dagestad and Röhrs, 2019; Fraser et al., 2018).

Table 3.1 Ocean current, Stokes drift, and wind velocity data used to build the particle tracking models, with links to the respective web access link.

Variable	Model/Dataset	Resolution	Access
Currents	HYCOM GLBy0.08 exp 93.0	1/12°	https://tds.hycom.org/thredds/catalog.html
Wind	NCEP Global Atmospheric Model/Best Time Series	1/2°	https://pae-paha.pacioos.hawaii.edu/thredds/catalog.html
Stokes drift	Global analysis forecast wav_001_027	1/12°	https://resources.marine.copernicus.eu/products

3.4.5 Scenarios

All scenarios were run with 1,000,000 particles being released each season (winter, spring, summer, autumn) within a 100 km radius of each study site. Particle trajectories were calculated every three hours for a period and were run for a period of 365 days. Results were output every seven days.

Aim 1: Identify potential source locations of planktonic marine species within and outside of the Southern Ocean by running back-tracking trajectory models.

For Aim 1 I was interested in finding potential source locations of planktonic larvae that could reach the Australian Antarctic research bases and subantarctic islands (see *Study Sites*). Particles that represent planktonic larvae were released at each of the five study sites for each season and tracked backwards through time. Particles were released January 15 (Austral summer), April 15 (Austral autumn), July 15 (Austral winter), and October 15 (Austral spring) in 2020 and backtracked for 12 months from each release date. All particle locations over the 12-month period for each season were collated to show all potential source locations and pathways. A heat map was produced showing the number of particles that passed through any one hexbin over the 12-month period to identify where the highest numbers of particles originated from or passed through.

The results thus far have indicated the theoretical maximum range of potential sources of planktonic marine species spread around the Southern Ocean in relation to the study sites of interest. However, the actual pelagic larval duration of most marine species is considerably less than a year. For example, the pelagic larval duration of the invasive Northern Pacific sea star (*Asterias amurensis*) is estimated to be up to 120 days (Dunstan and Bax, 2007) and the bivalve *Mytilus chilensis* can remain in the plankton for up to 45 days (Toro et al., 2006). Therefore, I also provide a snapshot of the positions of particles for two high profile MIS, the Northern Pacific sea star, *Asterias amurensis*, and the bivalve, *Mytilus chilensis*, at their respective maximum pelagic larval durations.

Aim 2: a) Quantify how many particles that leave those potential source locations come within a 100 km range of the Australian Antarctic bases and subantarctic islands; b) Identify sites that can be part of an early warning systems to detect MIS at Australia's Antarctic bases and subantarctic islands.

For Aim 2 I was interested in quantifying how many particles would reach the Australian Antarctic bases and subantarctic islands from the potential source locations identified in Aim 1. Where potential source locations were in within 100 km of each other, I combined them into one source reference location. Where there were no bases or islands identified as source locations, or they were already located close to the Australian Antarctic bases or subantarctic islands, no simulations for Aim 2 were run. Particles were released at each of the source reference locations on January 15, April 15, July 15, and October 15, 2019, and tracked forwards in time for 12 months from each release date. All particle locations over the 12-month period for each season were collated. Using a 100 km diameter buffer I identified the percentage of particles that passed within the buffer around the Australian Antarctic bases and subantarctic islands.

Aim 3: Explore the potential for the Australian Antarctic bases and subantarctic islands to act as an early warning signal for the other study sites.

For Aim 3 I explored the potential for the Australian Antarctic bases and subantarctic islands to act as an early warning of imminent threat to the other Australian Antarctic bases and subantarctic islands. Particles were released from each of the study sites on January 15, April 15, July 15, and October 15, 2019, and tracked forwards in time for 12 months from each release date. Using a 100 km buffer I identified how many particles were carried near the Australian Antarctic bases and subantarctic islands.

3.5 Results

Aim 1: Identifying potential source locations of marine planktonic species that arrive at Australia's Antarctic bases and subantarctic islands

Particles that arrive at the subantarctic islands originated from a much larger spatial range, compared to those that arrived at the Australian Antarctic bases (Figures 3.1 – 3.5). The tracks for the continental sites generally remained near the Antarctic coastline with high numbers of particles (Figures 3.1 – 3.3). The tracks for the subantarctic islands covered a larger spatial extent but the number of particles travelling through each hexbin was lower, with maximum densities a magnitude of order lower than for the bases (Figures 3.4 – 3.5).

Particles tracked in reverse from Casey station generally followed the anticlockwise Antarctic Coastal Current, with the highest densities staying well within this current near the coastline to the east of Casey station and extending into the Ross Sea (Figure 3.1). There are relatively few Antarctic stations along this stretch of East Antarctic Coastline, with only one permanently occupied station, France's Dumont d'Urville, being on the coast between Casey Station and the Ross Sea. Although the highest concentrations of particles are to the east of Dumont d'Urville, this station could represent a suitable site to act as a sentinel.

There are three permanent stations within the Ross Sea that particles could originate from: the Republic of Korea's Jang Bogo station, the United States McMurdo Station, and New Zealand's Scott Base Station, though the latter two are within 10 km of each other and will be treated as one location for the purposes of this study. Although the highest densities of particles were found to have originated from along the Antarctic coastline, there were a small number of particles that could have originated from the outside the coastal zone in the Antarctic Circumpolar Current from as far away as Heard Island.

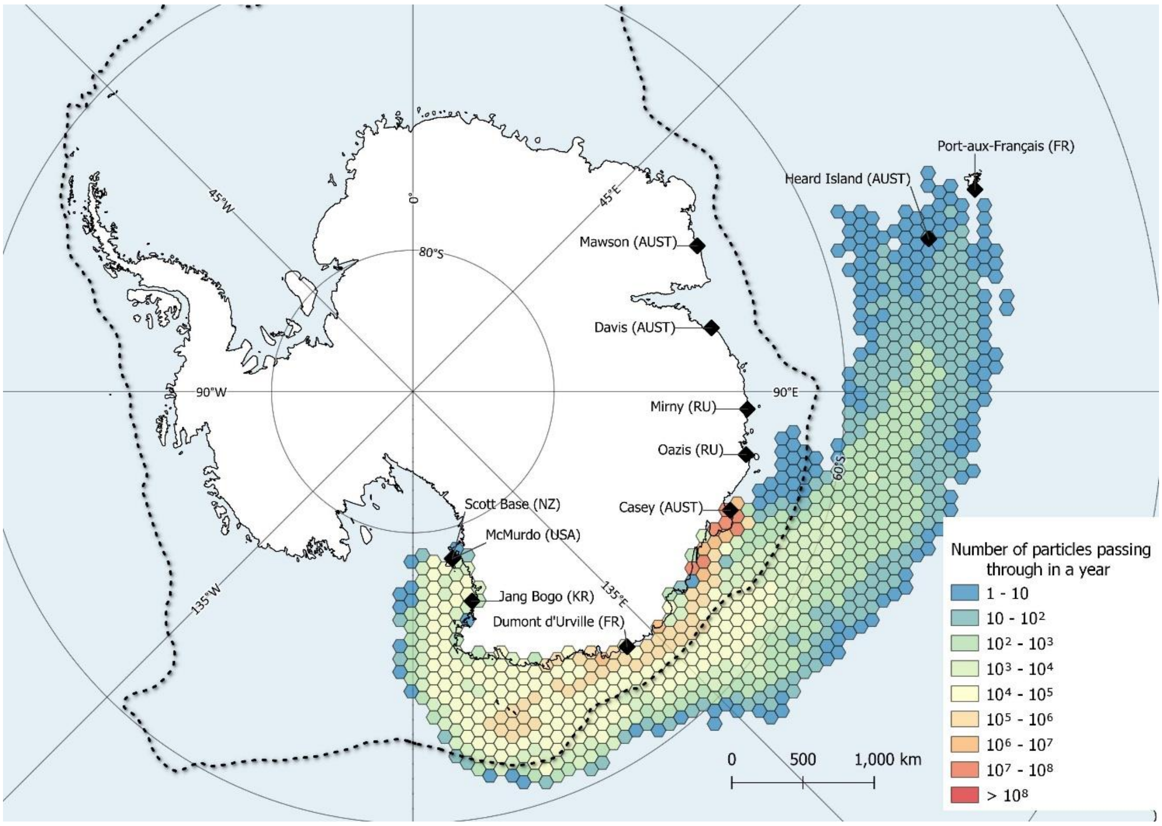

Figure 3.1 Number of backtracked particles released from Casey station over a period of one year. Hexbins are 100 km diameter and values indicate the cumulative number of particles that have passed through that bin over the period of one year, inclusive of all seasons. One million particles were released in each season (total = 4,000,000) and the time step output is 7 days, making a total of 212 million particle location points over the one-year period. Dotted line is the southern boundary of the Antarctic Circumpolar Current.

Unlike Casey, particles that arrived at Davis Station have most likely originated from nearby regions within Prydz Bay (Figure 3.2). There were a very small number of particles which originated from oceanic regions near the boundary of the Antarctic Coastal Current and the Antarctic Circumpolar Current.

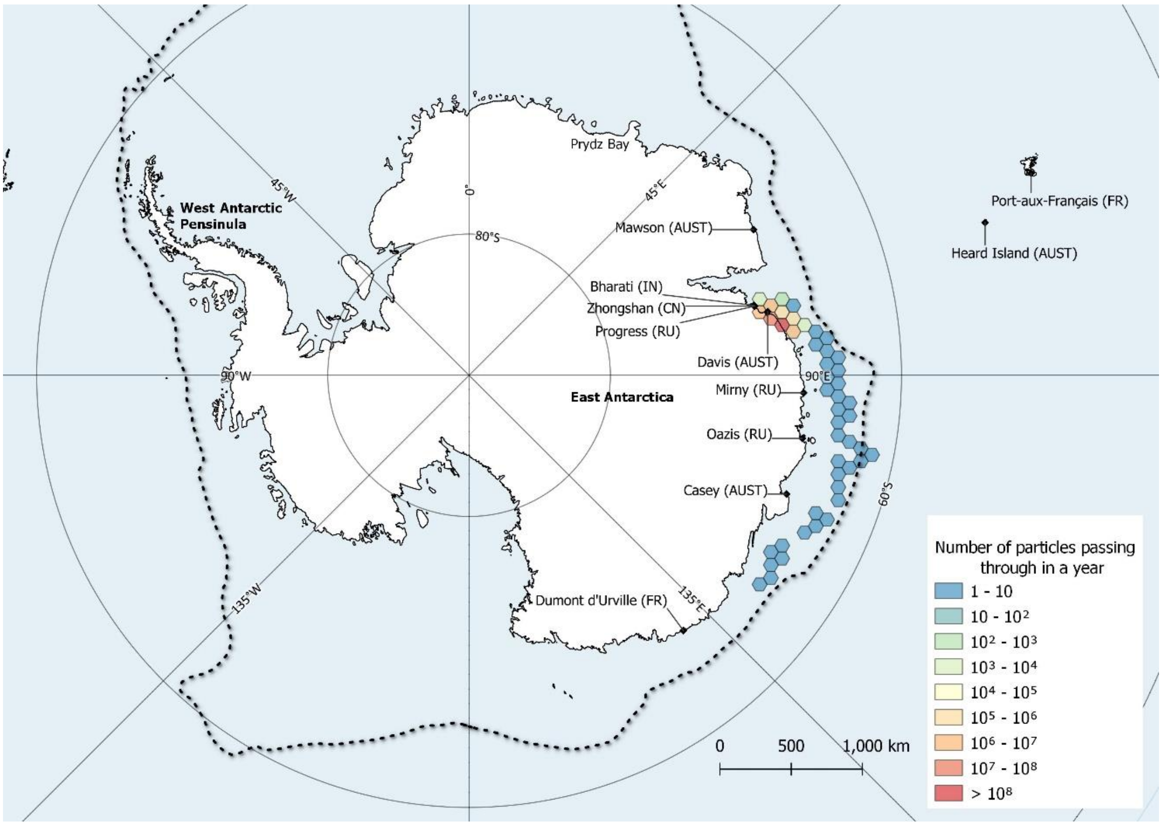

Figure 3.2 Number of backtracked particles released from Davis station over a period of one year. Hexbins are 100 km diameter and values indicate the cumulative number of particles that have passed through that bin over the period of one year, inclusive of all seasons. One million particles were released in each season (total = 4,000,000) and the time step output is 7 days, making a total of 212 million particle locations over the one-year period. Dotted line is the southern boundary of the Antarctic Circumpolar Current.

Similar to Davis station, the particles which arrived at Mawson Station originated from within the Prydz Bay region (Figure 3.3). Here, the spatial extent of particle origins was more reduced than Davis, and no particles had crossed from the Antarctic Circumpolar Current. There were some particles that reached Davis station from Mawson and passed by the three stations to west of Davis Station: Bharati station (India), Zhongshan station (China), and Progress station (Russia).

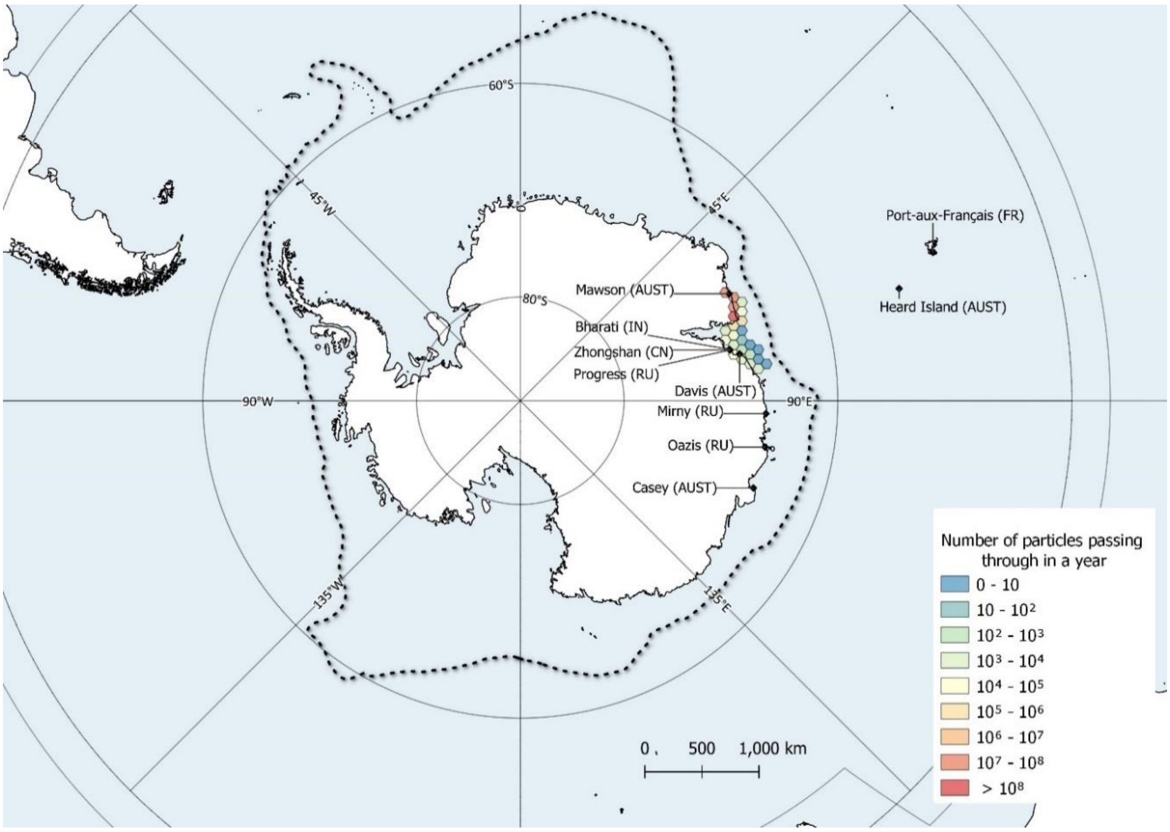

Figure 3.3 Number of backtracked particles released from Mawson station over a period of one year. Hexbins are 100 km diameter and values indicate the cumulative number of particles that have passed through that bin over the period of one year, inclusive of all seasons. One million particles were released in each season (total = 4,000,000) and the time step output is 7 days, making a total of 212 million particle locations over the one-year period. Dotted line is the southern boundary of the Antarctic Circumpolar Current.

Particles that arrived at the Australian subantarctic islands originated from much greater distances than those that arrived at the continental stations (Figures 3.4 and 3.5). Though the particles travelled further, the larger spatial area covered meant that the number of particles passing through each hexbin was a magnitude of order lower in the highest selected hexbins when compared to the continental stations. For both of the subantarctic sites, potential source locations included the Antarctic Peninsula region and the southern tip of South America. The highest concentrations of particles originated from within the subantarctic and around the Antarctic islands. For Macquarie Island, a small number of particles originated from two of the Australian Antarctic stations, Davis, and Mawson. Further a higher concentration of particles originated from Heard Island, indicating that these three locations could represent source locations of planktonic species. Further, South

Africa was shown as a potential source site of particles reaching Macquarie Island, though the number of particles was very low.

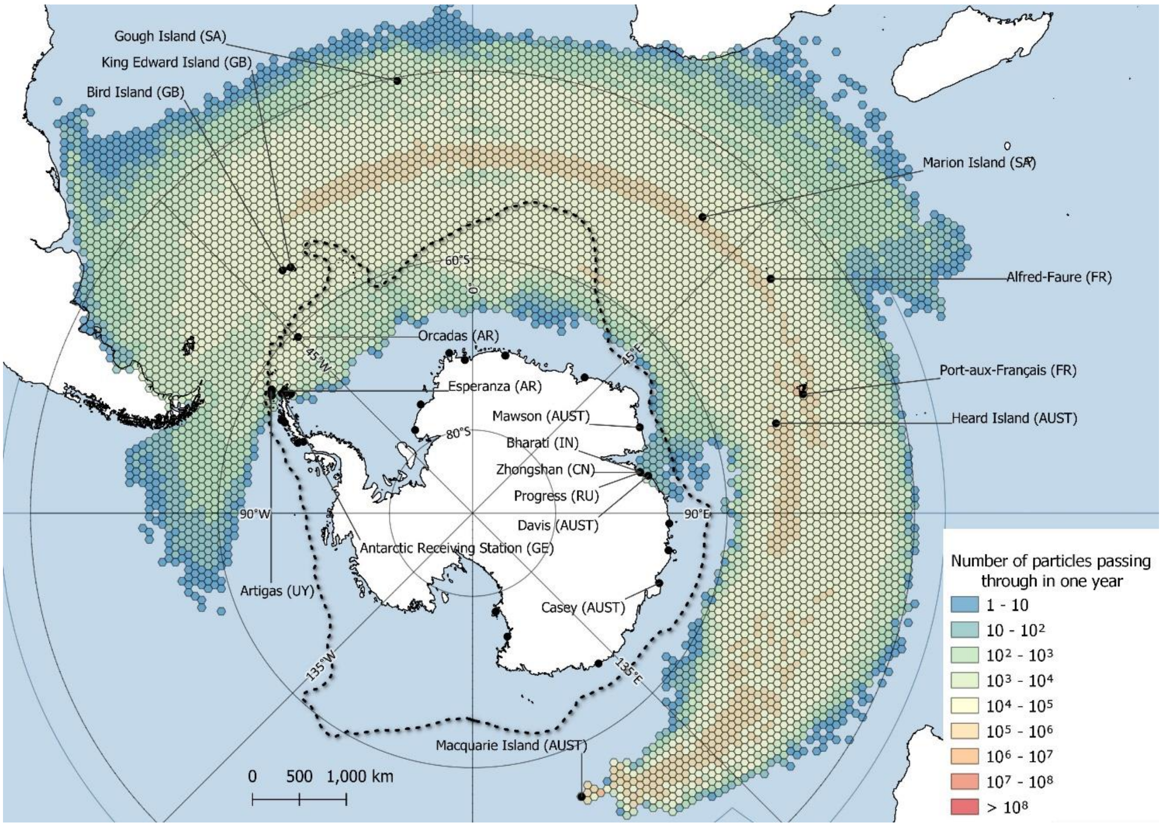

Figure 3.4 Number of backtracked particles released from Macquarie Island over a period of one year. Hexbins are 100 km diameter and values indicate the cumulative number of particles that have passed through that bin over the period of one year, inclusive of all seasons. One million particles were released in each season (total = 4,000,000) and the time step output is 7 days, making a total of 212 million particle locations over the one-year period. Dotted line is the southern boundary of the Antarctic Circumpolar Current.

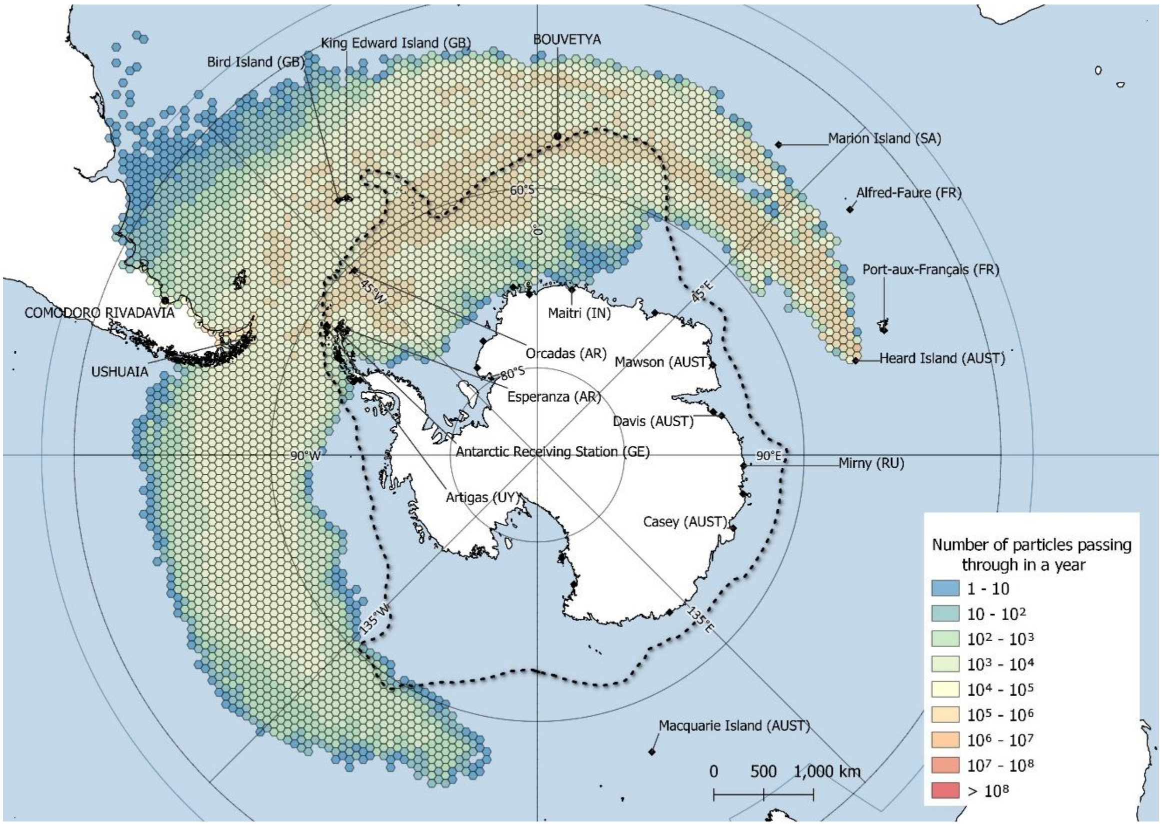

Figure 3.5 Number of backtracked particles released from Heard Island over a period of one year. Hexbins are 100 km diameter and values indicate the cumulative number of particles that have passed through that bin over the period of one year, inclusive of all seasons. One million particles were released in each season (total = 4,000,000) and the time step output is 7 days, making a total of 212 million particle locations over the one-year period. Dotted line is the southern boundary of the Antarctic Circumpolar Current.

Using the Northern Pacific sea star (*A. amurensis*) and the Chilean mussel (*M. chilensis*) as case studies (Figures 3.6 – 3.8), there is a considerable reduction in number of potential source locations of MIS to the Australian Antarctic research bases and subantarctic islands, compared to the year-long simulation. Assuming that there are no increases in pelagic larval duration for either *M. chilensis* or *A. amurensis*, there is no connectivity between Casey and other Antarctic research stations within the maximum 120-day pelagic larval duration (Figure 3.6). There is also a distinct lack of shallow coastal environments in this region, with only a few small patches (less than 1 km^2) loosely dispersed over the route (Southwell et al., 2021). The high number of particles that have remained near the release site (Casey Station) indicates that larvae have been retained in the area.

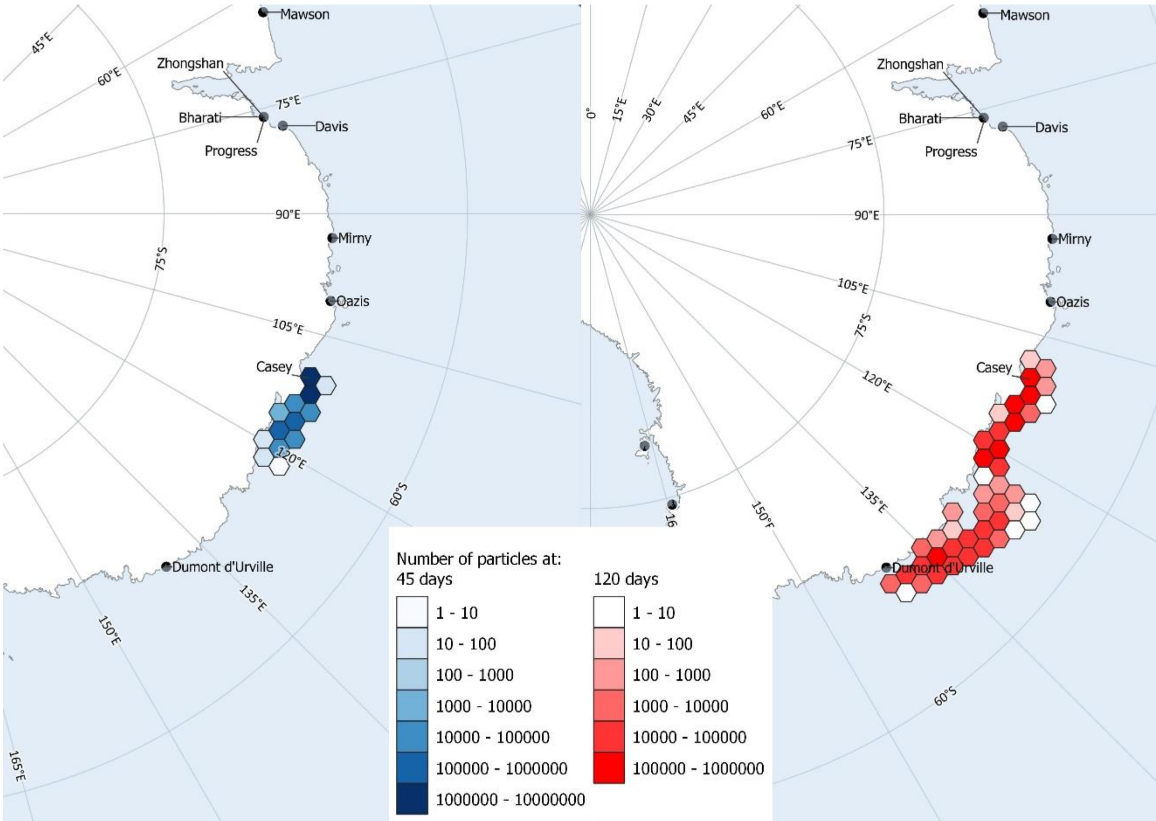

Figure 3.6 Position and number of particles released at Casey Station after being tracked backward in time for 45 days (blue) and 120 days (red).

Particles that arrived at Macquarie Island did not pass by any other islands within the maximum 45-day pelagic larval duration of *M. chilensis* (Figure 3.7). However, when particles were simulated to arrive in Macquarie Island within the maximum 120-day pelagic larval duration of *A. amurensis*, the particles could have originated from as far away as the Kerguelen Plateau, which encompasses Heard Island and the Port-aux-Français research base on Îles Kerguelen. Unlike Casey Station, there was no retention of particles at Macquarie Island.

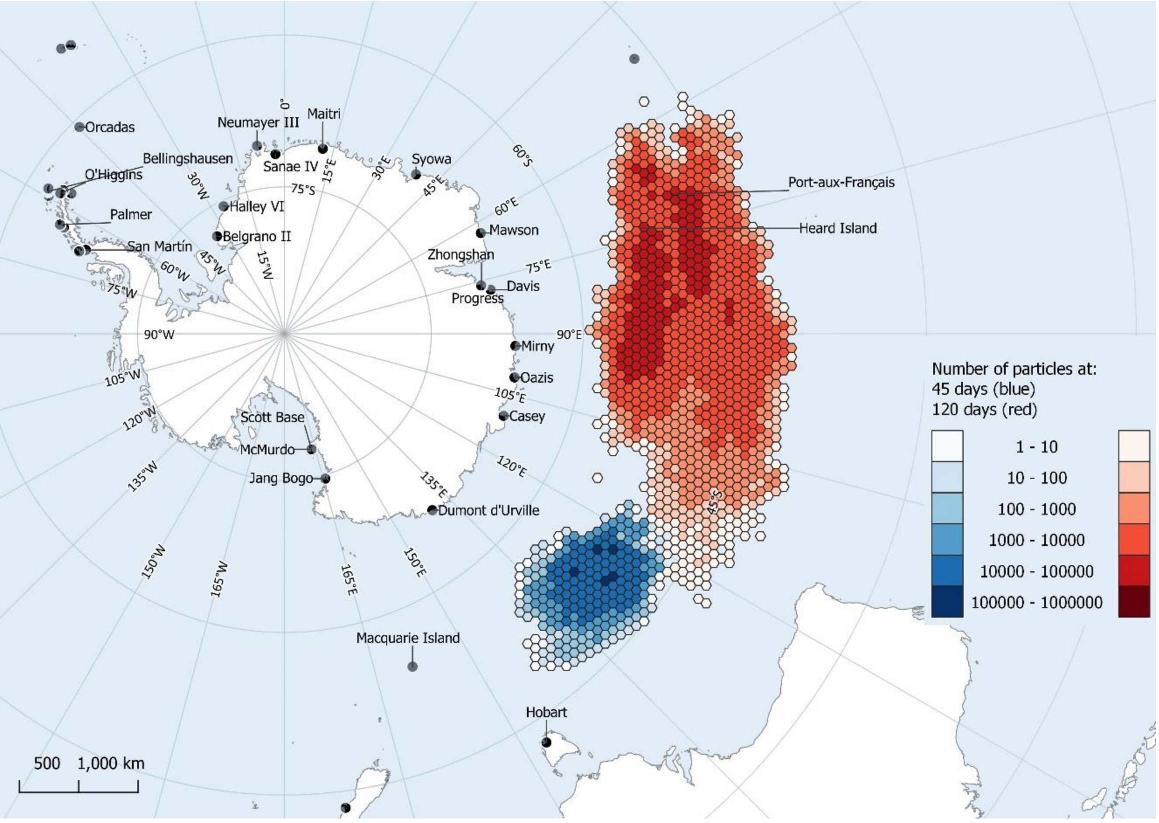

Figure 3.7 Position and number of particles released in a backward simulation at 45 days (blue) and at 120 days (red) from Macquarie Island.

Similarly, particles released in backward simulations from Heard Island did not pass by any other island within the maximum pelagic larval duration of *M. chilensis* (Figure 3.8). When the particles were released from 120 days the only other island that fell within the path was the uninhabited Norwegian subantarctic island, Bouvetøya. Similar to Macquarie Island, no particles were retained at Heard Island.

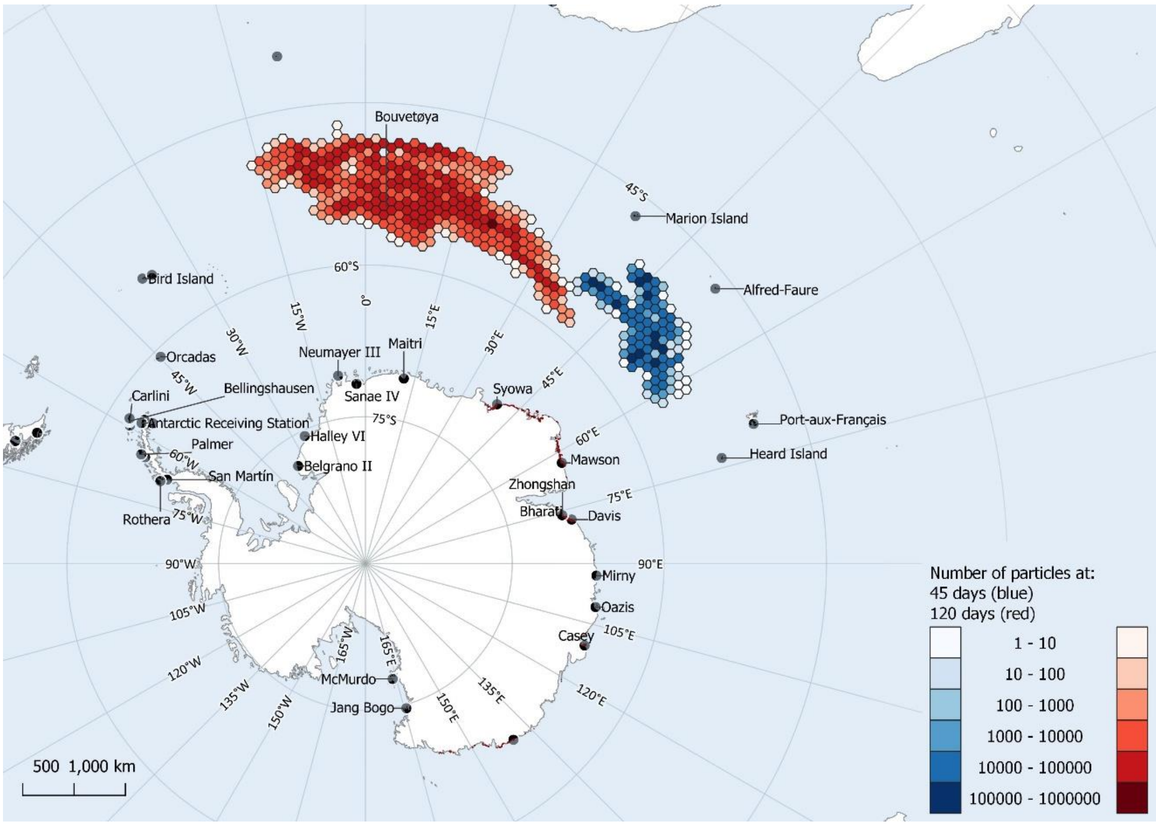

Figure 3.8 Position and number of particles released in a backward simulation for 45 days (blue) and 120 days (red) from Heard Island.

Aim 2: Quantification of particles from potential source locations and identification of any of these sites that may act as an early warning signal for imminent threat of invasion to the study sites.

Twelve general regions were highlighted as potential source locations for one or more of the Australian Antarctic and subantarctic sites (Table 3.2). Source location names are indicative of the area only, as there are multiple stations or islands in close proximity to each other, particularly in the Antarctic Peninsula region and the southern tip of South America. There were no source locations outside out the region of the Australian Antarctic research stations or subantarctic islands for either Davis or Mawson Stations. Although three potential source locations were identified for Casey Station, the number of particles from those three sites that reached Casey Station was very small, and in some cases the forward tracked models from the source reference locations, namely Jang Bogo Station and Scott Station, did not show any particles reaching Casey Station. Heard Island was the most connected to other regions of the Antarctic and subantarctic, in terms of number of potential source locations and the number of particles to reach the islands. Even so, the number of particles reaching Heard Island from any one site was still very low.

Table 3.2 The percentage of particles (from a total of 212 million) that passed within 100 km of the Australian Antarctic research bases and subantarctic islands over a period of one year after being released from source locations identified in Aim 1. Where multiple stations were in a similar geographical area, they have been combined into a single source reference location.

Source reference location	Casey	Davis	Mawson	Macquarie Island	Heard Island
Antarctic Receiving Station					2.17
Artigas Station					0.29
Bird Island				0.04	0.32
Comodoro Rivadavia					< 0.01
Dumont d'Urville Station	0.02			< 0.01	
Esperanza Station					0.23
Jang Bogo Station	< 0.01[a]				
Kerguelen Island				0.17	< 0.10
Marion Island				0.03	< 0.10
Orcadas Station				< 0.01	0.29
Scott Station	< 0.01[a]				
Ushuaia				< 0.01	0.02

[a] No particles reached Casey station after being dispersed from this station in the source location forward particle tracking simulations. However as there were particles that were tracked in reverse from Casey to this station, there is a slight risk that particles from this station could reach Casey station had I released more particles.

Aim 3: Australian Antarctic and subantarctic sites as early warning signals for the other Australian study sites

Four of the five Australian sites have some capacity to act as an early warning signal to other Australian sites of the MIS threat. However, the connectivity between these sites was very low (Table 3.3). Connectivity was strongest between Heard Island and Macquarie Island, though only 0.13% of particles that were released from Heard Island came within 100 km of Macquarie Island. There was no instance in my simulations where Macquarie Island represented a source location of particles arriving at the other Australian sites, and conversely Davis Station and Casey Station were never the recipient location from the other Australian sites. Also note that these figures are reflective of a 12-month pelagic larval duration, which is significantly longer than most species spend in the plankton, and so the true connectivity between these sites is likely to be considerably lower.

Table 3.3 Connectivity between the Australian Antarctic research bases and subantarctic islands. Values indicate the percentage of particles that have been released from the source locations and arrived within 100km of the recipient locations.

Source location \ Recipient location	Casey	Davis	Mawson	Macquarie Island	Heard and McDonald Islands
Casey		0	< 0.01	< 0.01	0
Davis	0		< 0.01	0	0
Mawson	0	0		0	< 0.01
Macquarie Island	0	0	0		0
Heard and McDonald Islands	0	0	0	0.13	

3.6 Discussion

In this study I explored the passive oceanic connectivity between nearshore environments near Australia's Antarctic research bases and subantarctic islands to determine potential source locations of MIS if such species were to establish in the greater Antarctic and subantarctic region. Generally, the subantarctic islands were connected to the greatest number of potential source locations throughout Antarctica and the subantarctic, though the density of particles that arrived at either of these islands was very low. Both Davis Station and Mawson station were highly disconnected from the greater Antarctic region, with most particles arriving at these sites originating from within the Prydz Bay region of East Antarctica, where the bases are located. Casey Station has connections from the east through to the Ross Sea region, as well as up into the subantarctic. The greater distances between potential source locations for the subantarctic islands, compared to the Antarctic sites, is due to their location within the strong eastward flowing Antarctic Circumpolar Current. Conversely, the Antarctic sites lay within the weaker, westward flowing Antarctic Coastal Current and are subject to localised oceanic processes such as the Prydz Bay gyre which can lead to retention of recirculated water (Brasier et al., 2017; Moffat et al., 2008). These differing oceanic processes lead to a situation where particles are dispersed wider and at lower densities in the Antarctic Circumpolar Current, as opposed to shorter distances but at higher densities near the Antarctic coastline.

I identified 12 potential source locations of planktonic MIS, including subantarctic islands, the West Antarctic Peninsula region, and the southern tip of the South American continent. These were sites where a higher density of particles backtracked to and are in close proximity to another base. These source locations therefore do not capture all possible source locations, particularly for the subantarctic islands. The simulation indicates there are weak connections between Macquarie Island and the southern coast of South Africa, as well as much of the south-eastern coast of South America. These areas are both home to a number of MIS, and some of these species, such as the *Mytilid* mussel group, have been indicated as a high threat to the Antarctic and subantarctic regions (see chapter 4) (Hughes et al., 2020).

Using *A. amurensis* and *M. chilensis* mussels as examples, I showed that particles that arrived at Casey Station would only originate from very small and sparse outcrops of coastal habitat within the maximum pelagic larval duration for each of these groups. Similarly, for particles that arrived at Heard Island, the only coastal intercept within the maximum pelagic larval duration of *A. amurensis* was the uninhabited island, Bouvetøya. From Macquarie Island, it is feasible for *A. amurensis* larvae to travel in the plankton from the islands atop the Kerguelen Plateau, including Heard Island. In contrast, it would be unlikely that the *Mytilid* group would be able to complete a journey from any subantarctic island to Macquarie Island or Heard Island as plankton. Though this mussel group has an estimated pelagic larval duration of 45 days (Toro et al., 2006), laboratory experiments have shown that this genus is capable of prolonging its pelagic larval duration by delaying metamorphosis (Pechenik et al., 1990).

It is important to note that this means that advection-only trajectory models are likely to show an overestimation of the spread of larvae in a real world setting (Cowen and Sponaugle, 2009). There are many species which have planktonic larvae, and the physiology and behaviour of these larvae is different between species. Some species only inhabit the plankton for hours, whereas others can remain in the plankton for more than a year (Shilling and Manahan, 1994). This pelagic larval duration therefore influences how far a species can travel, with longer durations associated with longer distance dispersal (Gaines et al., 2015). There is also a high degree of seasonality in species spawning times, meaning that there can be considerable differences in the oceanic processes affecting species that spawn at different times of the year (Kough and Paris, 2015). There are also species-specific ontogenetic behaviours that may influence the movement and settlement of larvae, such as vertical migration (Leis, 2007; Paris and Cowen, 2004), horizontal swimming (Faillettaz et al., 2018; Staaterman et al., 2012), and orientation behaviours based on environmental biotic and abiotic cues (Bottesch et al., 2016; Faillettaz et al., 2015; Leis et al., 2014; McQuaid and Phillips, 2000; Paris et al., 2013). Integrating physiological and behavioural data into particle tracking models can enhance the accuracy of predictions; however for many species, these parameters are poorly understood (Brasier et al., 2017).

While it was important to identify potential source locations of marine non-native species plankton, it was also important to quantify how many of the particles that were released at the source locations actually made it to the Australian Antarctic and subantarctic study sites. I found that a very low percentage of particles released from the source locations actually arrived within 100 km of these five study sites. Coupled with the fact that most species have a shorter pelagic larval duration than the 12 months simulated in this study, the level of threat from each of the potential source locations will be less than reported here. Similarly, there were very few particles that were released from the five Australian Antarctic sites and subantarctic islands which travelled to one or more of the other four study sites.

Although these low percentages may suggest only a weak connectivity between potential source locations and the study sites, it is important to remember that a small percentage of a very large number is still a large number. For example, each female *A. amurensis* can release between 0.4 and 15.5 million eggs each spawning season (Bruce, 1998; Hatanaka and Kosaka, 1959; Kas'yanov, 1988). If we take the example of Heard Island as a potential source location, with 0.13% of particles coming within 100 km of Macquarie Island, it suggests that between 520 and 20,150 larvae could reach the island within the known pelagic larval duration of 120 days from each spawning female. However, this would rely on the majority of larvae surviving the 4-month journey, which is inconsistent with larvae ecology.

As seen in the results, the numbers of particles passing through a hexbin are highest near the point of particle release and very rapidly disperse to numbers which are magnitudes of order lower as they are carried away from the release point. In the context of planktonic larvae, this means that generally the highest risk source locations are those that are close to the recipient location. This phenomenon is common amongst planktonic species, even those with a long pelagic larval duration. For example, the marine gastropod, *Austrolittorina cincta*, with a pelagic larval duration of 6 – 8 weeks largely settles within 5 km of the spawning site with relatively few reaching distances of 100 km from the spawning site (Salinas-de-León et al., 2012). This suggests that even though species may be able to travel quite extensive distances, the propagule pressure at those long distances is reduced compared to closer sites. However, this does not always hold true when strong ocean currents are present (Gilg et al., 2007; Planes et al., 2009). In terms of the flow of currents around Antarctica, this would suggest that species close to the coastline in the Antarctic Coastal Current (such as around the continental stations) may settle out of the plankton at relatively shorter distances than in the Antarctic Circumpolar Current (such as around the subantarctic islands) where the current moves faster.

A key factor in determining whether a non-native species will establish in a new region is the propagule pressure, or number of individual non-native organisms, at the new environment (Lockwood et al., 2009). Here, I have shown that only weak connections exist between potential source locations and the study sites, and that plankton ecology means that connections are strongest only in closely located sites. This leads to very low propagule pressure at most recipient sites. Taking this into account, the greatest oceanic connectivity to the study sites is likely between sites in the Prydz Bay region, which includes Davis Station and Mawson Station. However, the lack of connectivity to regions outside of the Prydz Bay region over biologically relevant timelines means that non-native species would have to be brought into the region by anthropogenic means. Currently, this would be unlikely, due to the extent of sea-ice in the region that ships would have to travel through to reach the stations of Prydz Bay. Traversal through sea-ice acts as a natural hull cleaner, by scouring clean most external hull surfaces (Lee and Chown, 2009), though there may still be a significant risk of non-native species introductions from the recessed regions of hulls (Lee and Chown, 2007). Also, shipping activity in East Antarctica is considerably lower than other regions of the world, including the West Antarctic Peninsula region (McCarthy et al., 2022, 2019).

The outcomes presented here provide a general guide to the threat of marine non-native species being carried to the Australian Antarctic and subantarctic sites. It is important to note that this study only looked at connectivity between bases and/or islands and does not completely encapsulate the whole suite of potential settlement sites for planktonic species. There are thousands of isolated shallow coastal ecosystems around the Antarctic continent, each a potential habitat for non-native species (Southwell et al., 2021). The focus on the bases and islands is largely because the identification of non-native species is likely to correspond with where there are people and ships, and because shallow water environments are rare and sparsely distributed, particularly in East Antarctica (Southwell et al., 2021). Future work could look at the connectivity between all shallow coastal regions to develop connectivity matrices based on time in the plankton as well as particle transfer between all the sites.

This study does not consider the impact of climate change on oceanic processes around the greater Antarctic region. Strengthening of westward winds will increase the strength of the Antarctic Circumpolar Current, and warming will lead to changes in sea ice volume and extent, potentially opening up new oceanic pathways between regions of the Antarctic and subantarctic (Hellmer et al., 2012; Rintoul et al., 2018). This could include increasing the number of sites that are connected within biologically relevant time periods by way of faster oceanic currents, or redirection of ocean currents into previously disconnected regions, particularly areas that are currently under ice shelves (Hellmer et al., 2012). These considerations will be important in the future, particularly as non-native species have yet to establish in the region.

There is currently only limited oceanic connection between most study sites and potential source locations of non-native marine species, particularly when considering biologically relevant time periods of plankton duration of known MIS. Non-native species are currently more likely to arrive by anthropogenic means (see Chapter 4). Therefore, the ability of an Antarctic or subantarctic site to act as an early warning signal to an imminent threat of passive non-native species introduction is very limited in most cases. Further work is needed to understand how the connections between sites will alter in the face of climate change. Non-native marine species in the Antarctic are an emerging threat, and tools such as particle tracking can help us to understand the ways in which an invasion could occur.

3.7 Acknowledgements

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3.8 Code Availability

An example of the code used to run the particle tracking simulations in available on github.

https://github.com/OakesHolland/Antarctic_PPT

DOI: 10.5281/zenodo.6573238

Chapter 4: Hull fouling marine invasive species pose a very low, but plausible, risk of introduction to East Antarctica in climate change scenarios

4.1 Preamble

In the context of the thesis as an overall risk assessment this chapter explores the ability of known hull fouling marine invasive species from around the world to survive in the harsh marine environments near Australia's Antarctic research stations and subantarctic islands. It is important to note that this chapter does not ascertain that these species *will* become established and invasive in these regions. Rather, it identifies species which *may* survive in the region based on environmental conditions, thus removing one barrier to invasion.

This chapter was published in the Q1 journal, *Diversity and Distributions*, as an Open-Access research paper which can be accessed at https://doi.org/10.1111/ddi.13246. The contents of the chapter herein match that of the published manuscript, though the content has been altered to one-column (from two in the manuscript) and figure captions and table headings have been updated to their relevant position within the thesis. The supplementary material of this paper has been moved to the overall thesis Appendices (Appendix A) and in text referencing herein has been updated to reflect this change. Cited literature has been collated into a single *References* section, following Chapter 8 of this thesis. I have retained the use of the plural 'we' for this chapter as this is how it appears in the published paper.

Keywords: climate change, invasion biology, gradient boosting, hull fouling, Northern Pacific Sea star, ports, machine learning, XGBoost, East Antarctica, subantarctic

4.2 Statement of Contribution of Co-Authors for Thesis by Published Paper

The authors listed below have certified that:

1. they meet the criteria for authorship and that they have participated in the conception, execution, or interpretation, of at least that part of the publication in their field of expertise;
2. they take public responsibility for their part of the publication, except for the responsible author who accepts overall responsibility for the publication;
3. there are no other authors of the publication according to these criteria;
4. potential conflicts of interest have been disclosed to (a) granting bodies, (b) the editor or publisher of journals or other publications, and (c) the head of the responsible academic unit, and
5. they agree to the use of the publication in the student's thesis and its publication on the QUT's ePrints site consistent with any limitations set by publisher requirements.

The reference for the publication associated with this chapter is:

Holland, O., Shaw, J., Stark, J.S. & Wilson, K.A. (2021) *Hull fouling marine invasive species pose a very low, but plausible, risk of introduction to East Antarctica in climate change scenarios*. Diversity and Distributions. 27, 973-988. https://doi.org/10.1111/ddi.13246

Contributor	**Statement of Contribution**
Oakes Holland Signed: Date:	Conducted the research, developed code for analysis, wrote the manuscript and edited the manuscript as per feedback from co-authors.

Justine Shaw	Supervised the research, provided feedback on manuscript.
Jonathan S. Stark	Supervised the research, provided feedback on the manuscript.
Kerrie A. Wilson	Supervised the research, provided feedback on the manuscript.

Principal Supervisor Confirmation

I have sighted email or other correspondence from all Co-authors confirming their certifying authorship. (If the Co-authors are not able to sign the form, please forward their email or other correspondence confirming the certifying authorship to the GRC).

Name	Signature	Date

4.3 Abstract

Aims: To identify potential hull fouling marine invasive species that could survive in East Antarctica presently and in the future.

Location: Australia's Antarctic continental stations: Davis, Mawson and Casey, East Antarctica; and subantarctic islands: Macquarie Island and Heard and McDonald Islands.

Methods: Our study uses a novel machine learning algorithm to predict which currently known hull fouling MIS could survive in shallow benthic ecosystems adjacent to Australian Antarctic research stations and subantarctic islands, where ship traffic is present. We used gradient boosted machine learning (XGBoost) with four important environmental variables (sea surface temperature, salinity, nitrate, and pH) to develop models of suitable environments for each potentially invasive species. We then used these models to determine if any of Australia's three Antarctic research stations and two subantarctic islands could be environmentally suitable for MIS now and under two future climate scenarios.

Results: Most of the species were predicted to be unable to survive at any location between now and the end of this century; however four species were identified as potential current threats, and five as threats under future climate change. *Asterias amurensis* was identified as a potential threat to all locations.

Main conclusions: This study suggests that the risks are very low, but plausible, that known hull fouling species could survive in the shallow benthic habitats near Australia's East Antarctica locations and suggest a precautionary approach is needed by way of surveillance and monitoring in this region, particularly if propagule pressure increases. Whilst some species could survive as adults in the region, their ability to reach these locations and undergo successful reproduction is considered unlikely based on current knowledge.

4.4 Introduction

Ports worldwide are recognized as ecologically disturbed areas; most are heavily polluted, have undergone significant habitat modification, and are repositories of marine invasive species (MIS). Many MIS have established populations in regions well beyond their original distributions and are now found in all oceans except the Southern Ocean

(McCarthy et al., 2019; Ruiz et al., 1997). This region has thus far been protected from this threat by environmental barriers, such as extreme cold temperatures and other harsh environmental conditions, the physical barrier created by the Antarctic Circumpolar Current and its associated polar fronts, and the deep oceans around Antarctica (Aronson et al., 2011, 2009; Barnes et al., 2006). The Antarctic region also receives far less shipping traffic than other regions of the world, thereby limiting the propagule pressure exerted in this region (McCarthy et al., 2019). However, as the world's climate changes and human presence increases in the region, these barriers to invasion are breaking down or being bypassed (Cheung et al., 2009; Duffy et al., 2017).

Most Antarctic marine regions are experiencing gradual warming (Ducklow et al., 2007; Meredith and King, 2005). Coupled with changes in temperature are other environmental changes, such as: a lowering of salinity; increased acidification; changes in productivity; and changes in the timing, extent, and thickness of sea ice (Stark et al., 2019). There is limited understanding of how these changes could influence the ability of MIS to survive and establish in the Southern Ocean (though, see Aronson et al., 2009). Whilst the threat of MIS in Arctic regions has been investigated (Ware et al., 2014), there has been a paucity of such research in the Antarctic, particularly in the East Antarctic and subantarctic regions (however, see Byrne, Gall, Wolfe, & Agüera, 2016; Lee & Chown, 2009; Lewis, Riddle, & Smith, 2005).

Antarctica is extremely remote in a global context. Non-anthropogenically assisted introductions of MIS are consequently limited to infrequent occurrences (Lewis et al., 2005), however recent molecular work has shown that drifting species are able to cross the various oceanic fronts of the region to reach the continent (Fraser et al., 2018), and that non-native kelp rafts have carried invasive species as passengers to an West Antarctic island (Avila et al., 2020). Shallow coastal marine ecosystems along the Antarctic continental shelf are relatively uncommon, fragmented, and separated by long distances (Clark et al., 2015). Corresponding ice-free coastal areas (within 5 km of the coast) are also rare, comprising approximately 0.06% of the continent (Brooks et al., 2019). This means there is limited habitat available for the establishment of shallow water MIS around the continent, nevertheless, the suitable habitat coincides with the locations of Antarctic research stations and their associated presence of ships (Stark et al., 2019). These small areas of habitat still have intrinsic biodiversity values as they support many endemic and novel species (Stark et al., 2019).

The presence of MIS in a novel ecosystem does not necessarily infer that an invasion has occurred, which is when a population becomes established and persistent (Arthur et al., 2015). Although no established populations of MIS have been found in the Antarctic region, five species of decapod MIS have been found free-living in the Antarctic marine environment: *Emerita sp.; Hyas araneus; Rochinia gracilipes; Halicarcinus planatus; and Pinnotheres sp.* (Aronson et al., 2015b; McCarthy et al., 2019; Tavares and De Melo, 2004; Thatje and Fuentes, 2003). Further, a settlement of the mussel *M. cf platensis* was discovered in the shallow subtidal region of the South Shetland Islands, however subsequent surveys indicate that this species no longer persists in the region (Cárdenas et al., 2020). Whether the presence of these species represents recent incursions or persistent populations is under debate, due mainly to the poor fossil record of the group and a lack of comprehensive biodiversity surveys in this region (Griffiths et al., 2013). Whilst terrestrial invasions in the subantarctic region are relatively well documented and researched (Chown et al., 2012; Frenot et al., 2005; Shaw et al., 2010), the Antarctic region, and particularly the marine realm, has remained understudied to date (however, see Hughes et al., 2020). A better understanding of the present day and future threat of MIS that could affect this region is required (Hughes and Pertierra, 2016; McCarthy et al., 2019).

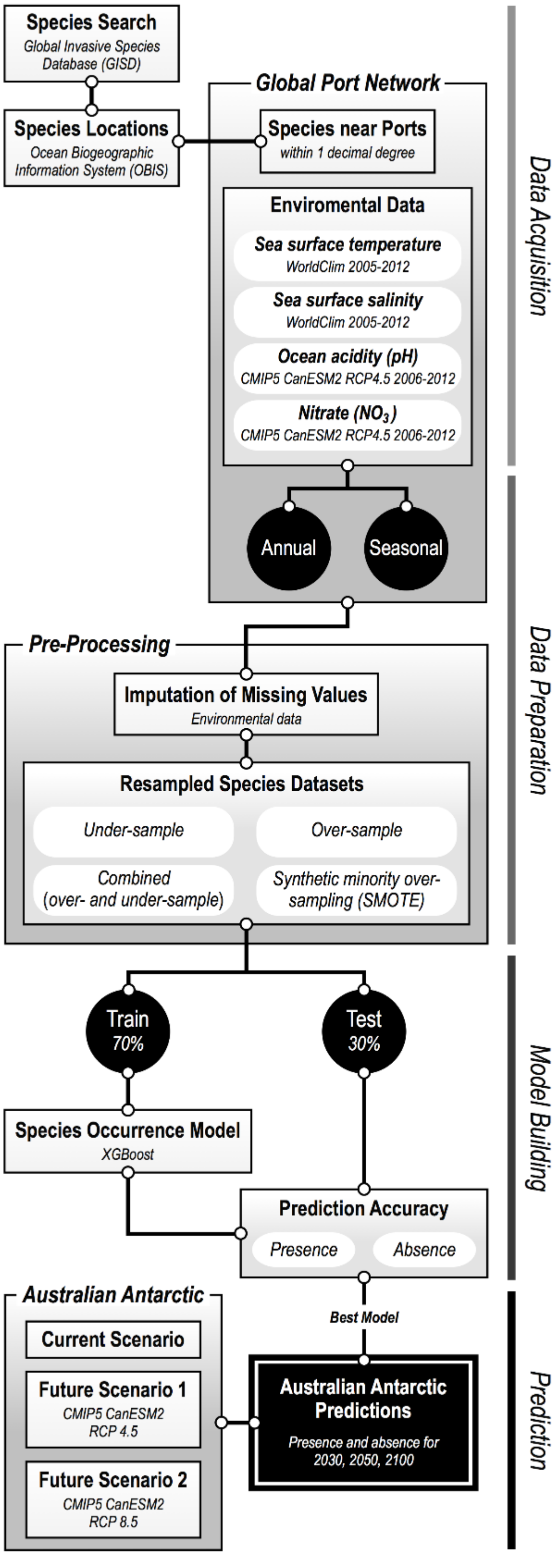
The international shipping network has been identified as they key driver of MIS transfer globally (Clarke et al., 2017; Molnar et al., 2008) with ports around the globe having established populations of MIS (Keller et al., 2011). Historically ship ballast had been the greatest contributor to ship-based MIS transfer, but this risk has declined with modern conventions and policies (Drake and Lodge, 2007). Hull fouling has received less attention as a source of MIS transfer, even though it likely poses a similar risk, if not greater, than ballast water (Drake and Lodge, 2007). Ships that travel to the Antarctic continent often have periods of passage through sea-ice which acts as a natural hull cleaner, removing most attached fouling (Hughes and Ashton, 2017; Lee and Chown, 2009; Lewis et al., 2004). However, there are several protected niche areas in ships, such as sea chests (a rectangular or cylindrical recess in the hull of a ship), which are not subject to sea-ice scour and represent a potential pathway for MIS introductions to occur (Chan et al., 2015; Hughes and Ashton, 2017; Lee and Chown, 2007). Further, climate change is expected to alter sea ice distribution which could reduce the efficacy of ice scour to remove fouling from ship (Hughes and Ashton, 2017; Stammerjohn et al., 2012). Finally, travel to the subantarctic islands may have no preceding periods of sea-ice traversal, increasing the risk of MIS introductions.

The arrival of MIS on the Antarctic continental shelf could be devastating to the endemic species which have been essentially isolated for the past 25 million years (Tavares and De Melo, 2004). It is well established that preventing incursions from occurring is more cost-effective than management of an introduced species (Finnoff et al., 2007; Hanley and Roberts, 2019; Leung et al., 2002). This is particularly true for remote locations where it may be more difficult or expensive to undertake surveillance and management of an invasive species, and the potential biodiversity impacts are greater (Rout et al., 2011). Successful eradications of MIS are rare, occurring only in spatially limited regions with ample access to resources after early detection (Giakoumi et al., 2019). Currently, there is no systematic MIS surveillance in the Antarctic region and the ability to detect and to act quickly is hindered by the isolation and harsh conditions, making eradication an unfeasible plan of action in this region.

This study explores how changes predicted to occur in the marine environment around Australia’s Antarctic stations, in East Antarctica, and Australia’s subantarctic islands could influence the vulnerability of these regions to hull fouling MIS establishment, both now and into the future. We use machine learning to develop models of known hull fouling marine invasive species presence and use these models to predict whether species could survive in the shallow benthic habitats adjacent to Australia’s Antarctic research stations and subantarctic islands. This will offer insight into what species are threats for future invasions and a potential focus for management.

4.5 Methods

The methods for this study can be broken down into four key steps: 1) data acquisition; 2) data preparation; 3) model building; and 4) prediction. The flow and components of these steps are shown in Figure 4.1 and described in detail below.



4.5.1 Study region

The Australian Antarctic research stations and subantarctic islands are located in the East Antarctic. There are three continental research stations: Davis (68° 34′ 36″ S, 77° 58′ 03″ E); Mawson (67° 36′ 10″ S, 62° 52′ 26″ E); and Casey (66° 16′ 55″ S, 110° 31′ 39″ E). Australia also manages two sub-Antarctic island groups: Heard and McDonald Islands (herein referred to as Heard Island due to their close proximity) group (53° 3' 0"S, 72° 37' 12"E); and Macquarie Island (54° 37′ 12″ S, 158° 51′ 40″ E). (Figure 4.2). This project is specifically focussed around these five discrete locations, as the logistic operations and science research are managed by the Australian Antarctic Division (under the Australian Government's Department of Agriculture, Water, and the Environment).

Figure 4.1 Methods flow chart of the four key steps of this study and the components that underpin each step.

4.5.2 Global port network

As we were interested in hull fouling species, we used the global port network to build our model of environmental suitability for each species. The locations of ports worldwide are described in the World Port Index (National Geospatial-Intelligence Agency, 2017). There are 3,645 ports in the global port network, with most located in the northern

hemisphere (n = 3,156). Of these, 439 are located on freshwater lakes, or had insufficient data, and thus were excluded from further analysis. The total number of ports used for model building was 3,206; with 2,757 in the northern hemisphere compared with 449 in the southern hemisphere. Most global ports are located in temperate regions (65.22%, n = 2,091), with less than half appearing in tropical regions (31.07%, n = 996), and relatively few ports in polar regions (3.71%, n = 119) (Spalding et al., 2007). The location of the Australian Antarctic stations and sub-Antarctic islands were manually inputted using coordinates from Google Earth.

Figure 4.2 Location of the Australian continental research stations: Mawson, Davis and Casey; and subantarctic islands: Heard and McDonald Island Group and Macquarie Island. Adapted from the Quantarctica 3 dataset (Matsuoka et al., 2018).

4.5.3 *Marine invasive species*

A global list of known invasive species associated with marine and brackish habitats was obtained from the Global Invasive Species Database (GISD – iucngisd.org). Using information from the GISD along with primary literature, species which did not have an association with fouling were removed from the list. This yielded a list of 160 species, with most (n = 112) belonging to the *Didemnum spp.* group which was not resolved to species level within GISD. However, most *Didemnum spp*. show no conclusive evidence of being invasive in the literature, thus we limited further investigation of this group to *D. vexillum*, which is known to be invasive.

We used the Ocean Biogeographic Information System mapper function (OBIS – https://mapper.obis.org/) to input the fouling species to find occurrence data of their worldwide distribution. It is important to note that these records are likely incomplete records of the species' total distribution for reasons such as, but not limited to, biased global sampling (Phillips et al., 2009). The OBIS location data was overlayed with the location of world ports, and the Australian Antarctic stations and sub-Antarctic islands, and species were matched to ports where they occurred within 1.0 decimal degree of the port. Data was condensed by port, so that only one record per species per port was taken, to avoid spatial autocorrelation that would occur with heavily sampled species in specific regions (Assis et al., 2015; Elith and Leathwick, 2009). By limiting our pool of potential pseudo-absences to the global port network we, in effect, apply a similar bias to that observed in the presence data, which enhances the ability of our simulations to model the environmental conditions that are suitable for each of our species (Phillips et al., 2009). Where there were fewer than 10 occurrences of a species worldwide, that species was excluded from further analysis as there were insufficient points for gradient boosting analysis. Furthermore, we limited the species predictions for the three Antarctic sites to those species which had a recorded distribution that included sub-freezing temperatures. Predictions for the subantarctic were limited to those species which had recorded distributions at a minimum temperature of 11°C or less to align with maximum temperatures expected in the subantarctic by the climate change scenarios. This resulted in a list of 33 species total, with a subset of 20 species which experience sub-freezing temperatures somewhere in their current range (Table 4.1).

4.5.4 Environmental variables for model building

Our study incorporates four environmental variables to determine habitat suitability for the suite of MIS, as discussed below. They are sea surface temperature (SST), sea surface salinity (SSS), nitrate (NO_3) and ocean acidity (pH). These variables were aggregated into three sets: A – SST only; B – SST plus SSS and NO_3; and C – all four variables. Where possible, it has been suggested to use multiple environmental variables with a statistical method that can adequately account for collinearity, such as regression trees (Hayes and Barry, 2008).

Sea surface temperature and sea surface salinity were selected for use as they are well known to influence species distributions (Barry et al., 2008; Ware et al., 2014). Present day sea surface temperature and sea surface salinity data were obtained using the World Ocean Atlas version 2 (http://www.nodc.noaa.gov/OC5/woa13/) at 1° resolution using objectively analysed means. World Ocean Atlas environmental variables were monthly averages over the period 2005 – 2012.

Due to the nature of sea ice dynamics, associated under-ice algae, and post-sea-ice breakout algal blooms, we could not adequately calculate present day and future predictions of chlorophyll. Instead, we used nitrate as a surrogate measure of productivity as it is often the limiting nutrient in coastal systems (Howarth and Marino, 2006). Nitrate has also been shown to be a stronger driver of species distributions than chlorophyll (Bosch et al., 2018). Ocean acidity (measured by pH) is known to have negative impacts on some species which rely on calcium carbonate for skeleton and shell formation, particularly in the early stages of development (Guinotte and Fabry, 2008; Karelitz et al., 2017). Evidence that pH is becoming a key driver of marine species distributions is accumulating in the Southern Ocean region, where changes in acidification are occurring faster than originally predicted (Guinotte and Fabry, 2008; Hancock et al., 2020; Roden et al., 2013). Present day data for nitrate and acidity were obtained from the CMIP5 (Coupled Model Intercomparison Project phase 5) CanESM2 (Canadian Earth System Model second generation) as monthly averages for the years 2006 – 2012 (Figures A1.1 – 1.5).

Iron is the limiting nutrient in much of the greater Southern Ocean region yet is excluded from our set of environmental variables. Recent work has shown that iron is rarely limited in shallow ocean environments due to deposition from terrestrial regions, particularly where there are areas of ice-free land – typical of the subantarctic and areas in the Antarctic where research stations are located on the coast (Duprat et al., 2019; Stark et al., 2019).

Table 4.1 Hull fouling species considered for modelling for the two subantarctic or three Antarctic locations. Species appearing in bold are only considered for the subantarctic. Species indicated for modelling in the Antarctic region are species which currently have a part of their distribution in areas which experience sub-freezing temperatures.

Kingdom	Phylum	Class	Species
Animalia	Annelida	Polychaeta	*Alitta succinea* (Leuckart, 1847)
			***Ficopomatus enigmaticus* (Fauvel, 1923)**
			Sabella spallanzanii (Gmelin, 1791)
	Arthropoda	Malacostraca	*Carcinus maenas* (Linnaeus, 1758)
			Charybdis japonica (A. Milne-Edwards, 1861)
			Hemigrapsus sanguineus (De Haan, 1835)
			Rhithropanopeus harrisii (Gould, 1841)
		Maxillopoda	***Elminius modestus* (Darwin, 1854)**
	Bryozoa	Gymnolaemata	*Bugula neritina* (Linnaeus, 1758)
			***Schizoporella errata* (Waters, 1878)**
			Schizoporella unicornis (Johnston in Wood, 1844)
			***Watersipora subtorquata* (d'Orbigny, 1852)**
	Chordata	Ascidiacea	*Ascidiella aspersa* (Müller, 1776)
			Ciona intestinalis (Linnaeus, 1767)
			***Didemnum vexillum* (Kott, 2002)**
			Styela clava (Herdman, 1881)
			***Styela plicata* (Leseur, 1823)**
	Echinodermata	Asteroidea	*Asterias amurensis* (Lutken, 1871)
	Mollusca	Bivalva	*Crassostrea gigas* (Thunberg, 1793)
			Geukensia demissa (Dillwyn, 1817)
			***Musculista senhousia* (Benson, 1842)**
			Mya arenaria (Linnaeus, 1758)
			***Mytilopsis leucophaeata* (Conrad, 1831)**
			***Mytilus galloprovincialis* (Lamarck, 1819)**
			***Rangia cuneata* (G. B. Sowerby I, 1832)**
		Gastropoda	*Crepidula fornicata* (Linnaeus, 1758)
			Littorina littorea (Linnaeus, 1758)
			Rapana venosa (Valenciennes, 1846)
Chromista	Ochrophyta	Phaeophyceae	***Undaria pinnatifida* ((Harvey) Suringar, 1873)**
Plantae	Chlorophyta	Ulvophyceae	*Codium fragile* ((Suringar) Hariot, 1867)
	Rhodophytina	Florideophyceae	***Gracilaria vermiculophylla* ((Ohmi) Papenfuss, 1967)**
			Hypnea musciformis ((Wulfen) J. V. Lamouroux, 1813)
			***Polysiphonia brodiei* ((Dillwyn) Sprengel, 1827)**

4.5.5 *Environmental variables for predictions*

Predictions for the present day for the five Australian Antarctic and subantarctic sites were made using the same data sources used for model building (see *Environmental variables for model building*). Future predictions were based on the CMIP5 CanESM2 future climate models for RCP 4.5 and RCP 8.5, which represent a curbing of emissions scenario and a business-as-usual scenario, respectively. The future projections were based on monthly predictions averaged for the years 2026 – 2030 (for 2030 predictions), 2046 – 2050 (for 2050 predictions), and 2096 – 2100 (for 2100 predictions). The RCP 8.5 models stop at October 2099; therefore, the months of November and December were averaged for the period 2096 – 2099 for RCP 8.5 2100 predictions. The CMIP5 CanESM2 model has an oceanic resolution of 1.14˚ latitude and l.4˚ longitude at the poles, with good spatial coverage of the Antarctic region and contains all environmental variables of interest, i.e. sea surface temperature, salinity, nitrate, and pH. We created two sets of environmental variables to explore the effect of seasonality on model outcomes: an annual model which used annual minimums, averages, and maximums of each environmental variable; and a seasonal model where the data were collated into seasonal maximums, averages, and minimums of each environmental variable (Figures A1.1 – 1.5). All regions are expected to experience rises in temperature from ~ 0.5˚C up to 4˚C by the end of century. Conversely, all other environmental variables are expected to decrease by the end of the century. The Supplementary Material (Appendix A) contains detailed environmental change information for each location.

4.5.6 *Data preparation*

The dataset for each species was divided into 'training' (70%) and 'test' (30%) sets using the 'createDataPartition' function in the R package 'caret' (Kuhn, 2019) to ensure the ratio presences to absences was maintained in the 'training' and 'test' datasets. The "class" of interest, in our case the presence of a MIS at a port, was the minority class in a highly imbalanced dataset. This can lead to high levels of classification accuracy due to the majority class being predicted in most cases (Leevy et al., 2018).

As the minority class is often the class we are most interested in, methods have been developed to overcome this imbalanced data - at the data-level and at the algorithm-level (Leevy et al., 2018). These methods can then be used independently, however, improved performance has been shown when data-level and algorithm-level methods are combined (Díez-Pastor et al., 2015). Resampling of the data is a common method for dealing with imbalanced data at the data-level (Díez-Pastor et al., 2015; Leevy et al., 2018). This process improves the ratio between majority and minority classes and there are several methods of resampling available. The resampling methods used in our study include: a) Over-sampling; b) Under-sampling; c) Both over- and under-sampling; and d) Synthetic minority over-sampling (ROSE) using the R packages, 'caret' and 'ROSE' (Kuhn, 2019; Lunardon et al., 2015). A more detailed explanation of the resampling techniques and their performance in this study is found in the Supplementary Material (Appendix A3) of this paper and the associated references. As the ratio of presence to absence data is unique for each species, the resampling was performed for each species, resulting in eight 'training' datasets for each species.

4.5.7 Gradient boosting analysis

Extreme gradient boosting is an ensemble machine learning technique that makes predictions based on combinations of multivariate predictor data producing a specific outcome. For this study we elected to use environmental variables at ports (the predictor variables) to predict the presence or absence of a MIS (the response variable). This method creates an ensemble of decision trees to create a strong classification (or regression) model based on a set of 'weak' classifiers of the response variable. Here we use the 'xgboost' in R (Chen et al., 2019; Kuhn, 2019). This is the first study to use the extreme gradient boosting system to model MIS, and only the second to use it to model any invasive species (Sandino et al., 2018). Despite the paucity of use in invasive species research, extreme gradient boosting is a popular modelling algorithm in many other fields; such as: critical care management (Chang et al., 2019; Zhang et al., 2019); financial fraud detection (Zhou et al., 2018) and credit scoring (Munkhdalai et al., 2019); and satellite image (Just et al., 2018) and astronomical feature classification (Tamayo et al., 2016). XGBoost has consistently outperformed other machine learning algorithms in data science competitions whilst being computational efficient through parallel processing (Chen and Guestrin, 2016). Additional benefits of this gradient boosted algorithm is that it is robust against overfitting, has customizable hyper-parameters, includes cross-validation, and its non-parametric nature makes it useful when working with correlated predictor variables (Shi et al., 2019).

We used the XGBoost algorithm to model the environmental suitability using each species ‘training’ datasets. Model accuracy for each species’ eight ‘training’ datasets was assessed against the ‘test’ subset from the original dataset to determine the predictive capability of each of the four resampling techniques with each of the model types, with a confusion matrix showing the accuracy for each analysis. The resampling technique that was able to predict the presence of a species with highest accuracy, whilst also maintaining a high accuracy in predicting the absence of a species, was deemed the best model. Higher accuracy in predicting the presence of species was given a higher preference as it is likely that the presence data is accurate, whereas absence data are likely to be less accurate as species distribution data is prone to Type II errors (Lobo et al., 2010). Environmental variable importance was recorded for each model to show which environmental variables contributed most to the model. The best model as described here was used to predict the presence of the species at the five Australian Antarctic and subantarctic locations currently; and at 2030, 2050, and 2100 under the two representative concentration pathways RCP 4.5 and RCP 8.5.

An outline of the R code used is available in the Supplementary Material (Appendix A4).

4.6 Results

4.6.1 Present day invasive potential

Of the 33 species investigated, 29 were not predicted to be suited to the environment of any location at the present-day, with only 4 out of 33 species predicting current environmental suitability in at least one location. These were: *Asterias amurensis* (Northern Pacific sea star); *Geukensia demissa* (ribbed mussel); *Hypnea musciformis* (red algae); and *Undaria pinnatifida* (brown algae) (Tables 4.2 & 4.3). The annual model predicted a higher number of species (n = 4) when compared with the seasonal model (n = 2). Further, the choice of variable aggregation strongly influenced the outcome of predictions, particularly in the seasonal model where only the aggregation of all four variables led to predictions of environmental suitability for any species (Figure 4.3).

4.6.2 *Future invasive potential*

Environmental suitability was predicted for 5 of the 33 species that were included in this study (Tables 4.2 & 4.3). Like the present-day predictions, the seasonal model only predicted environmental suitability for two species (*A. amurensis* and *U. pinnatifida*), and again, only when aggregating all four variables. Two of those species (*Charybdis japonica* (Asian paddle crab) and *H. musciformis*) were predicted to be environmentally suited to one location each and only at one time (2030), and in both cases only by the aggregation of all four variables. Predictions of environmental suitability for all five species did not greatly differ between RCP 4.5 and RCP 8.5. Only one species (*A. amurensis*) was common between the five locations, with all three aggregations of variables predicting presence in the annual models at all locations, and only the aggregation of all four variables in the seasonal model at all locations (Figure 4.3). *G. demissa* was common to the continental locations using the annual model, and *U. pinnatifida* was common to the subantarctic locations under both the annual and seasonal models.

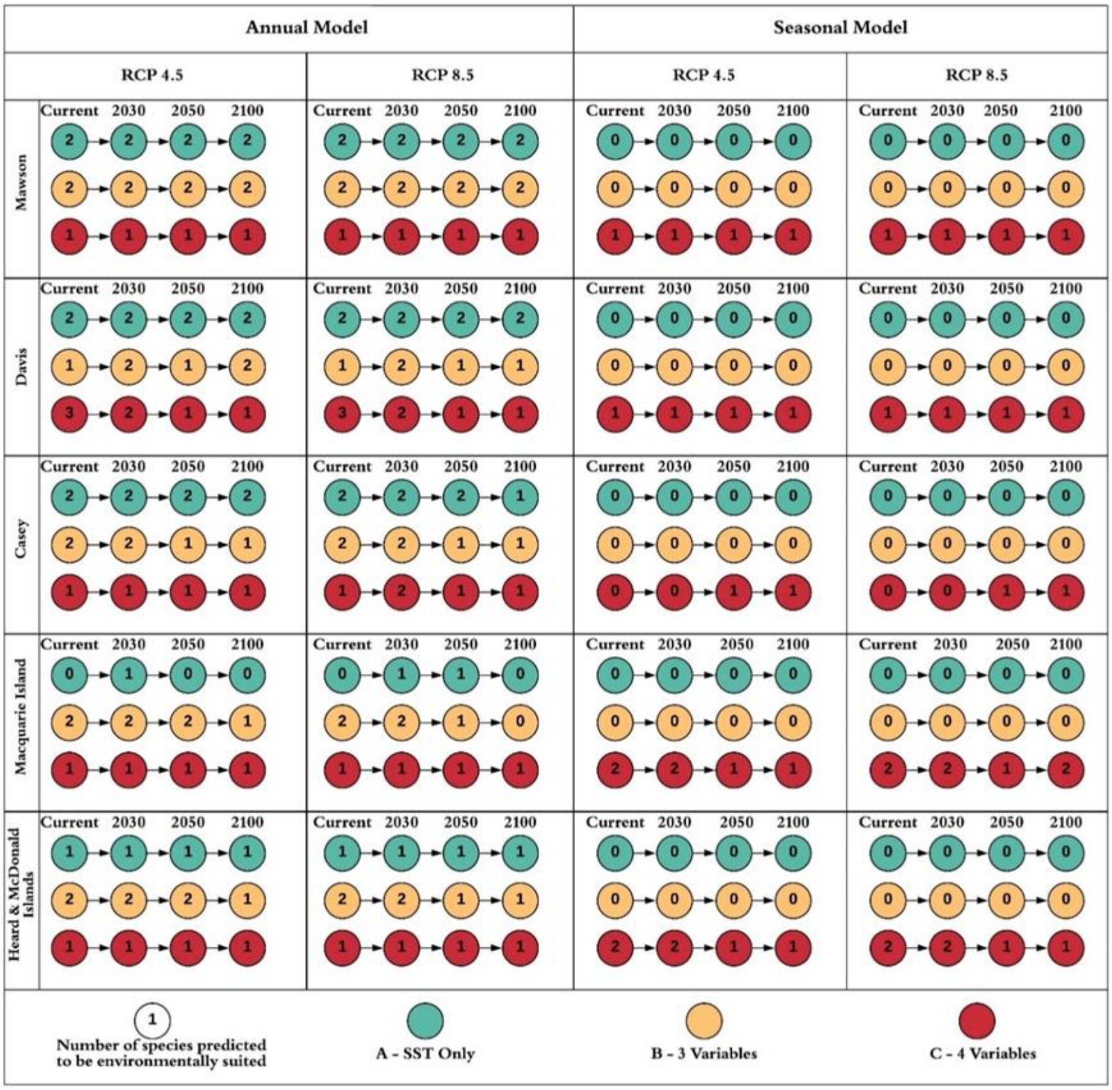


Figure 4.3 Number of species predicted to survive the environmental conditions at each of the five Australian Antarctic and subantarctic locations by model type and environmental variable aggregation. A (blue) is the sea surface temperature only variable aggregation; B (orange) is the sea surface temperature, salinity, and nitrate aggregation; and C (red) is the sea surface temperature, salinity, nitrate, and pH aggregation.

Table 4.2 Species which were predicted to be able to survive in the five Australian Antarctic and subantarctic locations using the annual model for the current day, and at 2030, 2050, and 2100. A is the sea surface temperature only variable aggregation; B is the sea surface temperature, salinity, and nitrate aggregation; and C is the sea surface temperature, salinity, nitrate, and pH aggregation. The environmental variable which contributed most to the model is listed for each species and variable aggregation.

		RCP 4.5			RCP 8.5			Most important variable		
Species	**Current day**	**2030**	**2050**	**2100**	**2030**	**2050**	**2100**	**A**	**B**	**C**
Mawson										
Asterias amurensis	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	Avg SST	Min SST	Max Nitrate
Geukensia demissa	A, B	A, B	A, B	A, B	A, B	A, B	A, B	Avg SST	Avg Nitrate	-
Davis										
Asterias amurensis	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	Avg SST	Min SST	Max Nitrate
Geukensia demissa	A, C	A, B	A	A, B	A, B	A	A	Avg SST	Avg Nitrate	Max Nitrate
Hypnea musciformis	C	C	-	-	C	-	-	-	-	Avg pH
Casey										
Asterias amurensis	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	Avg SST	Min SST	Max Nitrate
Charybdis japonica	-	-	-	-	C	-	-	-	-	Avg pH
Geukensia demissa	A, B	A, B	A	A	A	A	-	Avg SST	Avg Nitrate	-
Macquarie Island										
Asterias amurensis	B, C	A, B, C	B, C	B, C	A, B, C	A, B, C	C	Avg SST	Min SST	Max Nitrate
Undaria pinnatifida	B	B	B	-	B	-	-	-	Avg SST	Avg SST
Heard and McDonald Islands										
Asterias amurensis	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	A, B, C	Avg SST	Min SST	Max Nitrate
Undaria pinnatifida	B	B	B	-	B	-	-	-	Avg SST	-

4.6.3 Model performance

The accuracy to predict both presence and absence was very high for all species and models (Table 4.4). The seasonal models generally outperformed the annual models in accuracy for predicting both presence and absence of MIS. The type of variable aggregation also influenced the model's predictive performance, with aggregation of sea surface temperature, salinity, and nitrate performing best in predicting both presence and absence of the species. The full details of model accuracy for each species is included in the supplementary materials (Appendix A1.6).

Table 4.3 Species which were predicted to be able to survive in the five Australian Antarctic and subantarctic locations using the seasonal model. A is the sea surface temperature only variable aggregation; B is the sea surface temperature, salinity, and nitrate aggregation; and C is the sea surface temperature, salinity, nitrate, and pH aggregation. The environmental variable which contributed most to the model is listed for each species and variable aggregation.

		RCP 4.5			RCP 8.5			Most important variable		
Species	**Current day**	**2030**	**2050**	**2100**	**2030**	**2050**	**2100**	**A**	**B**	**C**
Mawson										
Asterias amurensis	C	C	C	C	C	C	C	-	-	Winter Nitrate
Davis										
Asterias amurensis	C	C	C	C	C	C	C	-	-	Winter Nitrate
Casey										
Asterias amurensis	-	-	C	C	-	C	C	-	-	Winter Nitrate
Macquarie Island										
Asterias amurensis	C	C	C	C	C	C	C	-	-	Winter Nitrate
Undaria pinnatifida	C	C	-	-	C	-	C	-	-	Spring SST
Heard and McDonald Islands										
Asterias amurensis	C	C	C	C	C	C	C	-	-	Winter Nitrate
Undaria pinnatifida	C	C	-	-	C	-	-	-	-	Spring SST

Over-sampling of the minority class produced the most accurate results for the annual model when using the SST only aggregation (n = 21), followed by both over- and under-sampling (n = 9), and under-sampling (n = 3). For the corresponding seasonal model over-sampling was the only resampling technique to produce the most accurate results (n = 33) and was also the case for the seasonal model when using the 3-variable aggregation. The 3-variable annual model was split evenly between over-sampling (n = 12), both over- and under-sampling (n = 12) and under-sampling (n = 9). Under-sampling of the majority class produced the most accurate annual and seasonal models in the 4-variable aggregation (n = 19 and 23, respectively), followed by both over- and under-sampling (n = 8 and 6, respectively) and over-sampling (n = 6 and 4, respectively). There was no case where the ROSE resampling technique resulted in the most accurate results, and often lead to overfitting of the presence class with low classification accuracy for the absence class. The best resampling technique for each species can be found in the Supplementary Material (Appendix A1.6).

Table 4.4 The mean accuracy of model predictions for all species by model type and variable aggregation. Standard error is provided in brackets after mean, with the range provided in the square brackets below.

	Annual Model Mean		Seasonal Model Mean	
Variable Aggregation	Presence	Absence	Presence	Absence
A - Sea surface temperature only	90.48% (± 0.48) [79.41% - 98.41%]	95.84% (± 0.95) [86.88% - 99.89%]	99.42% (± 0.11) [97.88% - 100%]	97.48% (± 0.23) [94.36% - 99.85%]
B - SST, Salinity & Nitrate	92.97% (± 0.92) [79.41% - 100%]	96.09% (± 0.44) [88.62% - 99.19%]	99.58% (± 0.08) [98.45% - 100%]	98.49% (± 0.14) [96.94% - 99.90%]
C - SST, Salinity, Nitrate & pH	91.40% (± 0.01) [71.79% - 97.95%]	93.87% (± 0.01) [81.84% - 99.43%]	92.72% (± 0.01) [73.68% - 100%]	93.17% (± 0.01) [85.73% - 99.68%]

4.6.4 Variable importance

Average sea surface temperature was the most important variable for the annual model (Table 4.5). For the seasonal model, in all aggregated variable sets, average Autumn sea surface temperature was the most important variable for the greatest number of species.

4.7 Discussion

We explored the environmental suitability of the Australian Antarctic region and subantarctic regions for MIS. Our models indicate that up to five of the species examined, are or will be able to survive in at least some locations by the end of the century. Two of these species were predicted to be able to survive in only one or a couple of the four time periods we examined and show different patterns of predicted survival for different regions. These predictions could be indicative of errors in the model predictions or perhaps that the environment is only borderline suitable for the species and warrant further investigation. In any case, a precautionary approach should be used due to the potential, as yet unknown, impacts of MIS in the East Antarctic region (Ojaveer et al., 2014). Further, all governance and management prescriptions for these areas aim to ensure that non-native species do not establish, in order to protect the wilderness values and natural ecosystems (Committee for Environmental Protection, 2019; Commonwealth of Australia, 2014; Parks and Wildlife

Service, 2006), though these are biased towards terrestrial systems and provide little guidance on practical measures to prevent or manage MIS in the Antarctic region.

Of greatest concern are those species which are predicted to survive in the environmental conditions for several continuous time periods. This shows consistent environmental suitability for those species and that they are less likely to represent modelling errors or species which are borderline matches, though it does not suggest that all life stages could successfully survive in the area. Using two different models with three different aggregations of environmental variables allowed us to investigate how sensitive predictions were to the choice of model and variables. Only one of the five species showed agreement between the annual model and seasonal model: *A. amurensis*. This mismatch between the annual and seasonal models may infer that the high seasonality of primary productivity in the Antarctic region, rather than specific extremes of the environmental variables, are the barrier to successful introduction of species.

ANNUAL		**SEASONAL**	
Variable	**No. of species**	**Variable**	**No. of Species**
A - Sea surface temperature only			
Average SST	18	Autumn SST	13
Maximum SST	8	Summer SST	8
Minimum SST	7	Spring SST	8
		Winter SST	4
B - SST, Salinity and Nitrate			
Average SST	12	Autumn SST	13
Maximum Salinity	6	Spring SST	5
Minimum SST	5	Winter SST	5
Maximum SST	4	Summer SST	4
Average Salinity	2	Winter Nitrate	3
Maximum Nitrate	2	Summer Salinity	1
Average Nitrate	1	Autumn Salinity	1
Minimum Salinity	1	Winter Salinity	1
C - SST, Salinity, Nitrate and pH			
Average SST	9	Autumn SST	7
Maximum SST	8	Spring pH	6
Minimum SST	5	Summer SST	5
Average Nitrate	3	Winter SST	4
Maximum pH	3	Spring SST	3
Maximum Nitrate	2	Winter Nitrate	2
Average pH	1	Autumn Nitrate	2
Maximum Salinity	1	Winter pH	1
Minimum Nitrate	1	Winter Salinity	1
		Spring Salinity	1
		Summer Salinity	1

Our results indicate that a common assumption that sea surface temperature is a key barrier to MIS may be flawed. It is important to point out, however, that environmental suitability does not necessarily confer that a species will become established in these locations, as there are other important factors, like propagule pressure and the availability of a mechanism for anthropogenic transfer, that are required for a species to reach a new region.

However, it does highlight that greater effort should be focused on preventing these species from entering the region.

For all regions, the predatory Northern Pacific sea star, *A. amurensis,* has been shown to be environmentally suited using both annual and seasonal models for all time periods and both RCPs (with the exception of the seasonal model applied to Casey presently or in 2030). *A. amurensis* is currently subject to a National Control Plan due to its "...*having significant and potential future impacts on Australia's marine environment, social uses of the marine environment and the economy*" (Aquenal Pty Ltd, 2008b, p. 11). This species displays considerable phenotypic plasticity and can alter spawning times to coincide with local conditions (Buttermore et al., 1994; Byrne et al., 1997; Ling et al., 2012). This species has an introduced range that includes Canada and Alaska, indicating its ability to tolerate cold conditions and the presence of sea ice (Byrne et al., 2016). It is not a species that is commonly associated with hull fouling, however there is evidence of hull settlement of juvenile *A. amurensis* in the Derwent River, Tasmania, Australia, along with an adult of the species being found in the sea chest of a vessel (Hewitt et al., 2004, 1999). More detailed information regarding this species is found in the Supplementary Materials (Appendix A2.1).

U. pinnatifida is another species which was predicted to be environmentally suited to the subantarctic islands now and at 2030 by an annual model (with 3 aggregated variables) and now and at 2030 and 2050 by a seasonal model (with 4 aggregated variables). The present day introduced range encompasses Australia, New Zealand, Europe, Argentina, and California (James et al., 2015). In Australia, it is also subject to a National Control Plan (Aquenal Pty Ltd, 2008b). *U. pinnatifida* is a poor competitor that struggles to establish in stable environments but thrives in disturbed environments (James and Shears, 2016a; Valentine and Johnson, 2003). Physical disturbance is predicted to increase in Antarctic and subantarctic benthic ecosystems via more frequent iceberg scouring (Barnes and Souster, 2011; Peck et al., 2005), less sea ice opening up new areas to iceberg scour, and increased winds leading to larger waves in coastal areas (Stark et al., 2019). As disturbance via iceberg scour can occur throughout the year and recovery from a scour event can take many years, there is potential for this species to find areas of suitable habitat if they are transported to the region from known source locations in Australia and New Zealand (Aquenal Pty Ltd, 2008b; Stark et al., 2019). In addition, other forms of anthropogenic disturbance around Antarctic stations, such as pollution (Stark et al., 2014), may enhance the chances of other species establishing in these areas if local species are intolerant of such disturbances (Piola and Johnston, 2008). This species attaches to suspended objects and vessel hulls at, or just below, the waterline and is more commonly encountered on vessels which are moored for extended periods of time (Hewitt et al., 1999) – such as by Australia's previous icebreaker ship *Aurora Australis* in Hobart for overwintering. This species was also recently predicted to be a high risk of invading the Antarctic Peninsula region in an expert analysis of the invasive species threat to the Peninsula region (Hughes et al., 2020). More detailed information regarding this species is found in the Supplementary Materials (Appendix A2.2).

Geukensia demissa is another species which was predicted to be environmentally suited to the three continental stations by the annual model. The occurrence records of this species on the Ocean Biogeographic Information System indicate that this species is found in the United States, northern Spain, and on the Antarctic Peninsula (King George Island). There is no information in the literature regarding the Antarctic finding of this species. The OBIS record makes reference to a published study (Tatián et al., 2008) as a bibliographic reference, however this paper mentions *G. demissa* as a comparative species for measuring carbon flux in suspension feeders. Therefore, this species entry was either: a) incorrectly identified as *G. demissa*, or b) the first record of this MIS in the greater Antarctic region. The prediction of this species as being environmentally suited to Antarctica should be treated with scepticism until confirmation of the taxonomy of the specimen from the Antarctic Peninsula region.

Environmental suitability is not the only factor determining if a species can become established in a new environment. Even though adults may be able to survive in an environment this does not necessarily infer that successful establishment will occur, as was recently highlighted in the case of the non-persistent population of *Mytilus cf. platensis* in the South Shetland Islands (Cárdenas et al., 2020). Recent work found that there is a risk of *A. amurensis* becoming a threat to the Australian subantarctic before the end of the century, however it did not find the same for the Antarctic (Byrne et al., 2016). The study experimentally found thermal limits for different life stages from the Tasmanian population of *A. amurensis*, and that these limits are narrower for larval development than for adult survival. Nevertheless, this species displays high phenotypic plasticity, and populations from other regions of the world may have the capacity to successfully reproduce at colder temperatures than the Tasmanian population (Buttermore et al., 1994; Byrne et al., 1997; Ling et al., 2012). Whilst it is highly likely that any introductions of this species to the Australian Antarctic and subantarctic locations will be sourced from Tasmania, ships from many other regions also visit Australia's Antarctic stations and subantarctic islands, including from Russia, China, France, Japan, New Zealand, Chile, as well as tourist ships visiting multiple ports.

There are examples of other invasive species, such as *Carcinus maenas* (European green crab) and *Rhinella marina* (cane toad), which have the ability to quickly acclimate to cold conditions well below those experienced in its native range, indicating that predictions of species distributions are likely to be underestimated in highly plastic species (Mccann et al., 2014; Tepolt and Somero, 2014) or in species with broader thermal tolerances (Janion-Scheepers et al., 2018). This means that the species we predicted as being suited to the environment only in one time period, may be able to persist into the future, despite current modelling predicting otherwise. Whilst some of our predicted species may represent error in the modelling, those species that are presently found in the higher latitudes (such as *C. japonica* and *H. musciformis*), could be better suited to the environment than we have predicted here, and some species that we predicted as not being able to survive may in fact pose a risk. We also do not understand the potential synergistic interactions that environmental variables could have that may enhance survival outside of the experimentally identified thermal tolerance limits. See Supplementary Materials (A2) for additional information on the predicted species.

Propagule pressure is also an important component of invasive species establishment (Lockwood et al., 2009). Unlike most global ports, East Antarctica and Australia's subantarctic receive relatively little shipping traffic. For example, the Australian Antarctic stations and Heard Island received less than 15 ship visits for the 2016 – 2017 season (McCarthy et al., 2019). Current shipping traffic is limited mostly to the warmer half of the year, with the shipping season running from October to March. At higher latitudes fast ice inhibits shipping traffic throughout the colder months. The passage of ships through sea ice in the warmer months acts as a natural hull cleaner, scouring fouling organisms from the hull of ships. However, the subantarctic islands are not surrounded by any sea ice, so if the first port of call after overwintering in a northern port, commonly Hobart (but not always) is a subantarctic island then there is no removal of hull fouling organisms before reaching these areas. One way to mitigate this is to schedule the first shipping voyage of the season following overwintering to include a period of traversal through sea ice before arriving at the subantarctic islands, though many vessels currently avoid sea ice where possible for safety reasons (Hughes and Ashton, 2017; Ware et al., 2014). Although traversal through sea ice has the capacity to scour the hull of fouling organisms, this does not include protected or enclosed niche areas, such as sea-chests, moon pools, wet wells, instrument cavities, and propeller shafts (Lee and Chown, 2007). Traversal through sea ice also damages antifouling that has been applied to the ship, making them more susceptible to fouling (Lee and Chown, 2009). Further research on fouling community composition in hull and protected niche areas is required to better understand the propagule load carried by research and resupply vessels (though, see Lee & Chown, 2007) as these niches may pose a substantial risk to marine invasions in the Antarctic that is currently overlooked.

One key limit of our study is that it is based on an incomplete record of global species presence. Some of the hull fouling marine invasive species identified may inhabit regions at higher latitudes, but they may simply not have been found yet, or have not been uploaded to the OBIS database. This means that even though species that we may have expected to see, such as *Mytilus galloprovincialis* (Lee and Chown, 2007), that were not predicted to survive the environmental conditions of the Australian Antarctic and subantarctic by the end of the century in our study, could actually be an example of Type II errors in the underlying dataset. The format of the dataset required for analysis, however, could be adapted to allow input of experimental data, and incorporating the results of thermal experiments and occurrence data from additional databases and primary literature sources could enhance the overall predictions.

Our study introduces a novel method of identifying and predicting when and where marine invasive species could occur using multiple environmental variables. The methods used in our study can be readily adapted to other regions of the Antarctic and subantarctic to identify species which may be environmentally suited now and in the future, particularly as new climate models become available. There is also a need to explore the potential for endemic Antarctic and subantarctic marine species to be carried by anthropogenic means to other regions of the Southern Ocean, and how species could be carried naturally throughout the region[1] (Hughes et al., 2019). Overall, the chance of an introduction occurring, now through to the end of the century, to the Australian Antarctic stations and subantarctic islands is currently deemed to be a very low risk. There is low propagule pressure, however increased interest in the Antarctic region, such as an increase in shipping activity, will increase this pressure (McCarthy et al., 2019). The highly cold-stenothermic endemic species in Antarctic and subantarctic ecosystems are under increasing stress as a result of climate change and are ill-prepared to cope with the additional stress of novel species (Ingels et al., 2012). Focus must now shift to improving quarantine procedures and investigating novel monitoring tools, such as eDNA, to prevent the introduction and establishment of MIS in the near-pristine marine ecosystems of the Antarctic and subantarctic regions.

4.8 Acknowledgements

Thanks to Dr Kate Helmstedt (QUT) for helpful advice on preparing the figures and providing feedback on the draft of this manuscript. Thanks to Dr Blake Alexander Simmons for producing the methods flow diagram.

[1] This suggestion for future work was undertaken as part of this thesis (Chapter 3).

Chapter 5: Ensemble ecosystem modelling to predict the impacts of marine invasive species to coastal Antarctic ecosystems

5.1 Preamble

In the context of this thesis as an overall risk assessment this chapter models the potential impacts that several high-risk MIS could have if they were to successfully establish in a native Antarctic coastal benthic ecosystem. The models show a range of plausible future scenarios, but do not capture the concurrent decrease in fitness that is likely to occur in many native species as a result of climate change.

5.2 Abstract

There are many examples of MIS that have caused irreparable damage in their invaded range. However, this is more the exception than the rule, with only roughly 10% of invasive species causing negative impacts to recipient ecosystems. Given the limited resources available for MIS monitoring and surveillance, we need tools that can help us identify those invasive species which could have impacts in a new environment. This is particularly true in areas that are difficult to access, such as the Southern Ocean. The Antarctic marine environment is one of the last regions on earth free of invasive species. There is a growing concern that climate change will cause conditions in the Antarctic marine environment to become suitable for non-native species establishment in the coming years. Here, I used an ensemble ecosystem model (EEM) to create a range of plausible future scenarios of changes to an Antarctic coastal marine food web following the successful establishment of a number of MIS. I found that some species groups, like macroalgae, reduce in abundance in most simulations following the successful establishment of all the invasive species. Other native species, such as predatory pycnogonids, are barely impacted by any of the invasive species. The effect on other native species is largely invasive-species dependent. In a significant portion of the simulations native and endemic species abundances reduced by more than 80% of the initial abundance, which is equivalent to becoming critically endangered. The *Mytilus sp* group struggled to successfully invade the simulated ecosystem, with only 4%

of simulations resulting in a successful invasion, indicating that this group may face a significant ecological barrier when trying to invade an Antarctic benthic ecosystem, even under ideal conditions. Other invasive species, particularly *Charybdis japonica* and *Hypnea musciformis* successfully established in more than half of the simulations and may warrant particular attention as higher risk species. This analysis helps us to better understand the potential range of impacts on native species from seven invasive species groups, so that we may better allocate resources to manage high impact invasive species.

5.3 Introduction

Detecting a new invasive species in an environment can be incredibly difficult and resource intensive due to time lag at the beginning of exponential population growth seen in species invasions (Crooks, 2005). This creates the paradox whereby the ideal time to detect an invasion is when we are least likely to know about the species. To direct the limited resources available in a meaningful way, methods to predict invasive species impacts have been developed (Branch and Nina Steffani, 2004; Paterson et al., 2015), so that we can identify those potentially devastating invasive species for surveillance and monitoring. This is particularly pertinent in regions that are exceptionally difficult to access and monitor, and where there is also a paucity of data on community composition, a lack of taxonomic identification resources and many new or undescribed species, such as the Southern Ocean.

The Antarctic marine ecosystem is one of the last pest-free regions on the planet. However, there are growing concerns that the changing climate and increased ship traffic in the region may make this region more vulnerable to MIS (Lewis et al., 2003; McCarthy et al., 2019). Whilst the addition of any invasive species into a new environment is not ideal, the potential impacts of different invasive species can range from benign to catastrophic (Dick et al., 2017). Recent work has identified several species that are a potential risk of invasion to the Antarctic region (Chapter 4; Hughes et al., 2020). These species span a range of taxa, and their potential impacts are likely to differ.

One way to make predictions of impact is to undertake a species interaction network analysis, where an invasive species is placed into a food web (Bender et al., 1984; Melbourne-Thomas et al., 2013). There are two key challenges when attempting to predict MIS impacts in the Antarctic region: a lack of information about endemic species and the strength of their interactions and a limited understanding of the potential interactions that could occur with invasive species. Therefore, methods of predicting impacts which require a quantitative understanding of ecosystem dynamics, such as functional responses (Guo et al., 2017; Paterson et al., 2015), are not currently suited to this environment.

A range of mathematical models have been developed which create quantitative models that are validated against qualitative data, whilst incorporating the uncertainty of interactions between species (Raymond et al., 2011). However, the reliability of predictions in these models is hindered by the fact that we are modelling potential networks that can move far from the initial state of equilibrium, such as with predator reintroductions and invasive species introductions. One way to overcome this problem is to use ensemble ecosystem modelling (EEM) (Baker et al., 2017). Whilst this method does not provide a single prediction of impact, it does predict a range of plausible outcomes, and gives insight into short-term network changes that may lead to undesirable long-term effects. This method is particularly suited to instances where there is limited information on the strength of species interactions, such as those in polar regions. Polar marine food webs are highly linked, with a high degree of omnivory and prey switching based on sea ice dynamics (Dayton et al., 1994; Gillies et al., 2012; Norkko et al., 2007; Rossi et al., 2019). The trophic structure of these ecosystems is representative of a continuum, rather than having discrete trophic levels (Gillies et al., 2013; Norkko et al., 2007; Rossi et al., 2019).

Here I use ensemble EEM to predict a range of plausible outcomes from the introduction of multiple 'high-risk' MIS to a known food web from Casey Station, East Antarctica (Gillies et al., 2012) with uncertain relationship strengths between many species (Baker et al., 2017). I identify the potential for endemic species to act as 'sentinel' species; those species that could indicate that a MIS has been introduced to the system and could allow for rapid management of an incursion. Further, I elucidate the fate of the invasive species.

5.4 Methods

There were four key steps to this study. First, I defined the food web models for an ice-free ecosystem in the nearshore marine environment located near Casey Station, East Antarctica. Second, potential MIS were identified and interactions with endemic species were predicted from known interactions with the MIS in their invaded ranges. Third, I created an ensemble of plausible models of species interactions using only the endemic species, and then added the MIS to those food webs to determine a range of potential impacts to endemic species. Finally, I identified which, if any, species could act as potential sentinels of the early stages of an invasion to facilitate a rapid response to MIS incursions.

5.4.1 Antarctic marine food web

I defined an Antarctic nearshore marine ice-free ecosystem based on the 53 species/species groups collected from nearshore benthic habitats near Casey Station, East Antarctica from 2006 – 2009 (Gillies et al., 2012). Species diet information was defined based on the literature, or that of similar species where information was lacking. This species list was condensed to 12 food web model ‘nodes’ (Table 5.1). Similar species were grouped, due to the quadratic-increase in node interactions that occur for each additional node in the food web and the consequential increase in computational time required to run the models (Baker et al., 2017). The interactions of the native species are described below (Figure 5.1). Ideally, all species would appear as a separate node in the food web; however it becomes computationally prohibitive beyond 15 nodes. The final node represents the invasive species (Table 5.2).

5.4.2 Marine Invasive Species

The inhospitable conditions of the Southern Ocean make it uninhabitable for most species. However, recent expert groups and species modelling have identified a small suite of potential ‘high-risk’ MIS that could survive in the cold Southern Ocean waters, although they likely lack the ability to successfully establish in these environments currently (Chapter 4) (Hughes et al., 2020). Regardless, these species have shown considerable tolerance and adaptation to novel environments and warrant caution.

Twelve species were considered for my study, which were collated into 8 groups (Table 5.2). Two species (*Undaria pinnatifida* and *Ciona intestinalis*) were later excluded from the study as current research indicates that they can only establish in disturbed environments, which are outside the scope of this study, i.e., this method presumes a stable ecosystem prior to species introduction. Further, the tube polychaete, *Chaetopterus variopedatus*, was excluded from analysis as there are currently several records of this species, or its synonymised species, *Chaetopterus antarcticus*, in the Antarctic and subantarctic region, hence its status as non-native to the region is unclear, leaving seven groups for analysis. As the interactions between endemic Antarctic species and the invasive species are completely unknown, I predicted potential interactions based on each species' current interactions in their native and invaded ranges as reported in the literature (See Appendix B).

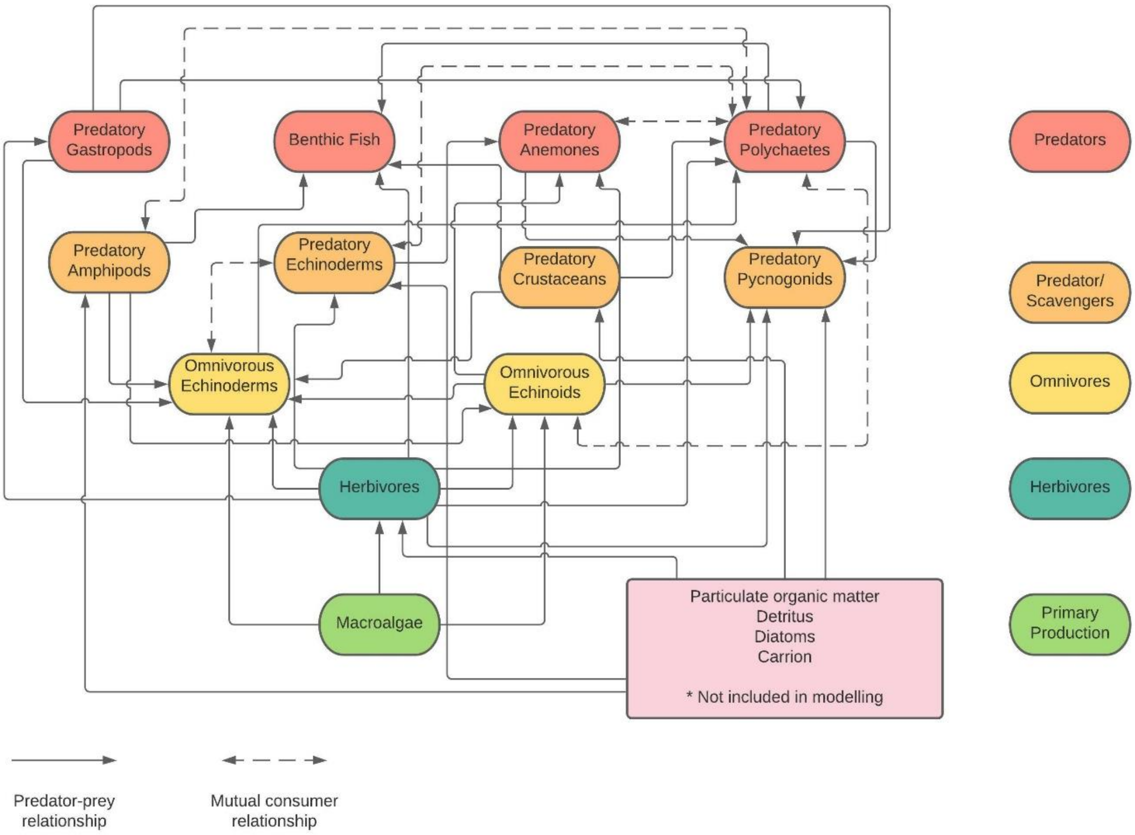


Figure 5.1 Condensed native food web model of nearshore benthic species near Casey Station.

Table 5.1 Node groupings for the native Antarctic species identified by Gillies et al. 2012.

Node	**Species**	**Reference/s**
Macroalgae	*Chaetomorpha sp.*	
	Desmarestia sp.	
	Himantothallus grandfolius	
	Iridaea cordata	
	Monostroma sp.	
	Palmaria decipiens	
	Phyllophora antarctica	
Herbivores*	*Abatus sp.*	(McClintock, 1994)
	Eudorella cf. splendida	(Blazewicz-Paszkowycz and Ligowski, 2002)
	Doloria sp.	(Chavtur et al., 2012; Chavtur and Keyser, 2016; Jumars et al., 2015)
	Ophiura crassa	(Clark and Souster, 2012; Lane and Riddle, 2004)
	Orbiniidae	(Jumars et al., 2015)
	Orchomenella franklini	(Gillies et al., 2012; Nyssen et al., 2002)
	Orchomenella pinguides	(Gillies et al., 2012; Nyssen et al., 2002)
	Scleroconcha sp.	(Chavtur and Keyser, 2016; Jumars et al., 2015)
	Cymodocella tubicauda	(Zemko et al., 2015)
	Paramoera walkeri	(Brown et al., 2015)
	Shennelle paludinoides	(Takeuchi and Watanabe, 2002)
	Adamussium colbecki	(Gillies et al., 2013; Norkko et al., 2007)
	Cucumaria sp.	(Gillies et al., 2013, 2012; McClintock, 1994)
	Homaxinella sp.	
	Isodictya sp.	
	Laternula elliptica	(Agüera et al., 2017; Ahn, 1994; Gillies et al., 2013; Norkko et al., 2007)
	Perkinsiana cf. antarctica	(Jumars et al., 2015)
	Staurocucumis sp.	(Gillies et al., 2013, 2012; McClintock, 1994)
Benthic fish	*Notothenia coriiceps*	(Casaux et al., 2003; Gillies et al., 2012; Iken et al., 1997; Moreira et al., 2020; Zamzow et al., 2011)
	Trematomus bernacchii	(Brown et al., 2015; Carlig et al., 2018; Kiest, 1993; La Mesa et al., 2004)

Omnivorous Echinoderms	*Odontaster Validus*	(McClintock, 1994; Norkko et al., 2007)
Omnivorous Echinoids	*Sterechinus neumayeri*	(McClintock, 1994; Norkko et al., 2007)
Predatory Crustaceans	*Nototanais antarcticus*	(Blazewicz-Paszkowycz and Ligowski, 2002)
	Nototanais dimorphus	(Blazewicz-Paszkowycz and Ligowski, 2002)
Predatory Amphipods	*Heterophoxus videns*	(Dauby et al., 2001; Oliver et al., 1982)
	Methalimedon nordenjoelski	(Dauby et al., 2001; Oliver et al., 1982)
Predatory Echinoderms	*Diplasterias brucei*	(Dayton et al., 2019; Llano, 1977; McClintock, 1994)
	Psilaster charcoti	(McClintock, 1994)
Predatory Gastropods	*Neobuccinium eatoni*	(Norkko et al., 2007)
Predatory Anemones	*Urticinopsis antarctica*	(Ivanova and Grebelnyi, 2017)
Predatory Pycnogonids	*Nymphon australe*	(Dietz et al., 2018; Soler-Membrives et al., 2011)
Predatory Polychaetes	*Nephtyidae sp.*	(Fauchald and Jumars, 1979; Jumars et al., 2015)
	Polynoidae sp.	(Fauchald and Jumars, 1979; Jumars et al., 2015; Mettam, 1980)
	Priapulida sp.	(Aarnio et al., 1998; Trott, 1998)

* Herbivore group includes all other first order consumers (e.g., suspension feeders, deposit feeders, etc)

5.4.3 Ensemble Ecosystem Modelling

My method borrows from the EEM method developed by (Baker et al., 2017) to assess potential impacts of predator reintroductions, which includes a detailed description of the mathematical principles underlying it. Herein, I will briefly describe those methods and provide details where alterations to the method have been made. As with the paper by (Baker et al., 2017), I refer to each grouping as ‘node’ as not all groupings need be species.

The EEM method involves creating thousands of plausible food web model simulations, where all nodes in the current ecosystem are able to persist through time. One limitation to this method, therefore, is that the EEM method cannot be used in its current form to model ecosystems which are either a) disturbed ecosystems (e.g., an ecosystem recovering after ice-berg scour), or b) transitional in nature (e.g. where significant ecosystem changes occur due to sea-ice breakout and subsequent refreezing). As such, I have limited the simulations to include only ice-free ecosystems.

I created a matrix of species interactions, noting only the direction of the interaction (e.g., herbivores eating macroalgae), where: '1' meant species X was eating species Y; '0' meant there was no interaction between species X and species Y; and '-1' meant that species X was being eaten by species Y. A separate matrix was created for species that eat each other, meaning both species in the relationship had both a positive and negative interaction. The strength of the EEM method is that it captures uncertainty within the model by randomising the strength of the interaction in each simulation. For those species which eat each other, two random numbers between (0,1) and (-1,0) were drawn to simulate the positive and negative aspects of the relationship. These two numbers were added together to represent a net interaction effect. It is possible to create any number of matrices to capture different strengths of interactions, for example, creating a matrix that includes incidentally eaten species and assigning a random strength that has a smaller range and/or value than the original matrices; however this was not done in my study.

In aggregate, the ensemble of these simulations provides a suite of possible impacts of the introduction of a species. Species abundances were modelled through generalised Lotka-Volterra equations. Initial abundances were selected at random and tested for equilibrium and stability. Simulations which did not pass the plausibility constraint were automatically discarded. The number of discarded unviable simulations was in the order of tens of billions in this study, similar to other studies using a similar methodology (Peterson and Bode, 2021). The number of plausible models to be retained was set at 10,000; however this number can be altered within the code to suit the users' requirements and computational limitations. To provide realistic time periods of changes to species' abundances, population growth rates should be supplied for at least one of the species in the network. Where species were grouped the growth rate was the average of the known growth rates for the species within the group. Where population growth rates were not known, the model randomly drew a population growth rate from a uniform distribution based around a specified mean. In this study I used a lognormal mean taken from the mean of known species population growth rates (Table 5.3). The initial abundance of the invasive species was set to be a fraction, randomly drawn from between 10% and 50% of the smallest abundance of the native species, to simulate the low starting populations of new species in an area.

Table 5.2 Groupings of the ‘high-risk’ marine invasive species explored in this study. Each group represents the invasive species node in the food web. Where there is only one representative species for a group, then the species name is used herein for simplicity.

Group	Species	Reference/s
Algae	*Hypnea musciformis*	(Alidoost Salimi et al., 2021)
	Undaria pinnatifida[a]	(Valentine and Johnson, 2003)
Colonial Ascidians	*Botryllus schlosseri*	(Cima et al., 2015; Giachetti et al., 2020)
	Ciona intestinalis[a]	(Dumont et al., 2011)
Tube Polychaetes	*Chaetopterus variopedatus*[b]	(Pabis and Sicinski, 2010)
Mytilid Bivalves	*Mytilus chilensis*	(Branch and Nina Steffani, 2004)
	Mytilus edulis	(Branch and Nina Steffani, 2004)
	Mytilus galloprovincialis	(Branch and Nina Steffani, 2004)
Omnivorous/Detritivore Crabs	*Halicarcinus planatus*	(Diez and Lovrich, 2010; López-Farrán et al., 2021)
Omnivorous Crabs	*Carcinus maenas*	(Garside et al., 2015; Grosholz and Ruiz, 1996)
Predatory Crabs	*Charybdis japonica*[c]	(Garside et al., 2015; Townsend et al., 2015)
Predatory Asteroids	*Asterias amurensis*	(Aquenal Pty Ltd, 2008a; Byrne et al., 2013; Parry, 2017; Ross et al., 2003a)

[a] These species are identified as ‘high-risk’; however, they are only known to establish in disturbed environments, which it outside the scope of this study.

[b] This species is identified as ‘high-risk’ however its status as invasive in the Antarctic region is unclear, given its presence in the Southern Ocean, and the presence of its synonymised species, *Chaetopterus antarcticus*, in the Antarctic region.

[c] There is a lack of information on the predators of *Charybdis japonica*. so I have used confamilial predators (*Carcinus maenas* and *Ovalipes catharus*)

Table 5.3 Growth rates for each group based on the listed species.

Group	Species	Growth rate ($year^{-1}$)	Reference
Macroalgae	*Many*[a]	0.0295	(Commonwealth Science and Industrial Research Organisation, n.d.; Wiencke and tom Dieck, 1990)
Herbivores	*Adamussium colbecki*	0.1200	(Brey and Clarke, 1993)
		0.0900	
		0.2520	
	Psammechinus miliaris[b]	0.2800	(Ebert, 1975)
		0.1300	
	Adamussium colbecki	0.1140	(Heilmayer et al., 2003)
Omnivorous Echinoids	*Sterechinus antarcticus*[c]	0.0170	(Brey and Clarke, 1993)
Benthic Fish	*Notothenia coriiceps*	0.0980	(Coggan, 1997)
	Trematomus bernacchii	0.0880	(La Mesa et al., 1996)
		0.1780	
		0.1270	
		0.3600	
		0.2200	
		0.1070	
		0.0550	
Invasive	*Asterias amurensis*	0.2910	(Dunstan and Bax, 2007)
	Botryllus schlosseri	1.7000	(Cockrell and Sorte, 2013)
	Charybdis japonica[d]	1.1200	(Tweedley et al., 2017)
	Mytilus galloprovincialis[e]	0.1000	(Branch and Nina Steffani, 2004)
		0.2000	
	Hypnea musciformis	0.0578	(de Faveri et al., 2015)
	Halicarcinus cookii[f]	0.8495	(McLay and Van Den Brink, 2009)
		1.4755	
		0.7993	
		0.8483	
		1.4518	
		0.6784	
		1.2113	
		1.0747	
		0.0892	
	Carcinus aestuarii[g]	0.7950	(Glamuzina et al., 2017)
		0.6780	

[a] Approximated from Wiencke & Dieck (1990). Averaged over Antarctic macroalgae species and determined population doubling time, ~23.5 days. Growth constant was solved for with the doubling time equation DT = 1/(k/ln(2)), where k is the growth constant and DT is doubling time (Bartlett 1993).
[b] Surrogate species for herbivorous echinoids from the Antarctic region.
[c] Surrogate species for omnivorous echinoids from the Antarctic region.
[d] Average of other portunid crab species for *Charybdis japonica*.
[e] Surrogate for the *Mytilus sp.* group.
[f] Surrogate for *Halicarcinus planatus*. Averaged growth rates from different sexes and life stages.
[g] Surrogate for *Carcinus maenas*. Averaged growth rates from different sexes.

With these plausible food web models, I introduced an invasive species/group to the food web and simulated changes to that food web model over a period of 100 years. This time frame is longer than the original reintroduction time frame (25 years) from (Baker et al., 2017) due to the slower growth rates and longer generation times experienced by Antarctic species because of the perpetually cold environment (Peck, 2005).

5.5 Results

The effect of invasive species on the native species ranged from benign, as seen in the predatory pycnogonids, to considerable, as seen with the herbivore group (Figure 5.3). Overall, the herbivore group fared worst against all the invasive species groups, with all invasive species leading to declines in herbivore abundance 100 years post introduction. At the other end of the scale, benthic fish fared best in the simulated invasions, with most simulations showing an increase in abundance, though not to the same magnitude as the herbivore declines. Macroalgae, on the other hand, had an invasive species-specific response; increasing in abundance when invaded by the portunid crab, *Charybdis japonica*, and declining in abundance when invaded by the competitively superior red algae, *Hypnea musciformis*. The effect of each invasive species is explored in further detail below.

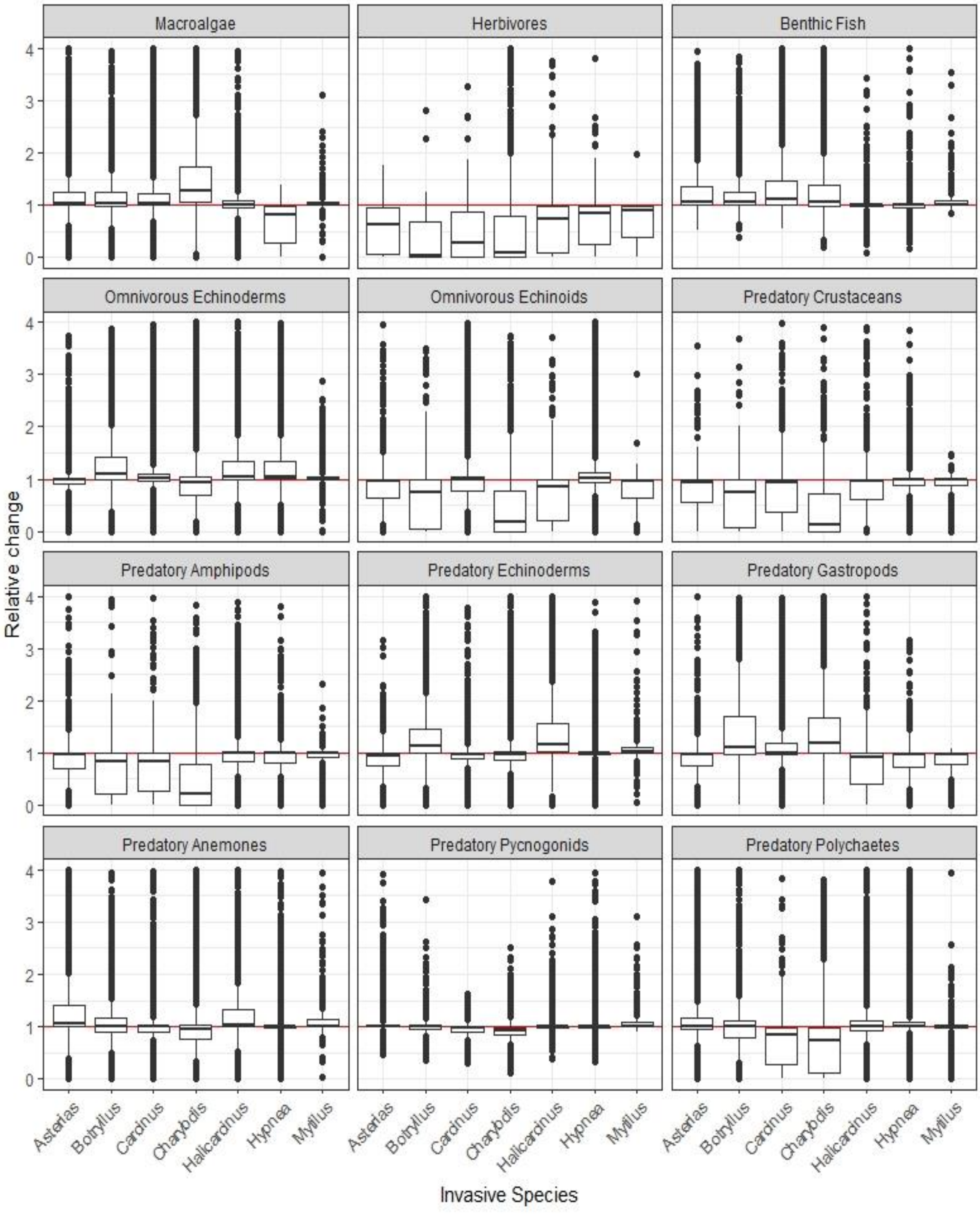


Figure 5.2 The relative change in abundance of native Antarctic species groups after 100 years of a simulated invasion for each of the seven invasive species groups explored in this study. Line at '1' represents the initial abundance of each species group. The number of simulations for each species group is species group-specific and represents the outcomes where the invasive species increased in abundance following introduction, i.e., outcomes were excluded where the invasive species declined following introduction.

5.5.1 Fate of Invasive Species

Not all simulations (n = 10,000) resulted in successful establishment of the invasive species (Figure 5.3). In particular, the *Mytilus* bivalve group only successfully established (where the invasive species increased in abundance following introduction) in less than 4% of simulations (n = 318), indicating that this species group may struggle to gain a foothold in an Antarctic benthic ecosystem with its known growth rate. Another three invasive species only successfully established in less than half of the simulations: the asteroid *Asterias amurensis* (n = 3,299), the ascidian *Botryllus schlosseri* (n = 1,401), and the omnivorous/detritivore crab *Halicarcinus planatus* (n = 1,764). No invasive species was overwhelmingly successful in establishing in the Antarctic ecosystem, though the omnivorous crab *Charybdis japonica,* the red algae *Hypnea musciformis,* and the portunid crab *Carcinus maenas*, successfully established in more than half of all simulations (n = 7,638 / 6,908 / 5,223, respectively). However, within the subset of simulations where the invasive species increased in abundance, the majority resulted in the invasive species abundance increasing significantly (Figure 5.4).

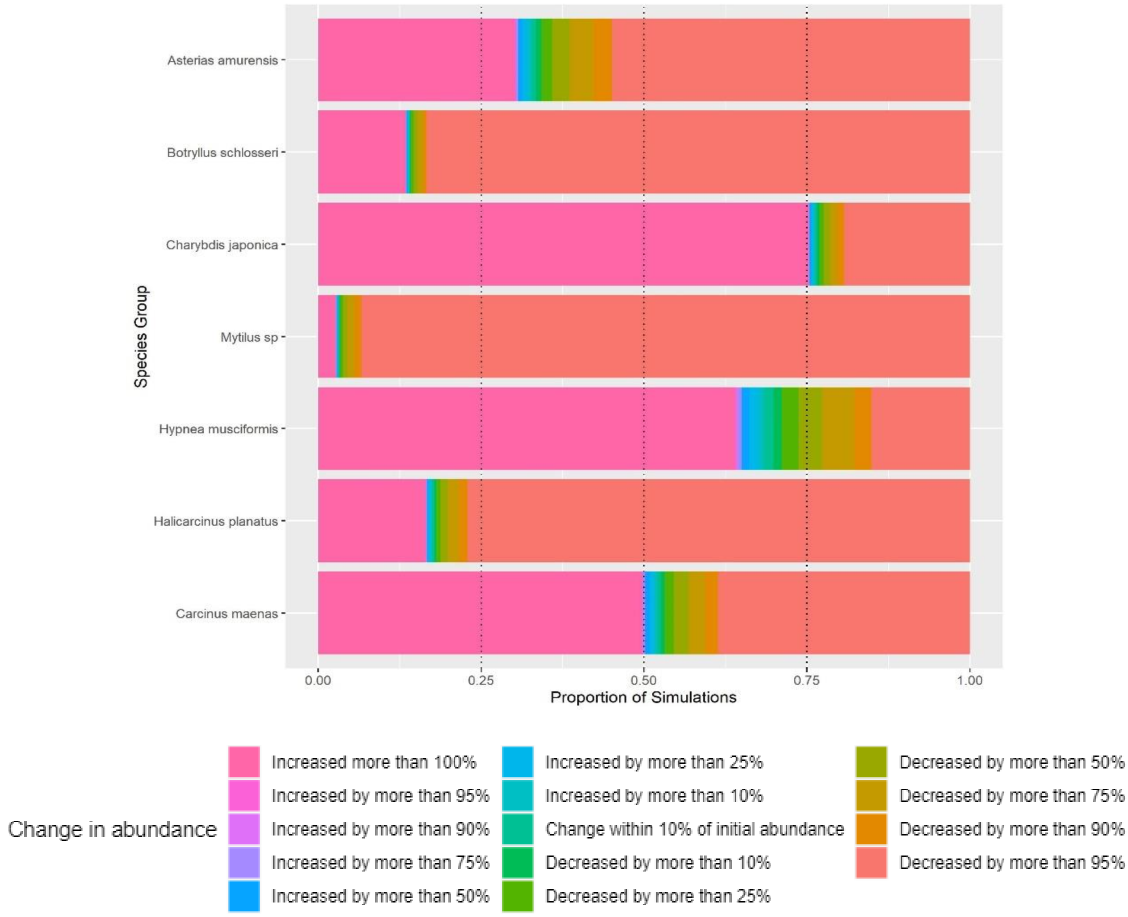


Figure 5.3 Overall change in abundance for the invasive species in all simulations (n = 10,000).

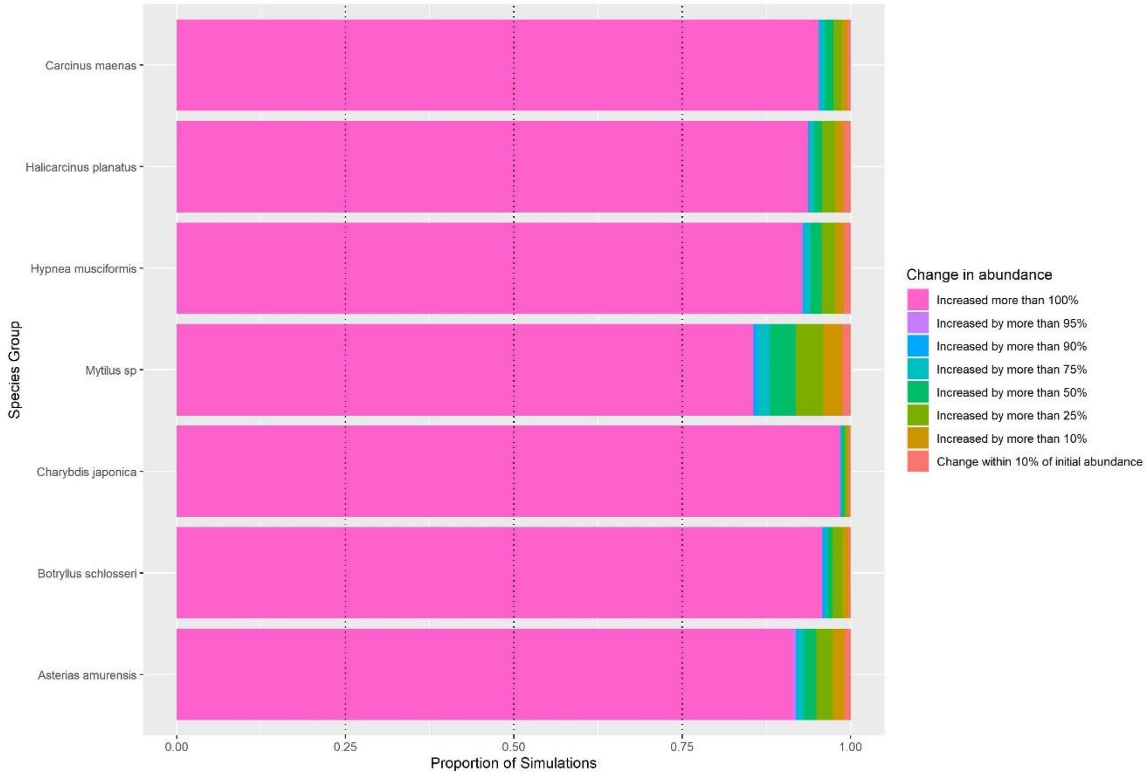


Figure 5.4 The overall change in abundance from the introduction abundance for each invasive species, where the invasive species increased in abundance after introduction.

5.5.2 Invasive Species: Asterias amurensis

The invasive asteroid, *A. amurensis*, declined by more than 95% of its initial abundance in more than half of the simulations (Figure 5.3). In the subset of simulations where the abundance of *A. amurensis* remained stable or increased (n = 3,299) (Figure 5.4), most simulations showed an at least doubling of the initial abundance in this species (Figure 5.5). In most simulations where successful establishment has occurred, the change in abundance of the endemic species at the end of 100 years was within 10% of the initial abundance. This subset also showed that herbivores were most at risk of substantial declines if *A. amurensis* successfully establishes, with more than half the simulations resulting in a decline in abundance of herbivore species and the remaining simulations showed little movement from the initial abundance. Similarly, nearly half the simulations showed a decrease in predatory crustaceans, with more than a quarter of simulations resulting in declines in abundance of more than 50%. Conversely, the successful establishment of *A. amurensis* led to an increased abundance for predatory anemones in half of the simulations.

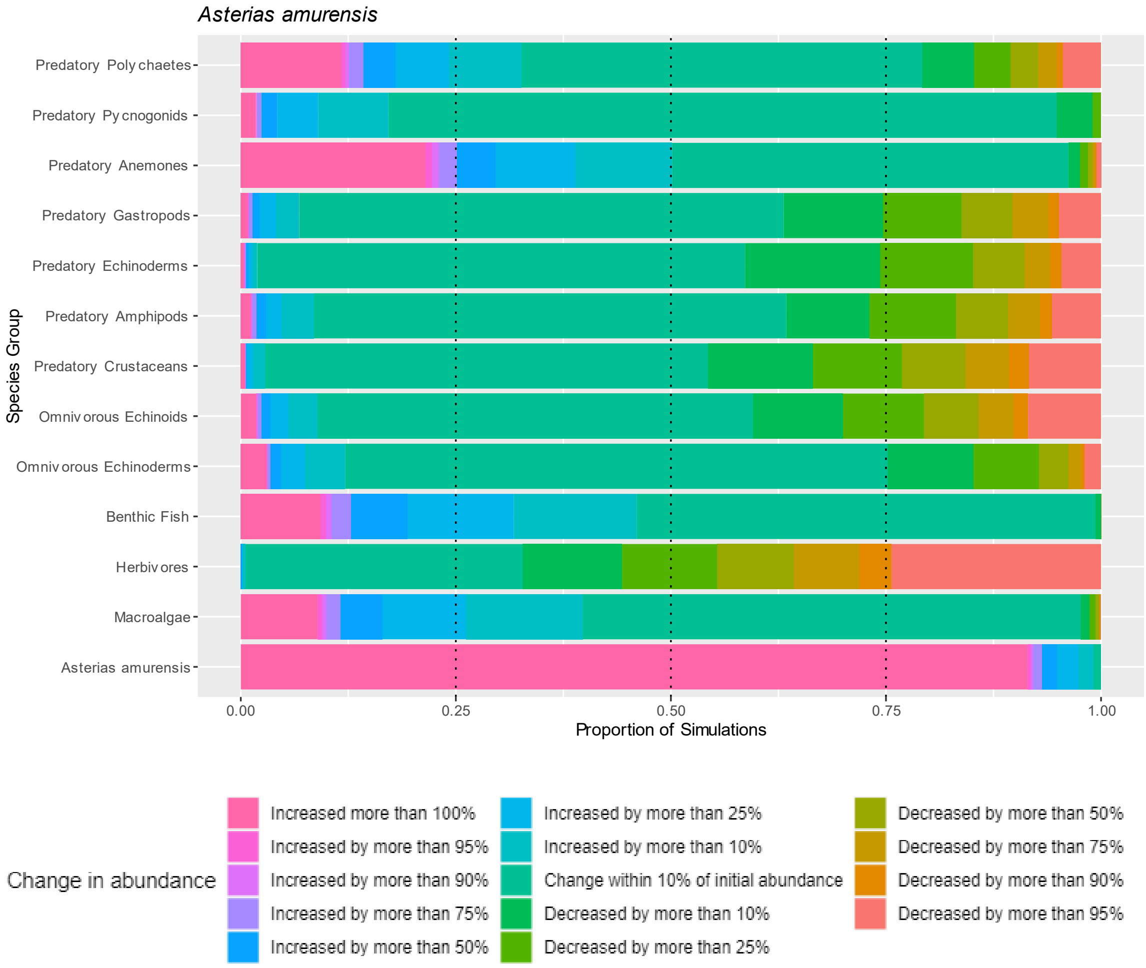


Figure 5.5 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Asterias amurensis*, in simulations where *A. amurensis* successfully established in the simulated invasion of a stable food web (n = 3,299).

On average, over the 100-year time period, the food web does not return to a state of equilibrium in simulations where the invasive species successfully establishes (Figure 5.6). The modelling shows that we might expect to see increases in benthic fish species, predatory anemones, predatory pycnogonids, and macroalgae. All other species groups decline over this 100-year period, with predatory echinoderms experiencing the largest declines in abundance. The effects of *A. amurensis* on the endemic food web generally do not present until the 10-year mark, which coincides with the end of the lag period following the introduction of the invasive species.

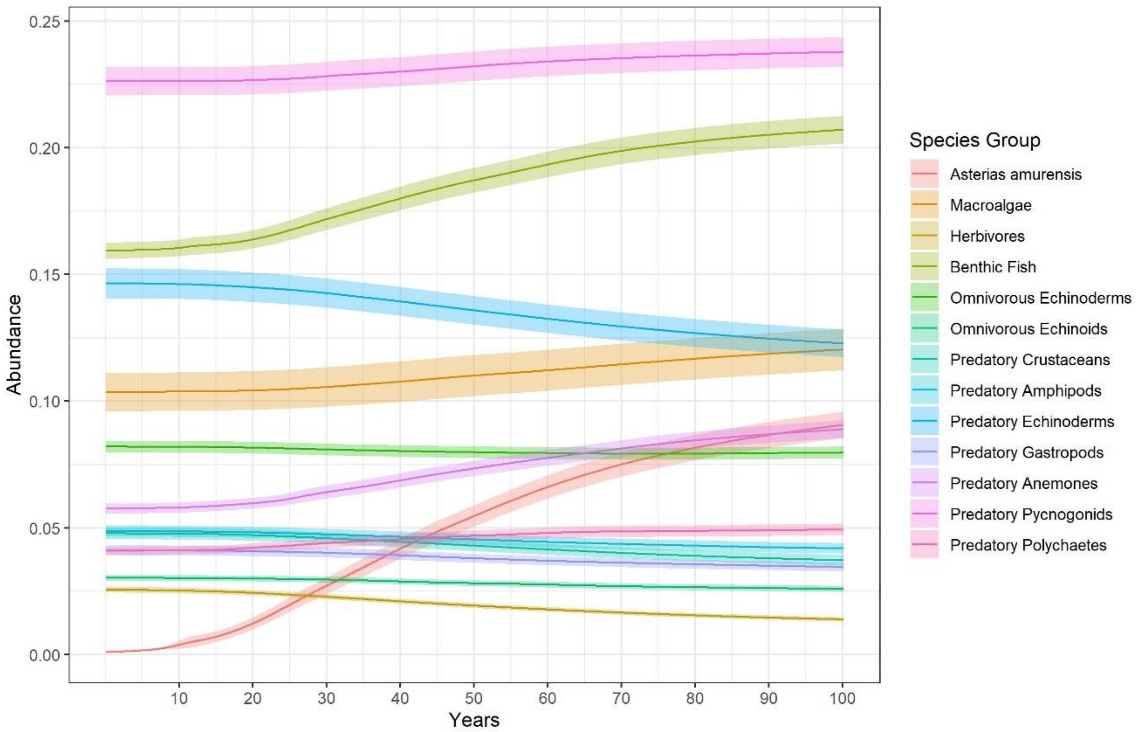


Figure 5.6 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Asterias amurensis* into an Antarctic endemic benthic ecosystem over a 100-year period, in simulations where *A. amurensis* successfully established in the simulated invasion of a stable food web (n = 3,299).

5.5.3 *Invasive Species: Botryllus schlosseri*

The colonial ascidian, *B. schlosseri*, decreased in abundance in more than 75% of simulations (Figure 5.3) However of those simulations which showed an increasing abundance (n = 1,401), most simulations led to a more than doubling of the species (Figure 5.4). In the subset of simulations where the invasive species increased, herbivores declined in more than 75% of simulations, and more than half of those were a decline of 95% or more (Figure 5.7). Many species experienced declines in their abundance in more than half of the simulations, such as, omnivorous echinoids, predatory crustaceans, and predatory amphipods. In contrast there were species which increased in abundance in more than half of the simulations; for example, predatory gastropods, predatory echinoderms, omnivorous echinoderms, and macroalgae.

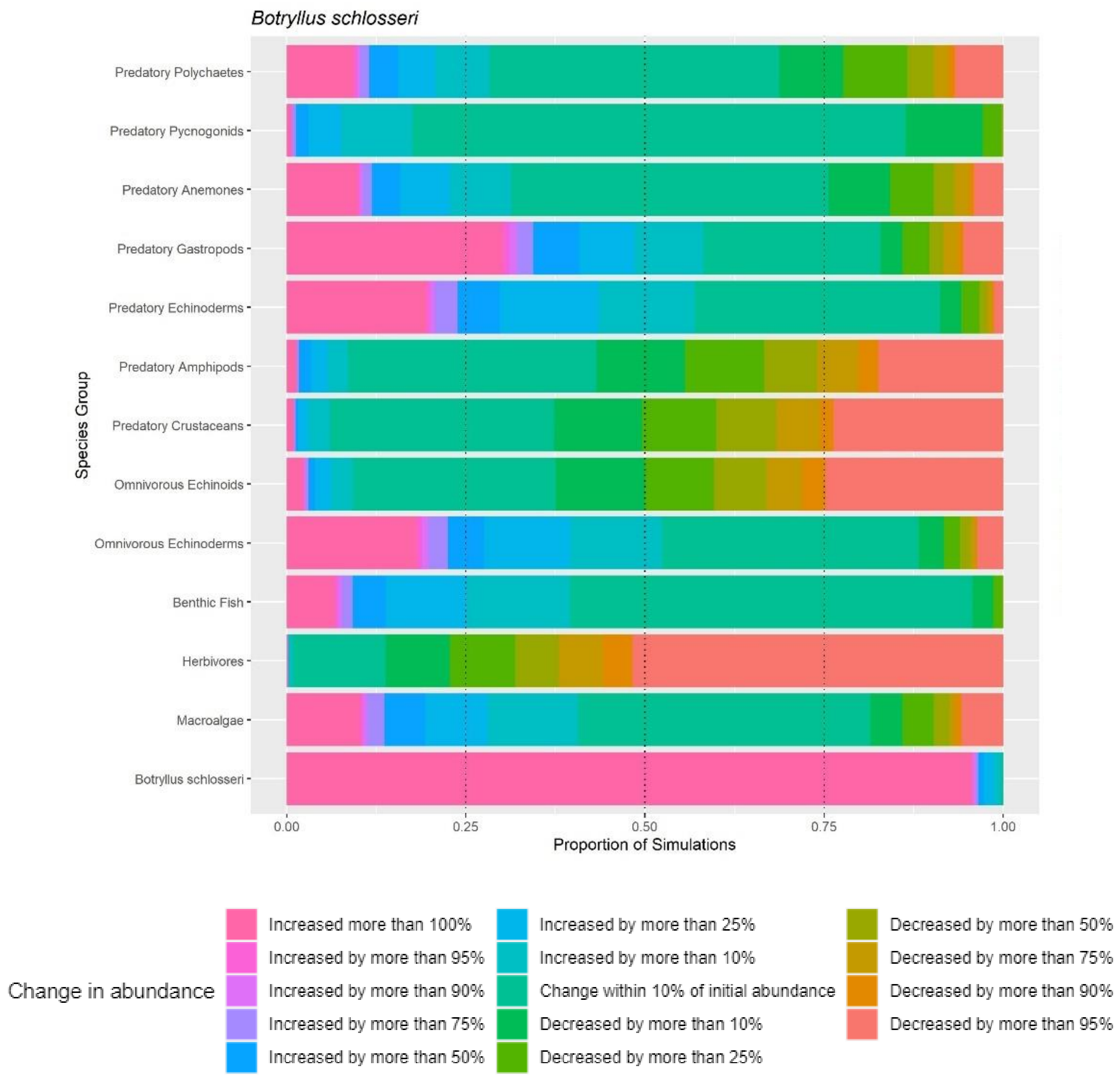


Figure 5.7 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Botryllus schlosseri*, in simulations where *B. schlosseri* successfully established in the simulated invasion of a stable food web (n = 1,401).

The increase in abundance of *B. schlosseri* occurs quite soon after establishment and continues to climb until reaching a plateau at around 50-years post establishment, after which only minor increases in abundance occur (Figure 5.8). The other species in the food web also reach general plateaus in their abundance around this time, with only minor increases or decreases observed in the latter half of the simulation period.

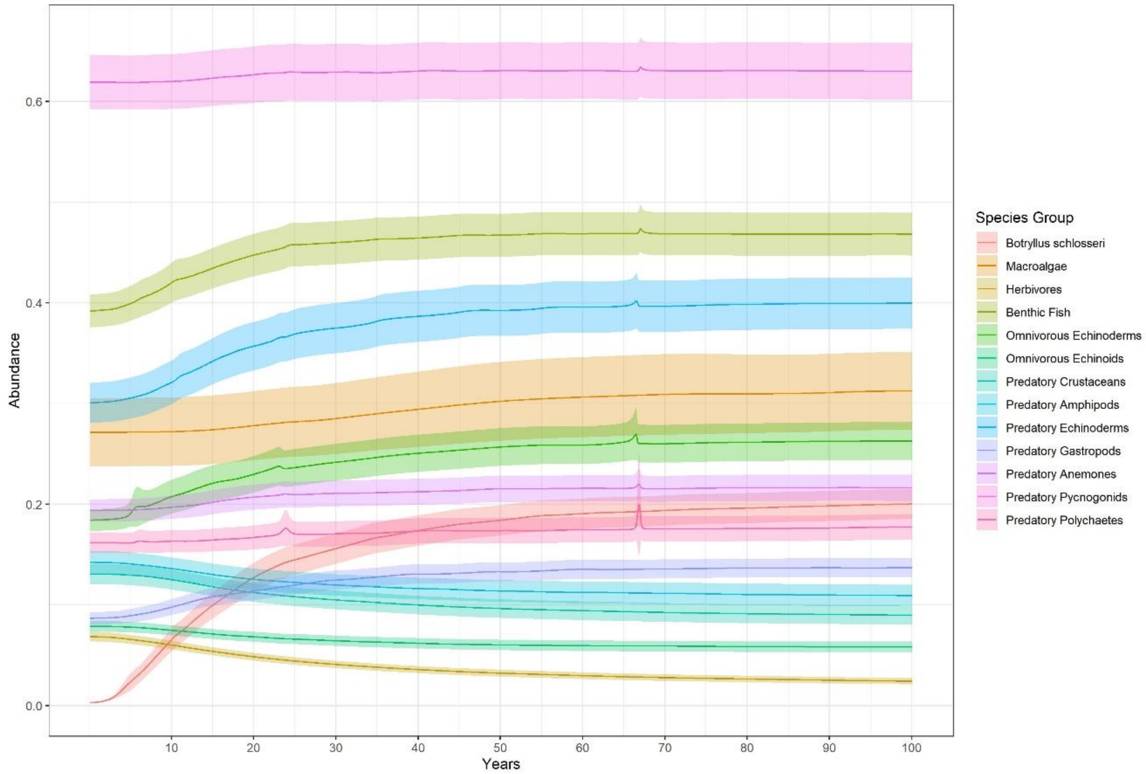


Figure 5.8 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Botryllus schlosseri* into an Antarctic endemic benthic ecosystem over a 100-year period, in simulations where *B. schlosseri* successfully established in the simulated invasion of a stable food web (n = 1,401).

5.5.4 Invasive Species: Charybdis japonica

The predatory crab, *C. japonica*, increased in abundance in more than 75% of simulations, and of those simulations (n = 7,638), nearly all resulted in an increase in abundance of more than 100% (Figures 5.3 and 5.4). In the subset, several species experienced declines in abundance in more than half of the simulations, such as predatory polychaetes, predatory amphipods, predatory crustaceans, and herbivores (Figure 5.9). In all those species, except predatory polychaetes, declines in abundance of more than 95% were seen in nearly half of all simulations where the invasive species increased in abundance. There was one group that declined to extinction, the herbivores; however this occurred in only one simulation.

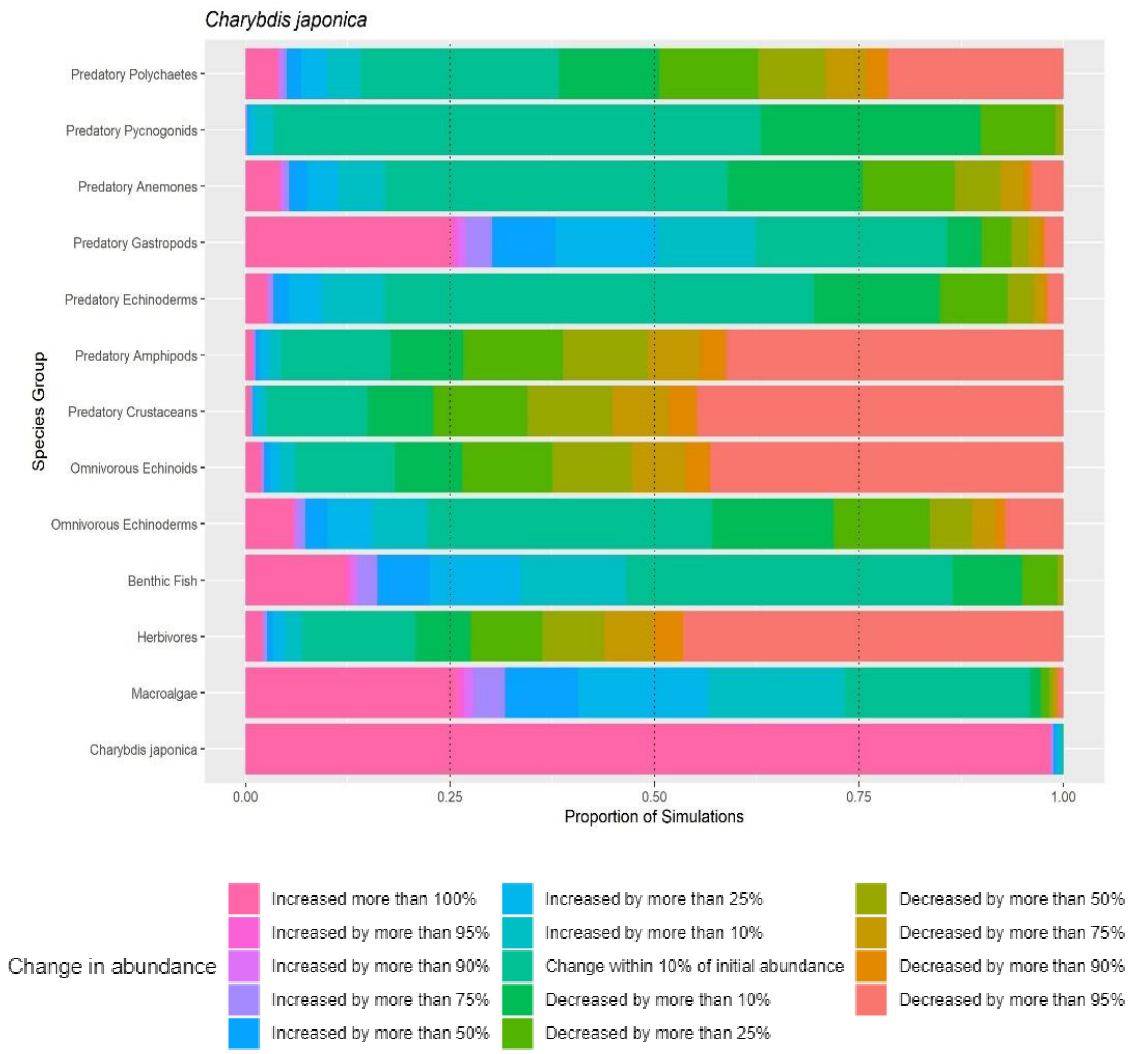


Figure 5.9 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Charybdis japonica*, in simulations where *C. japonica* successfully established in the simulated invasion of a stable food web (n = 7,638).

In simulations where *C. japonica* increased in abundance, on average it increased in abundance quickly before plateauing around 60 years post-introduction (Figure 5.10). The native species generally plateaued around the same time, with only small changes in abundance over the latter half of the simulation period.

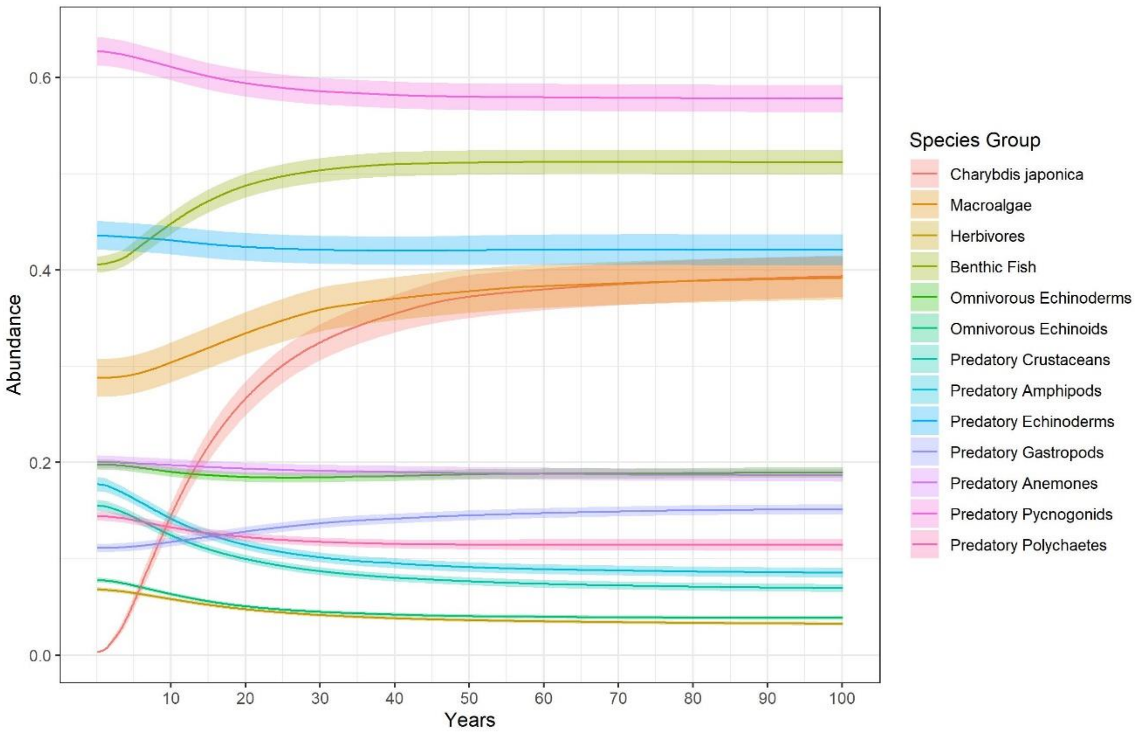


Figure 5.10 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Charybdis japonica* into an Antarctic endemic benthic ecosystem over a 100-year period from a subset of simulations where *C. japonica* increased in abundance following establishment (n = 7,638).

5.5.5 Invasive Species: Mytilus sp.

The bivalve genus, *Mytilus*, declined in abundance in the majority of simlations, with most of those being declines of 95% or more (Figures 5.3 and 5.4). In the subset of simulations where *Mytilus* successfully established (n = 318), native species largely remained within 10% of their initial abundance, whilst the invasive species increased by over 100% in most of the simulations (Figure 5.11). The exception to this was the herbivore group, which declined in abundance in just over half of the simulations.

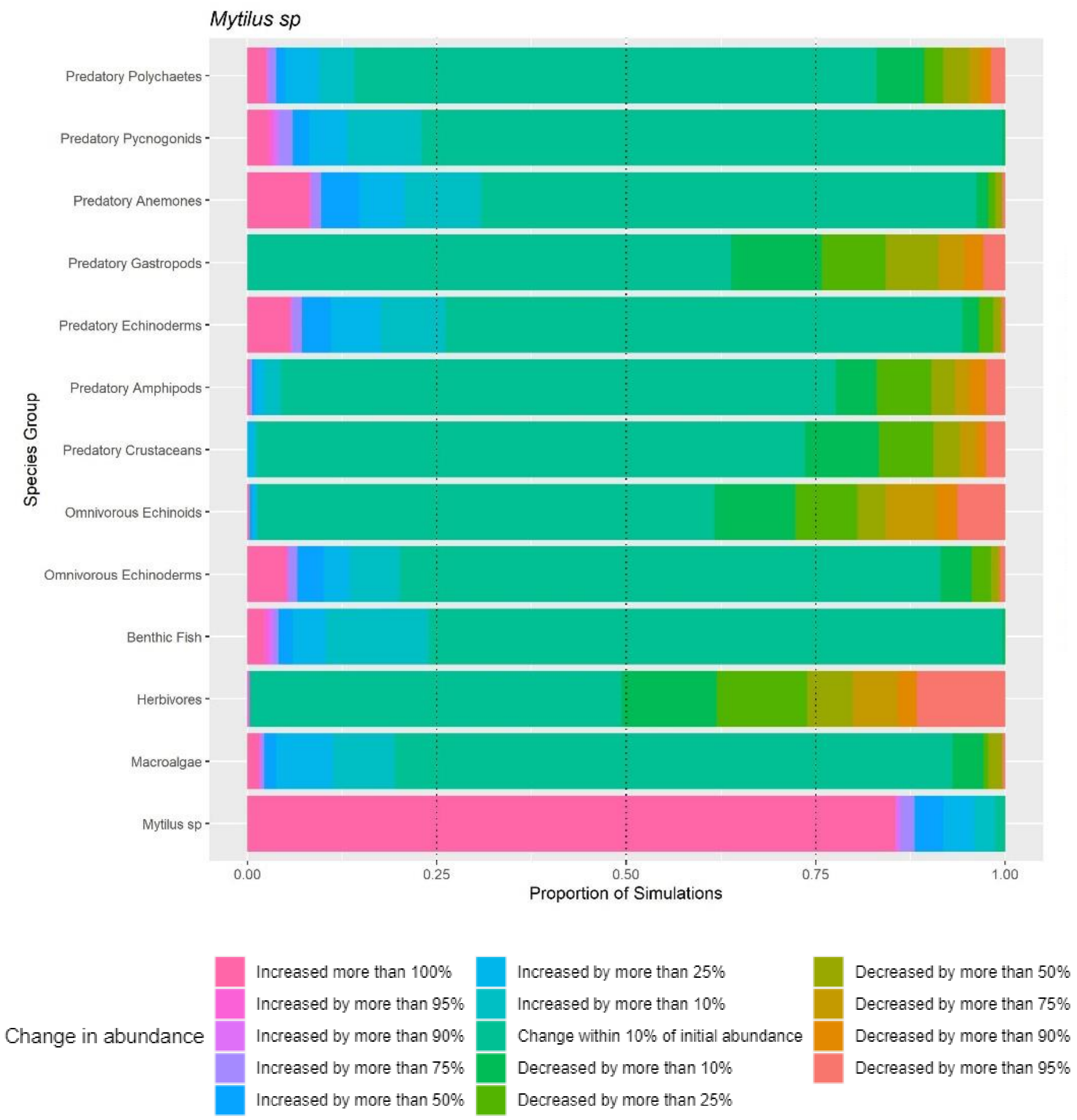


Figure 5.11 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Mytilus sp*, in simulations where *Mytilus* successfully established in the simulated invasion of a stable food web (n = 318).

Compared to the other invasive species, the *Mytilus* group took longer to initially increase in abundance (10 to 20 years) and had a generally shallower slope of abundance growth (Figure 5.12). The declines observed in the native species groups do not appear until at least 40 years after introduction of the invasive species, and there are no abrupt changes in the abudance of any native species.

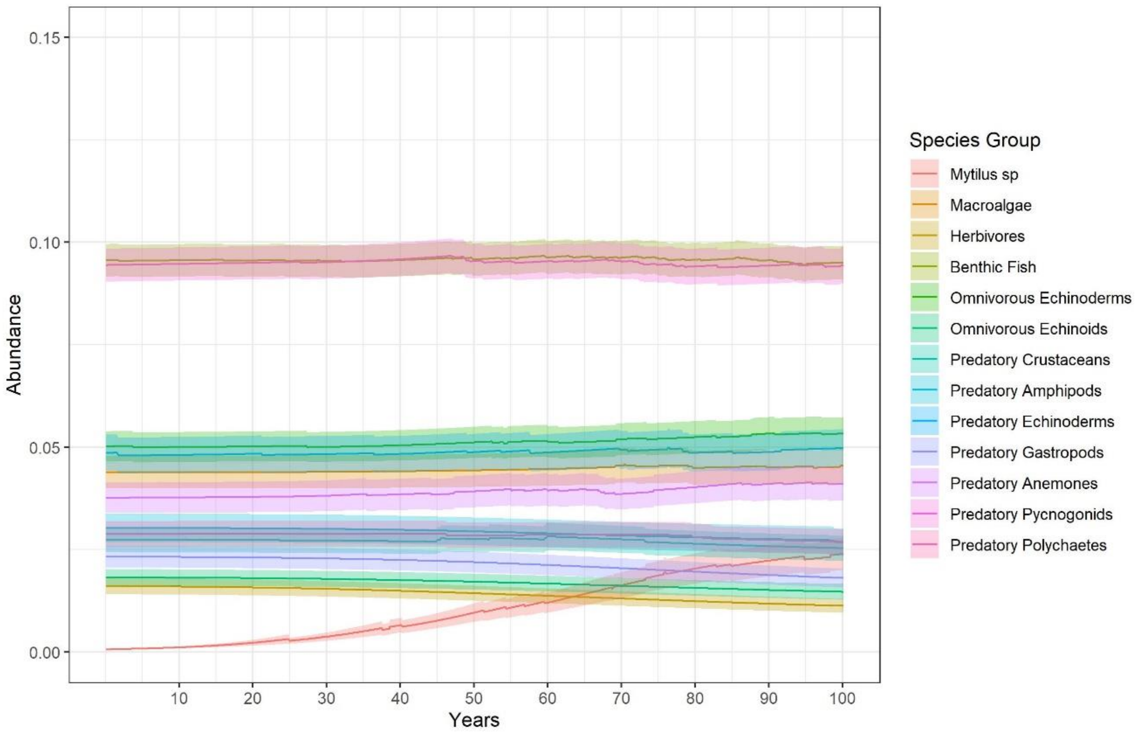


Figure 5.12 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Mytilus sp.* into an Antarctic endemic benthic ecosystem over a 100-year period from a subset of simulations where *Mytilus sp.* increased in abundance (n = 318).

5.5.6 Invasive Species: Hypnea musciformis

The red algae, *H. musciformis*, increased in abundance by at least 100% in more than half of the simulations (n = 6,908) (Figure 5.3). In the subset of simulations where this species increased in abundance, the majority of simulations showed an increase of more than 100% (Figure 5.4). For most endemic species, their final abundance remained within 10% of the initial abundance (Figure 5.13). The exceptions to these are the macroalgae and herbivore groups which declined in more than half of the simulations where the invasive species increased. In contrast, omnivorous echinoderms increased in abundance in nearly half of the simulations.

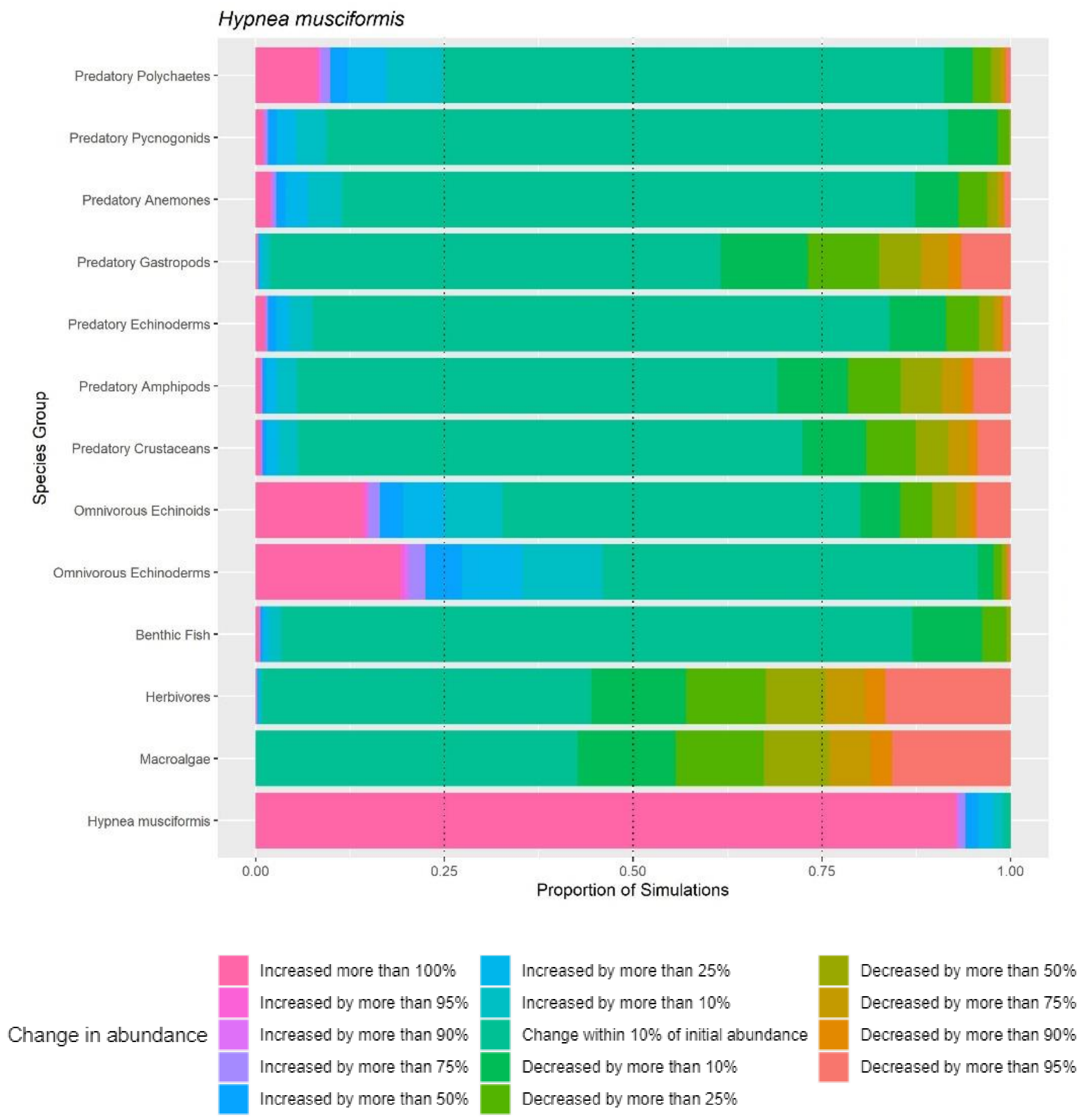


Figure 5.13 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Hypnea musciformis*, in simulations where Mytilus successfully established in the simulated invasion of a stable food web (n = 6,908).

There were still significant changes occurring in abundances at the end of the simulation period, most notably with the increase in abundance for the invasive algae, *H. musciformis*, and the decrease in abundance for the native macroalgae group (Figure 5.14). Omnivorous echinoderms also continued to rise in abundance at the end of the simulation period. A longer simulation period would be needed to elucidate the final community structure following the successful invasion of *H. musciformis*.

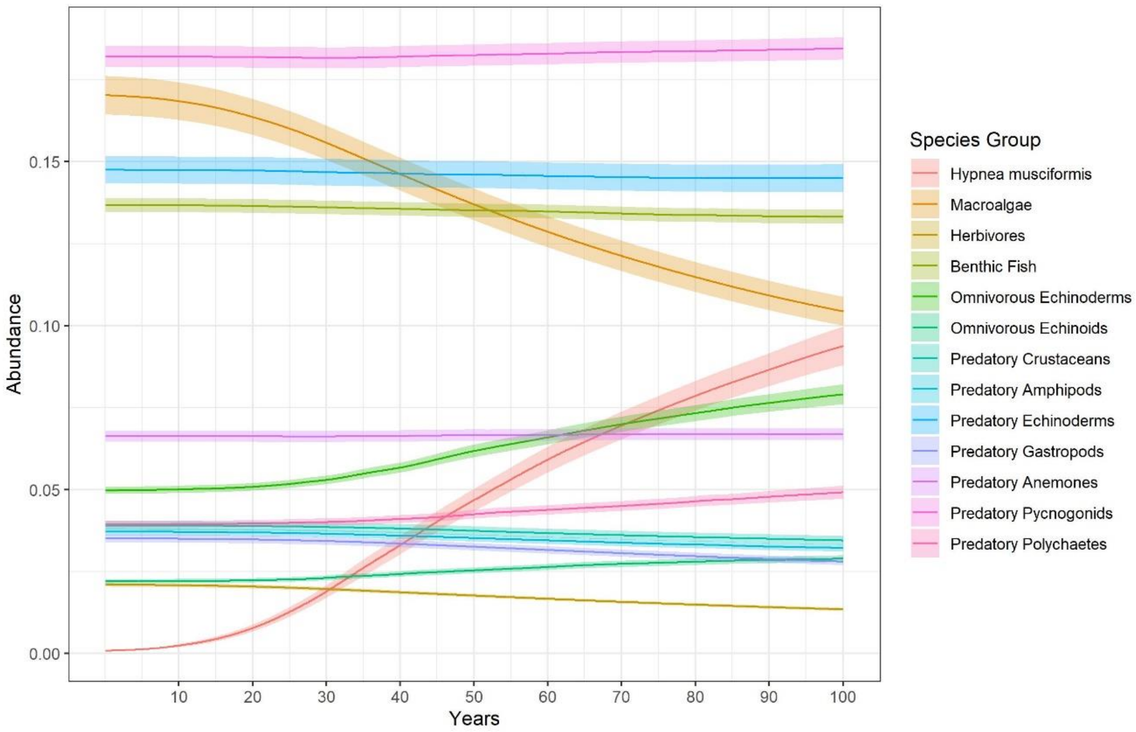


Figure 5.14 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Hypnea musciformis* into an Antarctic endemic benthic ecosystem over a 100-year period from a subset of simulations where Mytilus sp. increased in abundance (n = 6,908).

5.5.7 *Invasive Species: Halicarcinus planatus*

The omnivorous/detritivore crab *H. planatus* increased in abundance in ~17% of simulations (Figure 5.3) and within that subset most simulations showed an increase in abundance of more than 100% (Figure 5.4). These successful invasions led to decreases in abundances in more than 50% of simulations for herbivores, omnivorous echinoids, and predatory gastropods (Figure 5.15). There were no patterns of change in abundance for two groups: the predatory pycnogonids and benthic fish. Predatory echinoderms were the only group which increased in abundance in more than half of the simulations. Although the increase in abundance of *H. planatus* generally occurs quickly, the effects to the native species lag by around 10 years (Figure 5.16). At the end of the simulation period, *H. planatus*, continued to increase in abundance.

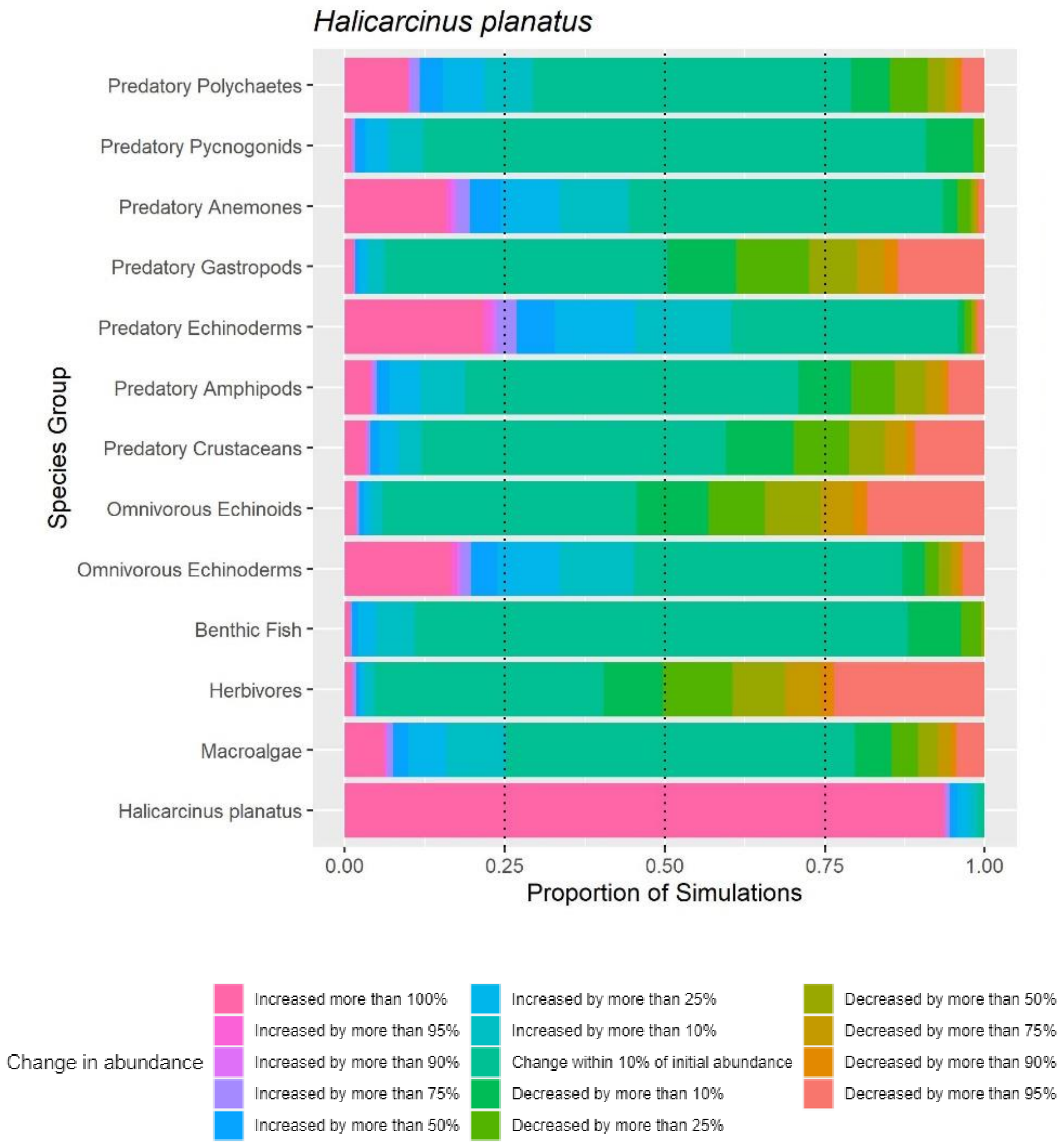


Figure 5.15 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Halicarcinus planatus*, in simulations where *H. planatus* successfully established in the simulated invasion of a stable food web (n = 1,764).

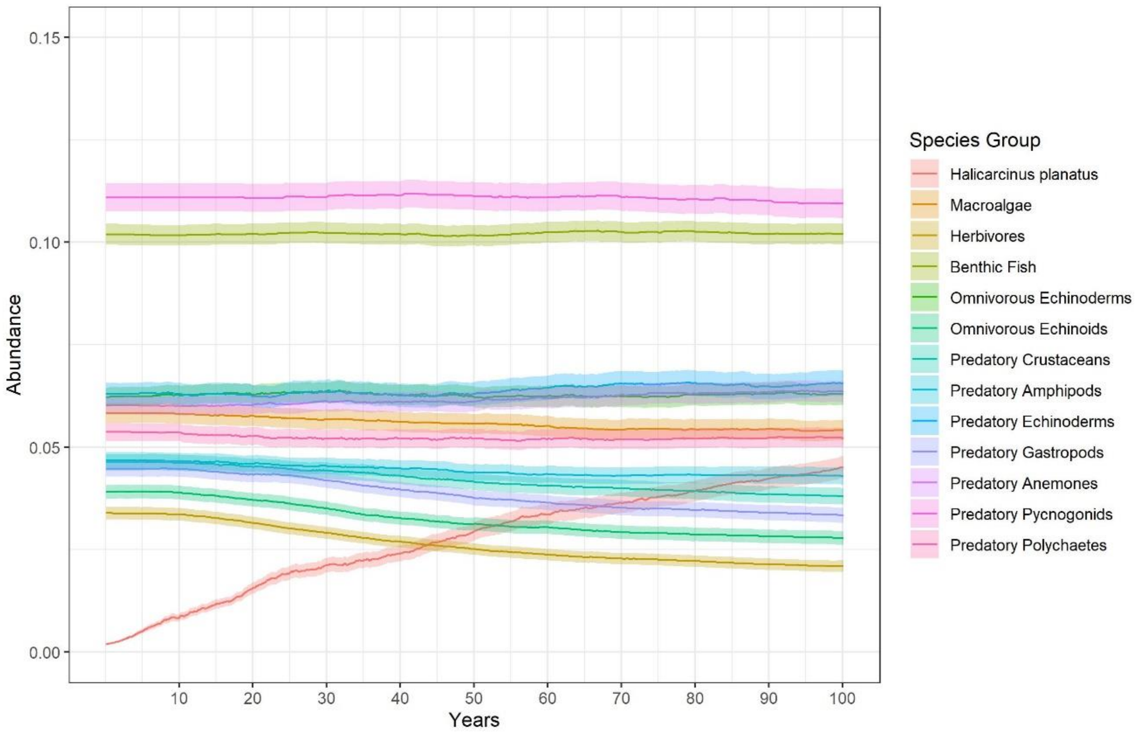


Figure 5.16 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Halicarcinus planatus* into an Antarctic endemic benthic ecosystem over a 100-year period from a subset of simulations where *H. planatus* increased in abundance (n = 1,764).

5.5.8 Invasive Species: Carcinus maenas

More than half of the simulations resulted in omnivorous *C. maenas* increasing in abundance (Figure 5.3) and of those, the majority simulated increases in abundance of more than 100% (Figure 5.4). Herbivores fared the worst, with decreases in abundance in more than 75% of simulations (Figure 5.17). Similarly, predatory polychaetes and predatory amphipods had decreases in abundance in more than half of the simulations. Only benthic fish increased in abundance in more than half of the simulations.

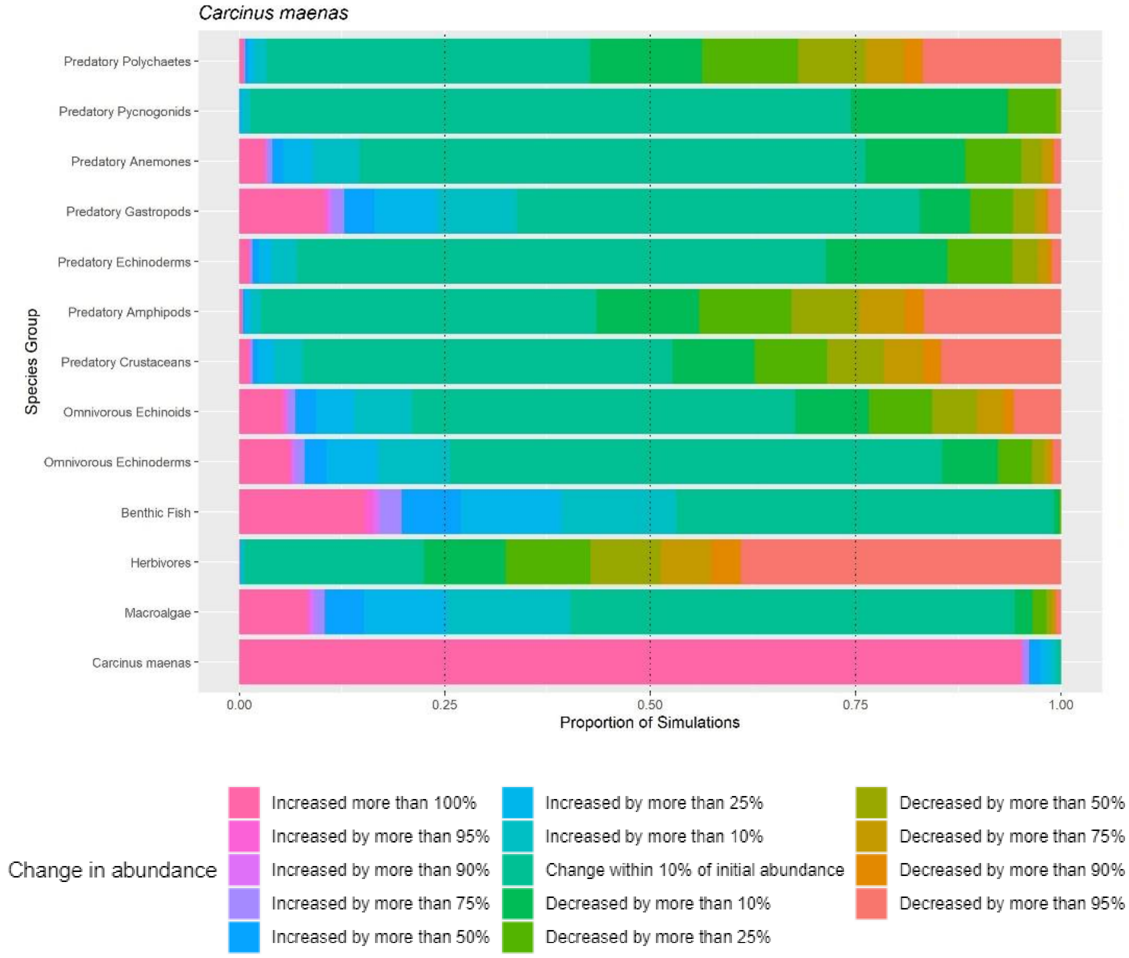


Figure 5.17 The proportion of change in abundance of endemic Antarctic species group and the invasive species, *Carcinus maenas*, in simulations where *C. maenas* successfully established in the simulated invasion of a stable food web (n = 5,223).

The abundance of *C. maenas* increases quickly from introduction and continues to increase in abundance throughout the simulation period (Figure 5.18). The abundance of benthic fish also increases dramatically in the first half of the simulation period, then tapers towards a plateau in the latter half of the simulation period.

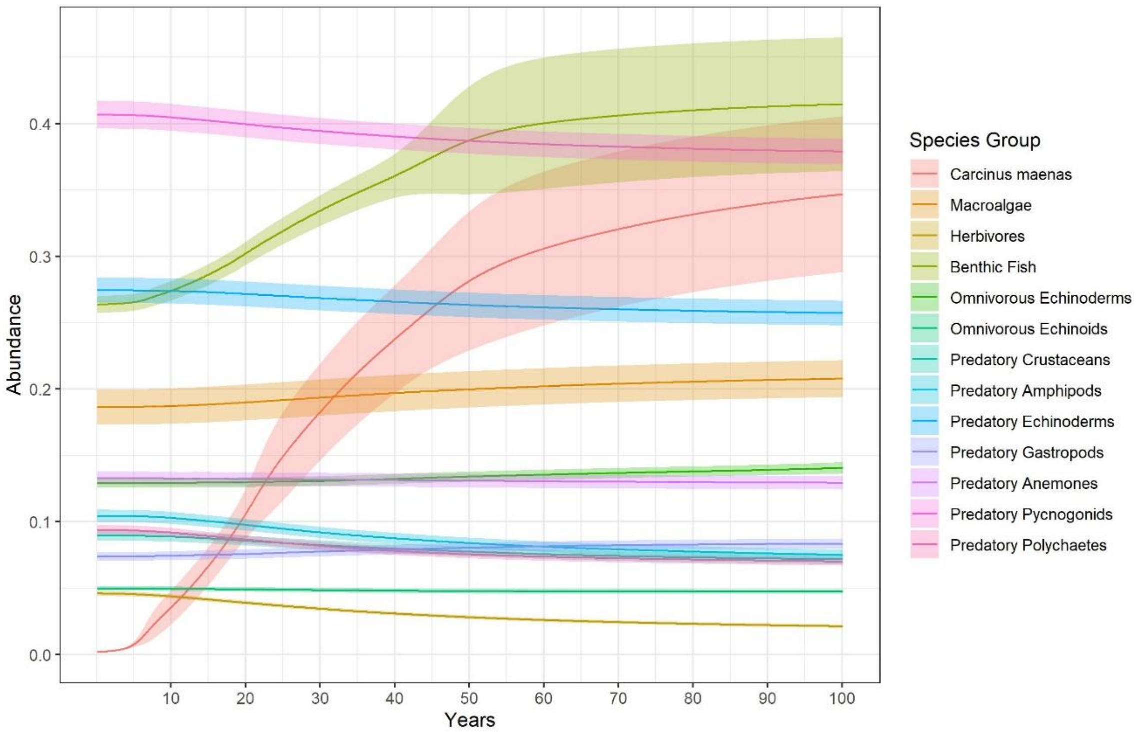


Figure 5.18 The mean abundance and 95% confidence interval of all species groups for a simulated invasion of *Carcinus maenas* into an Antarctic endemic benthic ecosystem over a 100-year period from a subset of simulations where *C. maenas* increased in abundance (n = 5,223).

5.6 Discussion

There are clear losers when it comes to the impact of MIS in an Antarctic ecosystem; namely: herbivores, predatory crustaceans, and predatory amphipods. In the simulations for all the invasive species, these groups largely reduce in abundance following the successful establishment of the invasive species. In contrast, there are no clear winners from the establishment of the invasive species, with the effect on each native species group differing depending on the invasive species being introduced. Predatory pycnogonids were largely unaffected by the establishment of any of the invasive species, and abundances generally remained around the same as the initial abundance. It is important to note, however, that these impacts rely on the successful establishment of the respective invasive species, and so far, there have been no reports of non-native species which have successfully established in the Antarctic region (Hughes et al., 2020; McCarthy et al., 2019) However there is almost no monitoring being done which would be capable of detecting such invasions. Further, the impacts reported here are based on invasive species growth rates that have been established for the species outside of Antarctic conditions, and it is noted that growth rates often decline as the temperature declines (Cockrell and Sorte, 2013). In general, the growth rates of Antarctic species are two to five times lower than their global counterparts (Peck, 2005), meaning that the potential impacts may be less than those reported herein. However, these models do not account for the possibility of multiple species being introduced, nor any ongoing propagule pressure.

The effects of multiple invasive species establishing in a new environment are difficult to predict. The impact of multiple invasive species may be additive, where the impact of each species acts independently (Johnson et al., 2009; Preston et al., 2012), synergistically, where the impact of all species is greater than the sum of impact from each separate species (Didham et al., 2005; Liversage et al., 2021), or antagonistic, where the overall effect is less than the sum of the impact from each species (Jackson, 2015). Indeed, a metanalysis of interactions between multiple invasive species in marine environments showed that the mean interaction was antagonistic (Jackson, 2015). Determining the interaction effects of multiple invasive species was outside the scope of this study, as was the determination of the concomitant effects of invasive species with changes to environmental conditions and its effect on native ecosystems. These are important considerations for determining when and how to manage invasive species, as management for a single invasive species when multiple invasive species are present may have unintended negative outcomes for native species (Bode et al., 2015; Doherty et al., 2015; Helmstedt et al., 2016).

The response of the invasive species being introduced to the native ecosystem differed between species, with some struggling to establish, such as the *Mytilus sp* bivalve group, while others, such as the crab *Charybdis japonica* and the red algae *Hypnea musciformis* successfully established more often. In the absence of optimal environmental conditions, these establishment rates are likely overestimates, and suggest that non-native species would continue to face a significant ecological barrier to invasion. Further, as the interactions between invasive species and the native Antarctic species are completely unknown, and were here based on interactions outside of the Antarctic ecosystem, there may be novel interactions that have not been observed in other regions of the world (Branch and Nina Steffani, 2004), either by a lack of co-existence of the species, or by adaptation by the invasive species to a new environment. However, the evidence from the simulations where invasive species did establish, showed that they tended to flourish, with most simulations modelling increases in abundance of more than 100%. This reinforces the general notion that preventing species from becoming established is better than trying to manage them once they are in the system.

Even though there were clear impacts on some native species group from the simulated invasions, the ability of any native species to act as a sentinel of an invasion in-progress is doubtful. Where effects are seen in the native species, they generally only occur when the invasive species is undergoing rapid increases in abundance. The only exception to this was the relationship between macroalgae and the invasive *H. musciformis*, where the increase in *H. musciformis* simultaneously mirrors the decline in native macroalgae. However, researchers would have to be knowledgeable on the taxonomic differences between native macroalgae species and the invasive species for this situation to be recognized as an invasion, as the overall abundance of algae would appear to remain constant.

The assumptions of the simulations that were discussed previously that invasive species growth rates would remain constant in an Antarctic context, and that conditions would be optimal for the invasive species, both indicate that the modelling will lead to overestimations the impact of invasive species in the Antarctic region. Conversely, another underlying assumption of the model, that the native ecosystem being modelled is stable and at equilibrium, could result in underestimating the predicted impacts (Bode et al., 2017). The oceans of the world are warming, and even though the East Antarctic is lagging behind other regions of the Antarctic with respect to warming (Meredith and King, 2005), it will not avoid temperature increases particularly in the long term (see Chapter 4). While my modelling shows that few species will suffer catastrophic levels of decline over a 100-year period, the reality is that these native species will already be under a level of stress. However, Antarctic marine ecosystems have been effectively isolated for millions of years (Tavares and De Melo, 2004) and are likely one of very few ecosystems in the world that could come close to satisfying the assumption of equilibrium.

Although there are exceptions, marine species found south of 60˚S have some of the narrowest thermal tolerances in the world (Griffiths et al., 2017; Sunday et al., 2011). Recent research shows that 79% of species found only south of the polar front are poised to lose at least some portion of their thermally suitable habitat by the end of this century (Griffiths et al., 2017). This is expected to be particularly pertinent in brooding species, like many Antarctic amphipods and isopods, where they have a limited ability to disperse in response to changes in the climate (Ingels et al., 2012).

However, warming temperatures are not the only change that will occur in the Antarctic region as a result of climate change. The increased melt of icesheets will decrease ocean salinity, and is predicted to negatively impact species like the echinoid *Sterechinus neumayeri* which have a stenohaline larval stage (Cowart et al., 2009). Further, increased acidification will have varying effect on different Antarctic species larval development, with isopods and amphipods being less affected than urchins or bivalves (Ingels et al., 2012). Ocean acidification can also affect other physiological processes such as growth, reproduction, feeding, and changed predator avoidance behaviours (Munday et al., 2020). In contrast to native species, invasive species by nature are much more tolerant of changes to the environment, although their response to changes in the environment will also likely affect the rate of successful establishment of invasions (Stachowicz et al., 2002).

The ensemble ecosystem modelling (EEM) method is an emerging tool to help predict the direct and indirect impacts of changes in a food web following a perturbance. However, given the uncertainty in the food web interactions in the Antarctic system, it would be folly to ignore expert opinion on potential impacts for each of the invasive species. The greatest disparity between the impacts predicted here, and those of expert elicitation arise in the *Mytilid* species group. The *Mytilid* species group is the highest profile potential invader to the Antarctic in the literature (Cárdenas et al., 2020; Hughes et al., 2020; Lee and Chown, 2007); however the EEM showed that it would struggle to establish into a known Antarctic food web, and would have limited impact in most scenarios. In cases such as this, it is more prudent to err on the side of caution and treat the species as having a potentially high negative impact on native species. The potential cost of managing this species as a high risk invader by developing monitoring tools and management action plans will be far less than the cost of managing this species once it has established and been found to have a higher impact on native species than predicted in my study (Bergstrom, 2022).

One current drawback to the EEM system I have used here is that networks beyond 12-15 nodes become computationally prohibitive. Testing the food web for stability required generating random strengths between species, random growth rates where growth rates were unknown, and generating random initial abundances. To end up at 10,000 plausible network models, I generated and discarded, tens of billions of simulations that did not pass the test for stability and equilibrium, for each invasive species. Ideally, we would be able to model each native species as a separate node to capture the differences in impact for the distinct species. Research is currently underway to increase the computational efficiency of this type of modelling that should allow for networks with more nodes to be simulated in a reasonable length of time.

While the EEM technique cannot predict extinctions, the abundances of some species groups may be effectively functionally extinct in ecological terms. Functional extinction is often described as occurring when there is no more successful reproduction or recruitment occurring in a population (Jarić et al., 2016). The Lotka-Volterra equation used to calculate the abundance of all the different species groups through time allows for species to decline to near-zero, but never actually zero. Another interpretation of functional extinction recognizes that the functions of species can be lost from an environment, without necessarily indicating that extinction of the species itself is inevitable (McConkey and O'Farrill, 2015). For example, the population decline of a predator may result in herbivore numbers increasing, and the subsequent removal of plant species may lead to a complete change of the ecosystem. In this example the loss of a herbivore controller is the function that has been lost, though the predatory species is still able to carry out successful reproduction. This type of functional extinction is captured in the EEM simulations, where the direct, as well as the indirect effects, of the invasive species on the (originally stable) native ecosystem are modelled through time. For example, many of the invasive species in this study were predicted to have a negative impact on herbivore species which lead to the flow on effect of increasing macroalgae abundance. Understanding the tipping points to changes of ecosystem regime shift should be the focus of further research.

This study is the first to model the potential impacts of an MIS in the East Antarctic region using an EEM. I show that even in ideal environmental conditions with no reduction in fitness, several known MIS would have difficulty establishing in an East Antarctic marine ecosystem. Those that do establish will, for the most part, have no significant impact on most native species groups, except the herbivores which are expected to decline on average in the presence of all of the invasive species. There are no clear sentinel species that would indicate the early stages of an invasion, apart from the red algae, *H. musciformis*, and even then, it would rely on an observer's ability to distinguish between the native macroalgae species and the invasive species. Information about the minimum viable populations for the native species would help interpretation of the results rather than relying on simple measures of percentage decline. Further, incorporation of the negative impacts that the native species will face under a changing climate should be explored in the context of model building to discern if the simultaneous stressors of climate change and invasive species would have a synergistic or additive effect on the native species in this region.

Chapter 6: Risk Assessment Synthesis

6.1 Preamble

This chapters synthesises the background knowledge with the outcomes of the three previous chapters to provide an overall risk assessment. The risk assessment matrix used here is adapted from the Environmental Impact Assessment for the *RSV Nuyina* (Australian Antarctic Division, 2021b) and has been updated to include the *IUCN Red List Categories* as definitions for the consequences of successful invasion. Descriptions may appear repetitive for sites with similar risk profiles.

6.2 Likelihood of Introduction

The first step on the path to invasion is for a non-native species to be introduced to a new area, either by natural or anthropogenic means. In terms of this study, the natural pathway means planktonic larvae being carried by oceanic currents from an area already hypothetically inhabited by the species, and the anthropogenic pathway means being carried as fouling on ships hulls or other marine equipment. Here, I discuss each pathway separately and define the likelihood of introduction to each of the study sites based on the criteria initially described in Chapter 2 (Table 6.1).

Table 6.1 Definitions for the likelihood of introduction.

Almost certain	Is expected to occur. Has occurred in the Antarctic region or subantarctic region in the past year.
Likely	Will probably occur. Has occurred in the Antarctic region or subantarctic region in the past two years.
Possible	Might occur at some time in the future. Has occurred in the Antarctic region or subantarctic region in the past five years.
Unlikely	Could occur but considered unlikely or doubtful. Has occurred in the Antarctic region or subantarctic region in the past ten years.
Remote	May occur in exceptional circumstances. Has not occurred in the Antarctic region or subantarctic region in the past ten years.

6.2.1 Passive Pathways

The likelihood of introduction by passive pathways to the locations studied in this thesis will be highly dependent upon the species in question. Most species have a shorter planktonic larval duration than the 12 months as discussed in Chapter 3, thus limiting the distances they are able to travel. Species such as *Asterias amurensis* and *Mytilus chilensis* have been shown to have a pelagic larval duration of 120 days and 45 days, respectively; however it is unclear how the colder waters of the polar regions may influence their pelagic larval durations. In reality, the West Antarctic Peninsula region is likely to be the first to be successfully invaded (Hughes et al., 2020; McCarthy et al., 2019).

6.2.1.1 Casey Station

Casey station is the most likely of the continental station to receive non-native species passively from other areas around the Antarctic continent and in the Southern Ocean. It is connected by oceanic processes to several other Antarctic research bases to the east of the station and showed the most extensive backtracking of any Australian continental station. There were also a significant number of particles that came from outside the coastal zone, up into the subantarctic. If non-native species are able to gain a foothold in the subantarctic region around the Kerguelen Plateau, then they may be passively spread to Casey station, assuming they have a long enough planktonic larval duration. Overall, the risk of introduction is still ***Remote,*** but of the Antarctic sites it is at the highest risk. The connection to the subantarctic is non-trivial and could provide a means of increased propagule pressure compared to the other continental sites for species with long planktonic larval durations.

6.2.1.2 Davis Station

Davis station is unlikely to receive non-native species passively from other areas of the Antarctic continent or from other areas within the Southern Ocean. It is connected by oceanic processes mainly to nearby stations in the Prydz Bay region of East Antarctica. There was very little connection outside of the Antarctic Coastal Current, with less than 0.01% of particles backtracked to cross into the Antarctic Circumpolar Current. Davis is therefore highly unlikely to receive non-native species in the plankton from the subantarctic. Overall, the risk of passive introduction of non-native species is ***Remote*** for Davis station.

6.2.1.3 Mawson Station

Mawson station is unlikely to receive non-native species passively from other areas of the Antarctic continent or from other areas within the Southern Ocean. It is connected by oceanic processes only to nearby stations within the Prydz Bay region of East Antarctic. There is no connection outside of the Antarctic Coastal Current, with no particles backtracked to cross into the Antarctic Circumpolar Current. Mawson is therefore highly unlikely to receive non-native species in the plankton from the subantarctic. Overall, the risk of passive introduction of non-native species is ***Remote*** for Mawson station.

6.2.1.4 Macquarie Island

Macquarie Island has oceanic connections to much of the subantarctic, the West Antarctic Peninsula, South America, and South Africa. However, while Macquarie Island is connected to many places, the time taken to reach these sites is often in excess of six months, and the number of particles carried from any one site to Macquarie Island is incredibly low. The closest connection is to Heard Island which is at least a three-month journey, and which is longer than many species spend in the plankton. Very few particles that left Heard Island would arrive near Macquarie Island. However, if there was a substantial population of invasive species with pelagic larval durations of at least three months, such as *A. amurensis* around Heard Island then the number of planktonic larvae that could be carried to Macquarie Island could be well into the millions or beyond. Overall, the risk of passive introduction is ***Unlikely*** for Macquarie Island.

6.2.1.5 Heard Island

Heard Island has oceanic connections to the West Antarctic subantarctic islands, the West Antarctic Peninsula, and South America. Similar to Macquarie Island, there is considerable distance between Heard Island and its closest connection, Bouvet Island, off the far west of

East Antarctica. Also, the number of particles carried from any one site to Heard Island is incredibly low. However, if non-native species were to establish on the Kerguelen Plateau, they may be able to passively spread to Heard Island. Overall, the risk of passive introduction is ***Unlikely*** for Heard Island.

6.2.2 Anthropogenic Pathways

The likelihood of introduction by anthropogenic means is currently difficult to comprehensively assess. There is a lack of fouling community data on ships that travel the Southern Ocean, and the few studies that do exist show a diverse assemblage of species being carried in Antarctic and subantarctic waters, including several known invasive species such as *Ciona intestinalis, Mytilus galloprovincialis*, and *Botryllus schlosseri* (Hughes and Ashton, 2017; Lee and Chown, 2009, 2007; Lewis et al., 2005, 2004, 2003). There have been no more recent examples of invasive species being carried as fouling on Antarctic-bound ships, but this is likely due to a lack of studies being conducted, rather than a lack of incidences. It is clear that ships pose an ongoing threat as vectors of non-native species to the Antarctic and subantarctic region, and that this is likely still occurring. Protected areas of ships' hulls, like sea chests, have been shown to harbour substantial communities of MIS (Lee and Chown, 2007).

6.2.2.1 Antarctic Coastal Locations

The Antarctic continental stations are connected by shipping mainly to Hobart, Tasmania and receive very little shipping traffic from other regions. Although there are a number of high-risk invasive species present in Hobart, travel through areas of sea-ice is likely to remove much of the fouling that may exist and leave only niche areas like sea chests and hull recesses to carry the most significant propagule loads. Overall, the propagule pressure to this region is very low, with less than 15 ship visits in any one year. Even so, there have been non-native species observed at higher latitudes than the Australian Antarctic bases, indicating that it is possible for non-native species to be introduced in areas subject to sea ice. Overall, the risk of introduction from anthropogenic vectors to the nearshore environments of the Australian Antarctic research stations is ***Likely***.

6.2.2.2 Macquarie Island

Macquarie Island is connected by shipping to Hobart, Tasmania, and to ports in New Zealand. It is the most visited of the Australian Antarctic and subantarctic sites as it receives a number of tourist vessels each year, along with research and resupply vessels. There are a number of high-risk hull-fouling invasive species in Tasmania and New Zealand. Unlike the continental sites, there is no travel through sea-ice, thereby limiting the amount of fouling that would be removed enroute. Compared to other regions of the world, Macquarie Island still receives very little shipping traffic; less than 50 ships per year. However, it is ***Almost certain*** that non-native species will be introduced via shipping.

6.2.2.3 Heard Island

Heard Island is connected by shipping to Hobart, Tasmania. It is the least visited of the Australian Antarctic and subantarctic sites, often going several years without a single ship visit. There are a number of high-risk hull-fouling invasive species in Tasmania, and unlike the Antarctic sites, there is no travel though sea ice, thereby limiting the amount of fouling that would be removed enroute. Propagule pressure from shipping is incredibly low, even compared to the Australian Antarctic sites and Macquarie Island. Overall, the risk of anthropogenic introduction of non-native species is ***Unlikely*** for Heard Island, given the very low volume of shipping traffic in the region.

6.3 Likelihood of Establishment

The second step on the path to invasion is for non-native species to successfully establish in the new area, which means they need to be able to not only survive in the new environment, but also be able to successfully undergo reproduction. This facet of the invasion process was covered in Chapter 3 (predicting species survival in the Australian Antarctic and subantarctic regions) and Chapter 4 (for the Antarctic sites: predicting the fate of invasive species in native food webs) of this thesis. Here, I discuss the findings of those chapters to classify the likelihood of success based on the criteria initially described in Chapter 2 (Table 6.2).

Table 6.2 Definitions for the likelihood of successful establishment used in the risk assessment matrix.

Likelihood of successful establishment	
Almost certain	Is expected to occur in most circumstances. Has occurred in the Antarctic region or subantarctic region in the past year.
Likely	Will probably occur. Has occurred in the Antarctic region or subantarctic region in the past two years.
Possible	Might occur at some time in the future. Has occurred in the Antarctic region or subantarctic region in the past five years.
Unlikely	Could occur but considered unlikely or doubtful. Has occurred in the Antarctic region or subantarctic region in the past ten years.
Remote	May occur in exceptional circumstances. Has not occurred in the Antarctic region or subantarctic region in the past ten years.

6.3.1 Casey Station

There were three hull-fouling species that were shown to be able to survive in the marine environment near Casey station: *Asterias amurensis*, *Charybdis japonica* and *Geukensia demissa*. Of these, only the asteroid *A. amurensis* was consistently modelled for survival in the region using varying aggregations of environmental variables. The crab *C. japonica* was shown to survive only in the highest RCP model (8.5) and the mussel *G. demissa* is doubtful as its presence was recorded only once in the Antarctic Peninsula region, never reported in the literature, and otherwise inhabits much warmer environments in North America and Spain.

To date, there is no record of *A. amurensis* being found on an Antarctic-bound vessel; however, there have been very few fouling community studies conducted on Antarctic-bound vessels. This species has been found in fouling in the Derwent River, Hobart, which is the same location as the home port for Australia's ice breaker ships, *Aurora Australis* (retired) and as of this year, the new *RSV Nuyina*. Physiological experiments on the Tasmanian population of *A. amurensis* currently show that they would be unable to complete their life cycle in the Antarctic waters; however much of the species global range extends into the Arctic. *A. amurensis* was unable to establish in most simulations when introduced to a native Antarctic ecosystem, with a population decline of more than 95% in more than half of the simulations. Still, if this species does become established it will likely proliferate. On the other hand, the crab *C. japonica* was predicted to successfully establish in most simulations and when it does establish the population increases significantly.

Another aspect that needs to be considered alongside these results is the fact that while Antarctic environmental conditions may become more suitable for non-native species, they will concurrently become less suitable for many of the native Antarctic species that have very little tolerance for changes in temperature. In addition to this, climate change is likely to cause an increase in disturbance in native ecosystems (e.g., increased iceberg scour) which could open up new habitats for non-native species to exploit.

While the likelihood of establishment is remote currently, given the environmental changes that will continue to threaten native Antarctic ecosystems, it is ***Unlikely*** that non-native species will be able to establish near Casey Station.

6.3.2 Davis Station

Three hull fouling species were shown to be able to survive in the marine environment near Davis station: *Asterias amurensis*, *Geukensia demissa* and *Hypnea musciformis*. The modelling for *G. demissa* is doubtful as its presence was recorded only once in the Antarctic Peninsula region, never reported on in the literature, and otherwise inhabits much warmer environments in North America and Spain. The red algae *H. musciformis* is a global invasive species with a wide latitudinal range from the tropics to the Arctic. However my modelling showed that the current distribution is driven by pH and that the conditions at Davis would only be suitable for a short time period before projected acidity increases by mid-century.

To date, there is no record of *A. amurensis* being found on an Antarctic-bound vessel; however, there have been very few fouling community studies conducted on Antarctic-bound vessels. This species has been found in fouling in the Derwent River, Hobart, which is the home port for Australia’s ice breaker ships, *Aurora Australis* (retired) and as of this year, the new *RSV Nuyina*. Physiological experiments on the Tasmanian population of *A. amurensis* currently show that they would be unable to complete their life cycle in the Antarctic waters; however much of the species global range extends into the Arctic. *A. amurensis* was unable to establish in most simulations when introduced to a native

Antarctic ecosystem, with a population decline of more than 95% in more than half of the simulations. Still, if this species does become established it will likely proliferate. On the other hand, *H. musciformis* was able to successfully establish in more than half of the simulations and when it does establish the population increases significantly.

Another aspect that needs to be considered alongside these results is the fact that while Antarctic environmental conditions may become more suitable for non-native species, they will concurrently become less suitable for many of the native Antarctic species that have very little tolerance for changes in temperature. In addition to this, climate change is likely to cause an increase in disturbance in native ecosystems (e.g., increased iceberg scour) which could open up new habitats for non-native species to exploit.

While the likelihood of establishment is remote currently, given the environmental changes that will continue to threaten native Antarctic ecosystems, it is ***Unlikely*** that non-native species will be able to establish near Davis Station.

6.3.3 Mawson Station

There were two hull fouling species that were shown to be able to survive in the marine environment near Davis station: *Asterias amurensis* and *Geukensia demissa*. The modelling for *G. demissa* is doubtful as its presence was recorded only once in the Antarctic Peninsula region, never reported in the literature, and otherwise inhabits much warmer environments in North America and Spain.

To date, there is no record of *A. amurensis* being found on an Antarctic-bound vessel; however, there have been very few fouling community studies conducted on Antarctic-bound vessels. This species has been found in fouling in the Derwent River, Hobart, which is the home port for Australia's ice breaker ships, *Aurora Australis* (retired) and as of this year, the new *RSV Nuyina*. Physiological experiments on the Tasmanian population of *A. amurensis* currently show that they would be unable to complete their life cycle in the Antarctic waters; however much of the species global range extends into the Arctic. *A. amurensis* was unable to establish in most simulations when introduced to a native Antarctic ecosystem, with a population decline of more than 95% in more than half of the simulations. Still, if this species does become established it will likely proliferate.

Another aspect that needs to be considered alongside these results is the fact that while Antarctic environmental conditions may become more suitable for non-native species, they will concurrently become less suitable for many of the native Antarctic species that have very little tolerance for changes in temperature. In addition to this, climate change is likely to cause an increase in disturbance in native ecosystems (e.g., increased iceberg scour) which could open up new habitats for non-native species to exploit.

While the likelihood of establishment is remote currently, given the environmental changes that will continue to threaten native Antarctic ecosystems, it is ***Unlikely*** that non-native species will be able to establish near Mawson Station.

6.3.4 Macquarie Island

There were two hull fouling species that were shown to be able to survive in the marine environment near Macquarie Island: *Asterias amurensis* and *Undaria pinnatifida*. To date, there is no record of either of these species being found on a vessel travelling in the subantarctic; however there have been few fouling community studies conducted on ships that operate in the region. Both *A. amurensis* and *U. pinnatifida* have been identified as potential threats to the region by other studies. Studies have shown that *A. amurensis* is able to undergo its complete life cycle in the waters around Macquarie Island. The ocean conditions of the subantarctic are already more suitable for non-native species than the Antarctic sites, though the lack of disturbance in shallow marine environments coupled with strong wave action and ocean currents are likely to be a barrier to the establishment of non-native species.

Based on this information, establishment is ***Likely*** at Macquarie Island.

6.3.5 Heard Island

Two hull fouling species were shown to be able to survive in the marine environment near Macquarie Island: *Asterias amurensis* and *Undaria pinnatifida*. To date, there is no record of either of these species being found on a vessel travelling in the subantarctic; however there have been few fouling community studies conducted on ships that operate in the region. Both *A. amurensis* and *U. pinnatifida* have been identified as potential threats to the region by other studies. The ocean conditions of the subantarctic are already more suitable for non-native species than the Antarctic sites, though the lack of disturbance in shallow marine environments coupled with strong wave action and ocean currents are likely to be a barrier to the establishment of non-native species. Also, depending on the season, the polar front which demarks the warmer northern waters from the cold polar waters sits either just above or just below Heard Island. This makes the ocean around Heard Island colder than Macquarie Island even though Heard Island is situated slightly north latitudinally of Macquarie Island.

Based on this information, establishment is ***Unlikely*** at Heard Island.

6.4 Consequence of Successful Invasion

Although many non-native species have established in seas around the world, not all of those have negative impacts on their invaded range. Impacts are notoriously difficult to predict, so I explored a range of possible future scenarios based on a single invasion event for each of several high-risk non-native species discussed in Chapter 5. Here, I discuss the findings of Chapter 5 to classify the potential impacts based on the criteria initially described in Chapter 2 (Table 6.3).

6.4.1 Antarctic Coastal Locations

There are a range of plausible future scenarios following the simulated release of a known MIS into the Antarctic food web, based on community composition surveys completed in the Windmill Islands near Casey Station. All seven MIS were capable of negatively impacting the native ecosystems, though there are varying degrees of severity. Even so, all MIS were predicted to reduce the abundance of several native species groups by more than 95% in at least some of the plausible future scenarios. Therefore, the potential consequences of successful invasions into the Antarctic coastal ecosystems are ***Catastrophic*.**

6.4.2 Subantarctic Islands

There is a paucity of information on shallow coastal marine community structure in the Australian subantarctic islands. This means it is difficult to predict the impacts of MIS in these areas. A range of overall risk will be provided for the potential consequences of successful invasions.

Table 6.3 Definitions for the consequence of successful invasion used in the risk assessment matrix, adapted from the RSV Nuyina risk assessment matrix, and updated to include the IUCN Red List Categories definitions.

Consequence of successful invasion	
Catastrophic	Severe, widespread, and irreversible environmental damage. Local extinction of species OR loss of genetic diversity OR impact to one or more threatened species. Native species decline to Critically Endangered levels (population decline of 80% or more).
Major	Major environmental damage with ongoing impacts. Substantial loss of population OR potential loss of genetic diversity OR loss of individuals of threatened species. Native species decline to Endangered levels (population decline of 50% or more).
Moderate	Significant environmental damage with the potential for reversal with intensive interventions. Minimal impact on populations OR some impacts on individuals of threatened species. Native species decline to Vulnerable levels (population decline of 30% or more).
Minor	Isolated, but significant environmental damage with the potential for reversal with intensive interventions. Some individuals are impacted. No population impacts and no impact to threatened species.
Insignificant	Minor environmental damage that can be reversed. No observable change to native species.

6.5 Overall risk

The overall risk assessment is based on the likelihood of a successful establishment of the invasive species and the potential impacts of that invasive species on the native ecosystem. However, it is also important to recognize that invasion is a stepwise process, where each step must be completed successfully for the next step to occur. Therefore, the assessment of risk is not complete without the risk of introduction occurring also. Here, I summarize the overall risk to each of the study sites in terms of risk of introduction and the overall risk if establishment of the invasive species occurs. The latter part of this assessment (Figure 6.1) is based on the *Risk Assessment Framework* described in Chapter 2. The continental stations have been combined as they shared the same risk profile.

	Consequence of successful invasion				
Likelihood of successful establishment	**Insignificant**	**Minor**	**Moderate**	**Major**	**Catastrophic**
Almost certain	Medium	Medium	High	Severe	Severe
Likely	Low	Medium	High	High	Severe
Possible	Low	Low	Medium	High	High
Unlikely	Low	Low	Medium	Medium	High
Remote	Low	Low	Low	Medium	Medium

Figure 6.1 Risk assessment framework adapted from the *RSV Nuyina* risk assessment matrix (Australian Antarctic Division, 2021b).

6.5.1 Continental Stations: Casey, Davis, and Mawson

Risk of Passive Introduction:	*Remote*
Risk of Anthropogenic Introduction:	*Likely*
Risk of Establishment:	*Unlikely*
Consequence of Successful Invasion:	*Catastrophic*
Overall Risk:	***High***

Risk of Introduction: Although passive connections to other shallow water environments of the Antarctic and subantarctic are very limited, there is a likely risk of introduction by anthropogenic means, most likely as fouling on ships. Living specimens of known invasive species have already been observed on ships which have undergone a voyage to latitudes higher than those of the Australian Antarctic research bases.

Risk of Establishment: There was only one invasive species that was predicted to be able to survive in the seas around all of the Australian Antarctic bases: *Asterias amurensis*. With present knowledge, this species would be unable to undergo successful reproduction, although this could change with changes in climate and subsequent changes that will likely disadvantage the native ecosystems. When introduced to a simulated native food web, *A. amurensis* struggled to successfully establish in most scenarios; however there were still simulations which did allow this sea star to establish and flourish.

Consequence of Successful Invasion: Simulated invasions of seven high risk MIS into the food web based on survey data from Casey Station showed that all invasive species can plausibly lead to reductions in the abundance of native species groups by 95% or more.

***Overall Risk*:** The risk of MIS to the Australian Antarctic regions is high, owing largely to the potential impacts that invasive species could have on native ecosystems.

6.5.2 Subantarctic Island: Macquarie Island

Risk of Passive Introduction:	*Unlikely*
Risk of Anthropogenic Introduction:	*Almost Certain*
Risk of Establishment:	*Likely*
Consequence of Successful Invasion:	*Insignificant – Catastrophic*
Overall Risk:	***Low to Severe***

Risk of Introduction: Macquarie Island has a passive connection to the Kerguelen Plateau, where Heard Island is located. Although the connection is weak, a significant number of propagules could arrive at Macquarie Island if there was a significant population of invasive species with a pelagic larval duration of three months or more. Macquarie Island is the most visited location of the five study sites and is the only site to receive tourist ships. The lack of sea ice to remove fouling species, and the fact that departure ports currently harbour a number of high-risk invasive species means that non-native species are almost certain to be introduced to this island.

Risk of Establishment: The warmer waters surrounding Macquarie Island are likely to be more suitable for non-native species establishing from lower latitudes. The sea star, *Asterias amurensis*, has been predicted to be able to undergo its entire life cycle in the waters around Macquarie Island. Strong waves and ocean currents along with a lack of disturbed habitats or maritime anthropogenic structures may hinder the establishment of non-native species in this region.

Consequence of Successful Invasion: There is a paucity of data on the marine community structure around Macquarie Island. This means that it is difficult to predict the impact of MIS in this region.

***Overall Risk*:** The overall risk for Macquarie Island is somewhere between low and severe, depending on the potential impacts on the native ecosystems.

6.5.3 Subantarctic Island: Heard Island

Risk of Passive Introduction:	*Unlikely*
Risk of Anthropogenic Introduction:	*Unlikely*
Risk of Establishment:	*Unlikely*
Consequence of Successful Invasion:	*Insignificant – Catastrophic*
Overall Risk:	***Low to High***

Risk of Introduction: Heard Island is the least visited of the five Australian regions of the Southern Ocean. Risk of introduction is unlikely due to its remoteness and its lack of visits, which may occur only once every several years or less. However, the islands sit atop the Kerguelen Plateau which also includes the French subantarctic islands, roughly 500 km to the northwest. The relatively shallower waters of the plateau may enable the passive spread of MIS were they able to establish in this region. However, it is relatively isolated from other coastal regions in the Southern Ocean area.

Risk of Establishment: Despite being located at a similar latitude to Macquarie Island, the waters around Heard Island are colder due to its proximity to the Polar Front. This likely makes the habitat less suited for MIS for lower latitudes, although conditions are still milder than those around the Antarctic continent. The endemic species around the islands are also likely to have a higher tolerance to environmental change than their Antarctic counterparts, giving them a higher resilience against climate change that will help prevent MIS establishment from occurring.

Consequence of Successful Invasion: There is a paucity of data on the marine community structure around Heard Island. This means that it is difficult to predict the impact of MIS in this region.

Overall Risk**:** The overall risk for Heard Island is somewhere between low and high, depending on the potential impacts on the native ecosystems.

Chapter 7: Evaluating management of marine non-native species in the Antarctic: a case study of regulatory responses covering Australia's Antarctic bases and subantarctic islands

7.1 Preamble

This chapter examines Australian policy related to MIS and identifies any gaps in these policy instruments. MIS are an emerging threat and require robust policy responses in place to reduce the likelihood of species introductions, monitor for non-native species in the region, and allow for swift control of a realized threat. Such policy responses will put Australia in a strong position to limit the potential impacts that invasive species could have on the coastal benthic systems near Australia's Antarctic research bases and subantarctic islands.

7.2 Abstract

There is growing recognition of the threat of MIS to the coastal ecosystems of Antarctica and the subantarctic islands. This is most likely to occur as a result of shipping to the area, with hull fouling being identified as the key pathway non-native species will take. To date, there are no known established non-native marine species in this region. However, to ensure that these environments remain pest-free there needs to be suitable policies in place to prevent the introduction of non-native species, to monitor for non-native species, and to manage non-native species if discovered in the Antarctic or subantarctic. Here, I used the driver-pressure-state-impact-response (DPSIR) framework to develop a social-ecological model of the system to identify a number of policy responses that could be taken to respond to this threat. Then, using these responses as search terms, I examined the range of Australian policy instruments to determine if the threat of MIS is covered in the existing policy landscape, or if there were gaps for any of the policy responses identified earlier. I found that all policy responses were present, in at least a basic sense, in the current policy instruments, but that policy responses specific to MIS were lacking. Of key concern, are

the lack of policies that deal with monitoring and surveillance for non-native species, as well as policies relating to monitoring of control activities that could occur in response to a realized case of non-native species in the region. MIS are particularly difficult to manage once they become established, especially so in an area as remote as Antarctica or the Australian subantarctic islands, Macquarie Island and Heard Island. Robust policy and strong regulatory practices can enhance the current regulatory instruments to ensure that these regions remain pest-free.

7.3 Introduction

The marine environment of the Antarctic and subantarctic remain one of the few places on Earth not yet affected by invasive species. However, with climate change and increased shipping activity in the region, this threat may yet be realized (McCarthy et al., 2019). Whilst it has been commonly assumed that the harsh environment would keep out potential MIS, recent work has highlighted that Antarctic and subantarctic ecosystems could already be under threat from species that can tolerate the conditions, now or in the future (Chapter 4; Byrne et al., 2016; Lee and Chown, 2007; Lewis et al., 2006). In order to establish in a new environment, invasive species must: a) be on a pathway to the new region, and b) be able to survive, reproduce and maintain a self-sustaining population in that new environment. As we are unable to control whether a species can survive in the new environment, we must work on preventing MIS from entering pathways to the Antarctic and subantarctic.

Throughout this thesis I have shown that the most likely point of entry for MIS to the Antarctic and subantarctic islands is via hull fouling – fouling on a ships' outer surface (Chapters 3 and 4). Historically, ballast water has driven the global expansion of MIS (Drake and Lodge, 2007). This is unlikely to occur in the Antarctic, as ships are emptied of ballast before being loaded in gateway ports, and ballast is taken on in the Antarctic after the unloading of goods, however the topic of ballast water will be reviewed alongside hull fouling as a pathway for MIS to this region (Barnes et al., 2006; Lewis et al., 2003). While there is a small chance of passive introductions of MIS occurring with species with long pelagic larval durations, it is the anthropogenic pathway over which we have greater control. The identification of hull fouling as the key pathway by which MIS could enter the

Antarctic and subantarctic regions facilitates an evaluation of the coverage and appropriateness of policy responses.

In order to prevent the accidental or intentional transfer of MIS to the Antarctic and subantarctic regions there must be polices and regulations in place to manage this threat. However, as the threat of MIS has received relatively little attention when compared to terrestrial invasive species, policy development and research to manage for marine invasions has lagged (Hughes and Pertierra, 2016). As such, policies and guidelines designed for non-native species management in the Antarctic and subantarctic are generally presented in terms of preventing the transfer of terrestrial species to these regions, for example, of plant seeds or insects (Hughes and Pertierra, 2016). However, there has been a recent push to enhance our understanding of the potential threat of MIS and how to predict and manage for them.

The issue of marine non-native species in the Antarctic is a social-ecological problem, where human behaviour influences the activities that occur in the region leading to an altered threat level for an introduction to occur (Folke et al., 2007; Ostrom, 2007; Young et al., 2008). Therefore, we need a conceptual model to explicitly capture the connections that underpin this system (Anderies et al., 2004; Guerrero et al., 2021; Ostrom, 2007). With this system model, a range of suitable policy responses can be identified that cover each critical component of the system. These policy responses then form the basis for a policy gap analysis, where the current institutional framework is examined to identify where there are policy responses are in place, and to identify where there are deficiencies in the existing policies. The responses are classified by the stage of the invasion process they target to elucidate areas requiring attention.

7.4 Methods

There were three key steps in undertaking the gap analysis of Australian policy responses to the threat of MIS transfer by ships to the continental Australian Antarctica bases, Macquarie Island and Heard Island. First, I developed a systems model based on the driver-pressure-state-impact-response (DPSIR) framework and categorized the types of policy responses of relevance to this problem. I also identified at which stage of the invasion process each of the policy responses would apply. Second, I scoped all existing policy instruments that relate to the natural environment of the study areas. Finally, to identify gaps I reviewed each of the policy instruments to determine if they contained mention of the policy responses, or synonyms of the policy responses (Appendix C), and to which stage/s of the invasion process they applied to.

7.4.1 Case study

Australia was one of the original twelve signatories to the Antarctic Treaty (adopted in 1959, entered into force 1961*)* and has remained an active member of the global Antarctic community. Australia has three Antarctic research stations, Casey, Davis, and Mawson, all located in East Antarctica. Australia also has two subantarctic island groups: Macquarie Island, and the Heard and McDonald Island group (herein referred to as Heard Island). The continental bases are managed under the Antarctic Treaty System, the Heard and McDonald Island group are under Australian Federal jurisdiction, and Macquarie Island is under the jurisdiction of the Australian state of Tasmania. Therefore, despite all these sites being managed by the federal Australian Antarctic Division, the policy and regulations that are in play vary and thus, the fit of policy instruments in relation to Australia's interests in the Antarctic and subantarctic region is not necessarily uniform.

7.4.2 System model

The first step in evaluating this problem from a system's perspective is to construct a system model of the MIS transfer problem, which in this case is the transfer of MIS by ships as hull fouling. The DPSIR framework consists of five components (Smeets and Weterings, 1999)(Figure 7.1).

The DPSIR framework was developed by the European Environmental Agency as a standard for environmental reporting by its member states (Smeets and Weterings, 1999). The framework has been used extensively in its original and modified forms as a decision-making tool in the environmental realm, including Australian national *State of the Environment* reports (Jackson, 2017) and in aquatic invasive species risk assessments (Majorošová, 2016; Panov et al., 2009). An identified advantage of the DPSIR framework is that it encapsulates environmental, societal, cultural, and economical components of a system and explicitly captures their relationships (Bradley and Yee, 2015). As invasion is a multi-step process, the DPSIR model allows for the range of institutional responses for the different stages of an invasion, from before a species is entrained in or on a vessel or equipment, through to management of an established population of MIS (Figure 7.1).

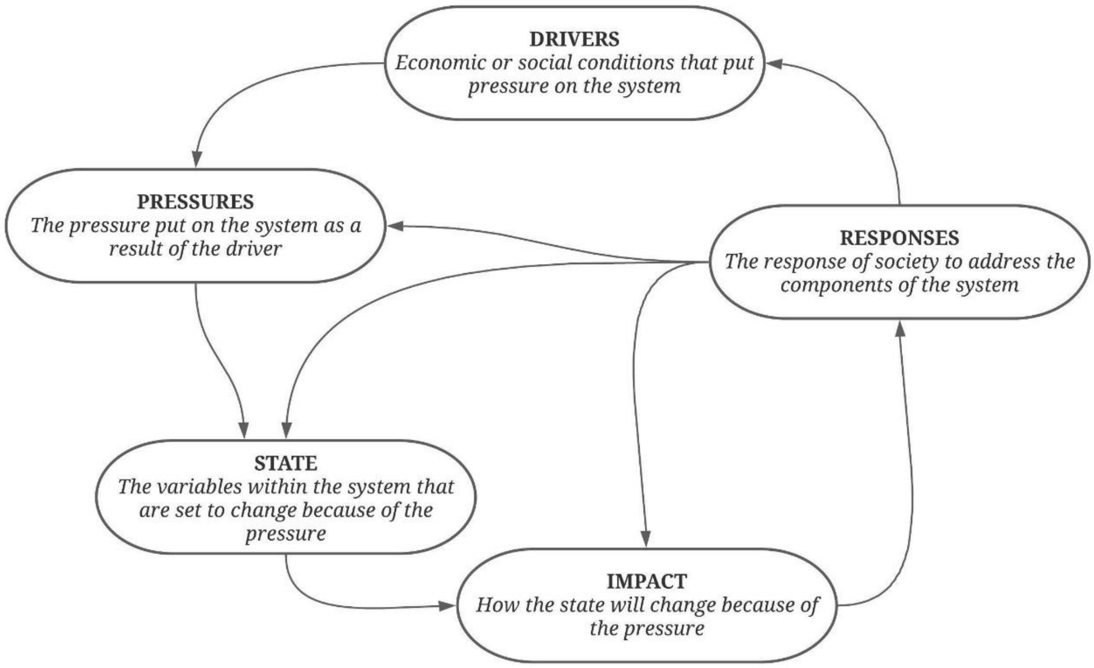


Figure 7.1 The DPSIR framework with interactions between the different components.

7.4.3 Scoping policy instruments

Policy instruments included all international policy instruments, Australian national and state legislation, regulations, policies, plans and other formal and informal instruments that relate to different components of the DPSIR model. Policy instruments were sourced from publicly available online databases and websites of international and governmental agencies, and industry-specific authorities. Note, however, that although climate change is a known driver of the social-ecological system explored in this project, policies specifically related only to climate change have been excluded from analysis.

7.4.4 Gap analysis

Each of the scoped policy documents were reviewed for keywords related to each of the potential policy responses identified in the systems model. Policy responses were classified according to the stage of invasion they applied: prevention of introduction and establishment, monitoring and surveillance, or control of realized introduction or establishment.

For each stage of the invasion responses the policy responses were classified into five levels of applicability:

- Explicitly mentioned in the policy document and the policy response is directly linked to MIS
- Implicitly mentioned in the policy document with a policy response directly linked to invasive species in a general scope that includes MIS
- Unrelated invasive species mentioned in the policy document but does not and cannot apply to marine invasive species
- General comments in the policy document relating to invasive species but focused on indirect impacts of undefined activities
- Not mentioned in the policy document with no policy responses that are linked to invasive species.

Policies directly related only to the system drivers were excluded from the analysis.

7.5 Results

7.5.1 Systems model

Antarctica and the greater Southern Ocean region are currently free of many of the drivers of change affecting other regions of the world (Figure 7.2). However, there are two key drivers that could increase the risk of MIS reaching and establishing in these regions. The first is climate change, where changes to ocean temperatures and other physical conditions mean that the once hostile environment of the polar and subpolar waters are becoming more suitable for non-native species to inhabit (Chapter 4; McCarthy et al., 2019). The pressures being exerted on the system fall under two key categories: increased human presence in the region, and changes to the marine environment as a result of climate change. Separately these pressures increase the risk of successful marine non-native species introductions by increasing the potential load of propagules being carried to the Antarctic region, and by reducing the fitness of Antarctic and subantarctic marine ecosystems and increasing the likelihood of habitat suitability by non-native species, respectively. Both are equally important in ensuring the success of a non-native species introduction, but as the invasion process goes through a sequential series of steps, with each step being dependent upon the successful completion of the previous step, the most effective way to stop non-native species introductions from shipping is to prevent them being on pathways to begin with (Hughes and Pertierra, 2016).

Unfortunately, introductions of MIS are already happening in the Antarctic region (Cárdenas et al., 2020; Lee and Chown, 2007). These generally occur in the warmer West Antarctic Peninsula region, which may lead to another problem. Marine organisms with a planktonic life phase could be carried around the continent in ocean currents. Although some locations may be protected from these ocean pathways (Chapter 2), there is still a risk that these regions are at risk of passive marine invasive introductions. Therefore, it is prudent to ensure that there are institutional responses at each stage of an invasion, as acting quickly ensures the best possible outcomes when dealing with marine non-native species (Committee for Environmental Protection, 2019).

Policy responses can be broken down into three key categories: *prevention*, *monitoring*, and *control* of a realized threat (Committee for Environmental Protection, 2019). Each potential policy response can apply to one or more than one of these key categories, with varying levels of effectiveness and cost and/or effort to implement (Figure 7.2).

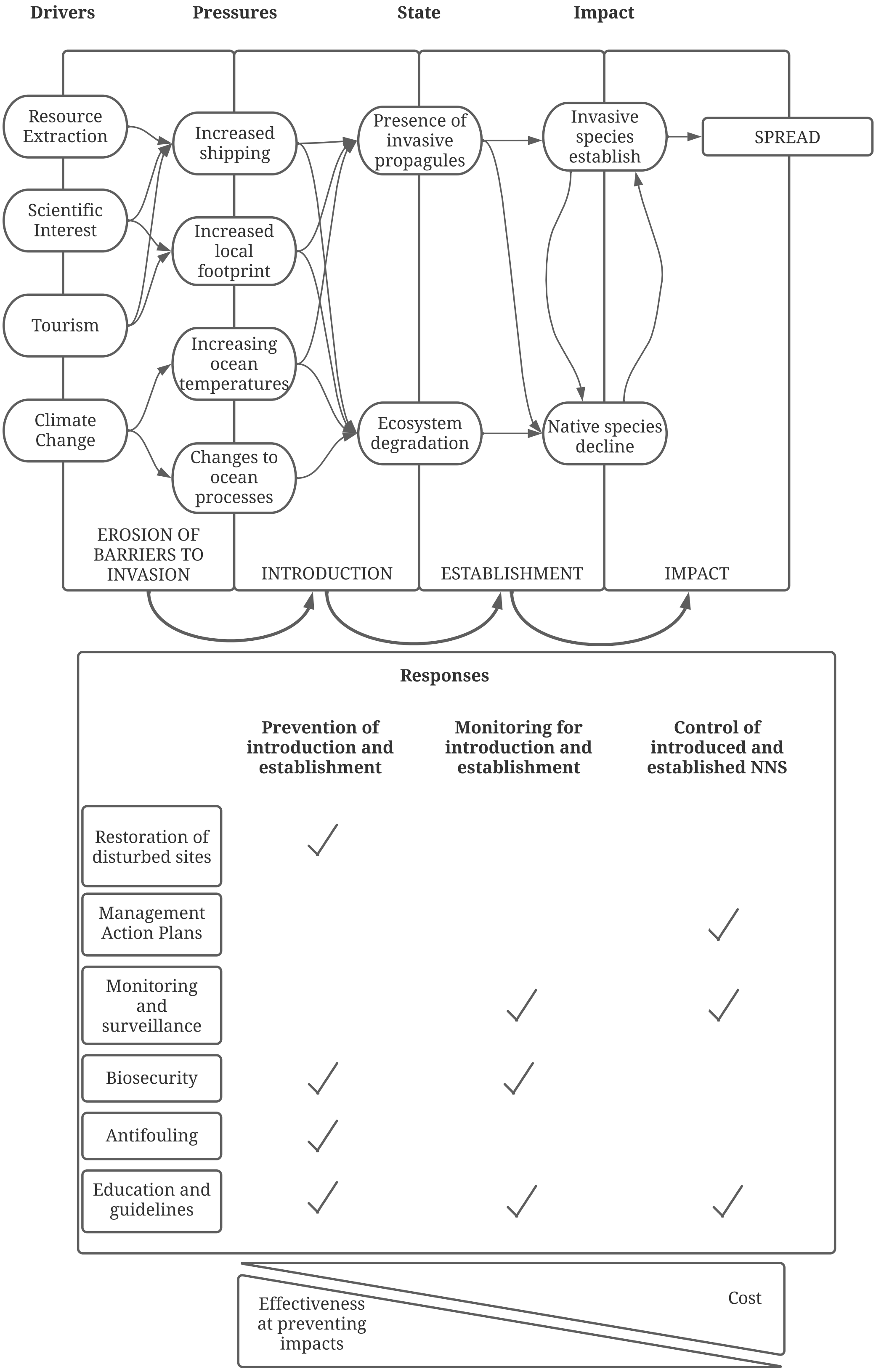


Figure 7.2 DPSIR model of the marine invasive species risk to Antarctica, and how each component relates to the invasion process. Also shows the potential policy responses from the various stages of the invasion process.

7.5.1.1 Prevention of introduction and establishment

The first step to invasion is introduction to a new area. In respect to this study, this means the initial introduction of a non-native species from outside of the Southern Ocean region to either the shallow marine environments near Australia's Antarctic research stations or the Australian subantarctic islands. It also covers the potential for non-native propagules to be introduced after a species has already established as policy responses would be similar. Policy responses to *prevent* introducing non-native species by shipping to these regions would include biosecurity protocols that prohibit or restrict the movement of non-native species. Antifouling includes all aspects relating to hull cleaning, antifouling treatments, and ballast water exchange before being allowed to operate in the Southern Ocean, whether that be as a national operator, tourist vessel, or fishing vessel. Increased resilience in complex native ecosystems makes them less prone to invasion, which can be aided by policy to restore and rehabilitate degraded ecosystems (Downing et al., 2012). Around the Australian Antarctic bases there are remediation programs that aim to clean up pollution from anthropogenic means, such as fuel spills and sewerage outflows (Stark et al., 2014). Education and guidelines are discussed further below.

7.5.1.2 Monitoring for introduction and establishment

The greater Antarctic region currently has no known established populations of MIS. However, there have been several recordings of non-native species in this region, and these have been happened upon by chance (Cárdenas et al., 2020; Hughes et al., 2020; McCarthy et al., 2019). There is currently no systematic approach of monitoring for non-native species, though there is growing recognition amongst the academic community and national operators that this needs to be a priority area (Committee for Environmental Protection, 2018). Monitoring for MIS is a time consuming and often laborious exercise (Bott et al., 2010), which is exacerbated by the hostile environment and remoteness of the Southern Ocean. Biosecurity is also included as a policy response for monitoring for invasive species. For this purpose, monitoring in a biosecurity sense means that pathways are monitored for non-native species, as opposed to monitoring the potential recipient regions for non-native species.

7.5.1.3 Control of introduced and established non-native species

The identification of a non-native species in the Antarctic and Southern Ocean region should trigger a management plan to deal with the threat (Committee for Environmental Protection, 2019). It should include methods of rapid responses, but also allow for adaptive practices due to the wide range of non-native species that could be encountered, as well as working in a hostile and unpredictable environment. Further, there should be protocols to facilitate dissemination of information regarding the non-native species discovery and management to other regional stakeholders (León et al., 2021). Importantly, there should be effective monitoring and surveillance following a management action to ensure a favourable outcome from the management activity, as well as monitoring nearby sites for spread of the invasive species. At this stage the prevention of intra-Antarctic transfer of non-native species should be prioritized, by for example, quarantining ships in regions with known non-native species from travelling to other regions of the Antarctic or subantarctic. Extra caution should be taken to prevent ongoing propagule pressure by ensuring strong biosecurity at departure points as well as keeping a high standard of suitable antifouling.

7.5.1.4 Education and guidelines

These are documents provided by organizations which provide users with useful information and guidance around the five other policy responses. Education and guidelines appear as a separate table with respect to these informal plans, strategies, and guideline documents.

7.5.2 Scoped policy documents

Australia has ratified numerous international agreements related to Antarctica, conservation of biodiversity, and shipping practices and standards (Table D1.1). The Antarctic Treaty System (ATS) oversees the governance of the Antarctic region south of 60˚S. Within the scope of the ATS sits the *Antarctic Treaty* (adopted 1959, entered into force 1960). Under the *Antarctic Treaty* are several policy instruments that are applicable to the conservation of coastal marine biodiversity in the Antarctic Treaty area.

The *Protocol on Environmental Protection* (known as the Madrid Protocol, adopted 1991, entered into force 1998) is the key policy instrument tasked with protecting the natural environment and ecosystems in the Antarctic Treaty area. The *Madrid Protocol* requires the completion of environmental impact assessments (EIA) for any activities that may cause an impact in the Antarctic Treaty area, including the deployment of new expedition vessels such as the new Australian icebreaker ship, *RSV Nuyina*, which has recently commenced use.

The International Maritime Organization (IMO) has also developed policy instruments related to shipping: the *International Convention for the Control and Management of Ships' Ballast Water and Sediments* (adopted 2004, entered into force 2017), the *International Convention on the Control of Harmful Anti-fouling Systems* (adopted 2001, entered into force 2008). The IMO has since developed the *Polar Code* (resolution *MEPC.264(68)*, adopted 2014, entered into force 2015), specifically for vessels operating in the polar regions. Under the *Polar Code* there are specific operational requirements that relate to ships operating in polar waters, given the increased hazardous nature of the polar regions. In relation to non-native species, the *Polar Code* references two key IMO resolutions: *MEPC.163(56): Guidelines for Ballast Water Exchange in the Antarctic Treaty Area* and *MEPC.207(62): Guidelines for the control and management of ships' biofouling to minimize the transfer of invasive aquatic species*.

The United Nations Division for Ocean Affairs and the Law of the Sea has developed the *Convention on the Law of Sea* (UNCLOS, adopted 1982, entered into force 1994) as a legal framework for maritime activities globally. Related to non-native species are the requirements on the Parties to avoid the transfer of non-native species to other Parties' waters or to international waters.

Heard Island is not part of the ATS, though it is captured within the *Convention on the Conservation of Antarctic Marine Living Resources* (adopted 1980, entered into force 1982) due to its location just inside the Antarctic Convergence Zone, which is the demarcation between the cold polar waters of the Southern Ocean and the warmer northern waters of the world's other oceans.

Macquarie Island and Heard Island are located outside the Antarctic Treaty area. Therefore, they fall under the guise of the United Nations Environment Programme with respect to biodiversity conservation at the international level through the *Convention on Biological Diversity* (CBD, adopted 1992, entered into force 1993). Both islands were added to the World Heritage List in 1997 under the *World Heritage Convention* (adopted 1972, entered into force 1975) for their Outstanding Universal Values.

These international agreements are given effect in Australia at the federal and state (where applicable) level by various legislative instruments (Table D1.2). Due to the lack of sovereignty by Australia over the region designated the Australian Antarctic Territory, the enforcement of some of these instruments only applies to Australian citizens and Australian ships. This definition is not applied equally throughout the policy documents. For example, the *Protection of the Sea (Prevention of Pollution from Ships) Act 1983* explicitly indicates that the spatial coverage includes external territories, including the Australian Antarctic Territory, and that both Australian and international ships that discharge in these waters are subject to penalties under Australian law. However, the *Biosecurity Act 2015* specifies that a foreign vessel operating within the Australian Antarctic Territory territorial waters, or within the 200 nautical mile economic exclusive zone would not be treated as being in Australian waters.

A range of organizations have also developed plans, strategies, and guidelines to help Parties to the Treaty to operate in environmentally safe ways in the Antarctic, though none are legally binding (Table D1.3). Under the Madrid Protocol, the Committee for Environmental Protection was established to help Antarctic Treaty Parties comply with their requirements under the Protocol. In relation to non-native species specifically, they have developed the 'Non-native species manual' as a guide to help Parties prevent non-native species being unintentionally carried into the Antarctic Treaty area and advise on how to manage non-native species found in the Antarctic Treaty area or in transit to the Antarctic Treaty area.

The IMO has adopted the Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species in Resolution MEPC.207(62) in 2011. These guidelines provide general advice to member States on ways to minimize the risk of transferring non-native species as biofouling on ships.

At the national level the Australian Department of Agriculture and the Department of the Environment have co-developed the Anti-fouling and In-water Cleaning Guidelines with the New Zealand Ministry for Primary Industries (Australian Department of the Environment and New Zealand Ministry for Primary Industries, 2015). These guidelines provide advice on the types of in-water cleaning that are available as well as a decision-making tool to determine whether in-water cleaning is appropriate in a given situation.

Other examples of non-binding plans, strategies, and guidelines include: the Scientific Committee on Antarctic Research (SCAR); the Council of Managers of National Antarctic Programs (COMNAP); the International Association of Antarctic Tour Operators (IAATO); and the United Nations, Educational, Scientific and Cultural Organization (UNESCO). SCAR and COMNAP have published handbooks on topics such as reducing supply chain risk of non-native species transfer (Scientific Committee on Antarctic Research and Council of Managers of National Antarctic Programmes, 2019). IAATO have developed visitor guidelines specifically for the tourism industry, both for tourism operators (https://iaato.org/visiting-antarctica/guidance-for-organizers/) and tourists (International Association of Antarctic Tour Operators, 2011). These IAATO guidelines largely summarize the Madrid Protocol in an industry-specific format.

7.5.3 Gap analysis

7.5.3.1 Antarctica

Policy responses for each stage of an invasion process were present in the Australian institutional framework for Antarctica (Table 7.1). However, many were general in nature rather than specific for the threat of MIS. The strongest policy responses came from the *Biosecurity Act 2015* (Cth) with five out of seven policy responses explicitly mentioned in the document. The *Biosecurity Act 2015* (Cth) stated spatial extent includes the Australian Antarctic Territory as an external territory, for the purposes of Australian citizens and ships. However, the Act does not extend to citizens and ships of other countries due to the issues of non-sovereignty of Antarctica and the waters that surround it. There were two policy responses that were not explicitly mentioned in any of the policy documents: restoration of disturbed sites as a prevention response; and monitoring and surveillance of a control response.

7.5.3.2 Macquarie Island

Policy responses for each stage of an invasion process were present in the Australian institutional framework for Macquarie Island (Table 7.2). Similar to Antarctica, the *Biosecurity Act 2015 (Cth)* was the strongest statutory instrument, covering five of the seven potential policy responses. As Macquarie Island is under the jurisdiction of the state of Tasmania, the act applies to citizens and non-citizens alike. There were more policy responses for the prevention and restoration of disturbed sites for Macquarie Island than Antarctica, although none are specifically aimed at MIS. The institutional framework is strengthened by the Macquarie Island Nature Reserve and World Heritage Area Management Plan, which is the only document that explicitly or implicitly mentions all of the policy responses at each stage of the invasion process. There are no national level statutory instruments for monitoring of control measures, though they are present at the state level in the *Biosecurity Act 2019* (Tas) and the aforementioned management plan document.

7.5.3.3 Heard Island

Like Antarctica and Macquarie Island, the *Biosecurity Act 2006* (Cth) was the strongest statutory instrument, and it applies to Australian citizens as well as foreign citizens (Table 7.3). There were no statutory instruments for monitoring of control measures, though they were present in the Heard Island and MacDonald Island Marine Reserve Management Plan 2014 – 2024. This management plan was also the only document which explicitly covered all of the policy responses for each stage of the invasion process, out of all the study sites.

7.5.3.4 Informal plans, strategies, and guidelines

The CEP non-native species manual was the most comprehensive document related to marine non-native species in the Antarctic region, with all seven policy responses mentioned in the document (Table 7.4). Even so, the document was lacking practical guidance for monitoring for marine non-native species, though the CEP five-year work plan has identified non-native species, including marine non-native species as a key area for research. The non-native species manual was the only document to include monitoring and surveillance for control as a policy response. Several documents were applicable to only one stage of the invasion process. For example, the IAATO guidelines for visitors to the Antarctic stress the importance of preventing non-native species introductions, whilst the CEP clean-up manual is concerned with the restoration of degraded Antarctic ecosystems.

Table 7.1 Gap analysis of Australian national policy and formal management plans for Antarctica. Grey indicates there was no mention of the policy response in the policy instrument. Orange indicates there were general comments related to the policy instrument, but not necessarily in relation to invasive species. Black indicates that the policy response appeared in the policy document, but only as it relates to terrestrial or freshwater ecosystems. Purple indicates that the policy response was implicitly mentioned in the policy document for invasive species in general, but the intention is that it would cover marine species. Teal indicates that the policy response was explicitly mentioned in the policy document as it relates to marine species.

		Policy Responses						
		Prevention			**Monitoring**		**Control**	
Policy Level	**Policy Instrument**	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
National	*Antarctic Treaty Act 1960*							
	Antarctic Treaty Regulations 1993							
	Antarctic Treaty (Environment Protection) Act 1980							
	Antarctic Treaty (Environment Protection) (Environmental Impact Assessment) Regulations 1994							
	Antarctic Treaty (Environment Protection) (Waste Management) Regulations 1994							
	Antarctic Marine Living Resources Conservation Act 1981							
	Antarctic Marine Living Resources Conservation Regulations 1994							
	Protection of the Sea (Prevention of Pollution from Ships) Act 1983							
	Protection of the Sea (Prevention of Pollution from Ships) (Orders) Regulations 1994							
	Protection of the Sea (Prevention of Pollution from Ships) Amendment (Polar Code) Act 2017							
	Biosecurity Act 2015							
	Biosecurity Regulations 2016							

		Policy Responses						
		Prevention			Monitoring		Control	
Policy Level	**Policy Instrument**	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
	Protection of the Sea (Harmful Anti-fouling Systems) Act 2006							
Formal management plans	Australian Antarctic Division Environmental Policy 2018 - 2022							

Table 7.2 Gap analysis of Australian national and state policy and formal management plans for Macquarie Island. Grey indicates there was no mention of the policy response in the policy instrument. Orange indicates there were general comments related to the policy instrument, but not necessarily in relation to invasive species. Black indicates that the policy response appeared in the policy document, but only as it relates to terrestrial or freshwater ecosystems. Purple indicates that the policy response was implicitly mentioned in the policy document for invasive species in general, but the intention is that it would cover marine species. Teal indicates that the policy response was explicitly mentioned in the policy document as it relates to marine species.

		Policy Responses						
		Prevention			Monitoring		Control	
Policy Level	**Policy Instrument**	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
National	*Protection of the Sea (Prevention of Pollution from Ships) Act 1983*							
	Protection of the Sea (Prevention of Pollution from Ships) (Orders) Regulations 1994							
	Biosecurity Act 2015							
	Biosecurity Regulations 2016							
	Protection of the Sea (Harmful Anti-fouling Systems) Act 2006							
	Environment Protection and Biodiversity Conservation Act 1999							
	Environment Protection and Biodiversity Conservation Regulations 2000							
State	*National Parks and Reserves Management Act 2002*							
	National Parks and Reserves Management Regulations 2019							
	Nature Conservation Act 2002							
	Threatened Species Protection Act 1995							
	Threatened Species Regulations 2016							
	Biosecurity Act 2019							
	Australian Antarctic Division Environmental Policy 2018 - 2022							

		Policy Responses						
		Prevention			Monitoring		Control	
Policy Level	**Policy Instrument**	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
Formal management plans	Macquarie Island Marine Park Management Plan 2001							
	Macquarie Island Nature Reserve and World Heritage Area Management Plan 2006							
	Macquarie Island Toothfish Fishery Management Plan 2006							

Table 7.3 Gap analysis of Australian national policy and formal management plans for Heard Island. Grey indicates there was no mention of the policy response in the policy instrument. Orange indicates there were general comments related to the policy instrument, but not necessarily in relation to invasive species. Black indicates that the policy response appeared in the policy document, but only as it relates to terrestrial or freshwater ecosystems. Purple indicates that the policy response was implicitly mentioned in the policy document for invasive species in general, but the intention is that it would cover marine species. Teal indicates that the policy response was explicitly mentioned in the policy document as it relates to marine species.

		Policy Responses						
		Prevention			**Monitoring**		**Control**	
Policy Level	**Policy Instrument**	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
National	*Protection of the Sea (Prevention of Pollution from Ships) Act 1983*							
	Protection of the Sea (Prevention of Pollution from Ships) (Orders) Regulations 1994							
	Biosecurity Act 2015							
	Biosecurity Regulations 2016							
	Protection of the Sea (Harmful Anti-fouling Systems) Act 2006							
	Environment Protection and Biodiversity Conservation Act 1999							
	Environment Protection and Biodiversity Conservation Regulations 2000							
	Heard Island and McDonald Islands Act 1953							
Formal management plans	Australian Antarctic Division Environmental Policy 2018 - 2022							
	Heard Island Fisheries Management Plan 2002							
	Heard Island and McDonald Island Marine Reserve Management Plan 2014 - 2024							

Table 7.4 Gap analysis of informal plans, strategies, and guidelines that relate to non-native species management around Antarctica. Grey indicates there was no mention of the policy response in the policy instrument. Orange indicates there were general comments related to the policy instrument, but not necessarily in relation to invasive species. Black indicates that the policy response appeared in the policy document, but only as it relates to terrestrial or freshwater ecosystems. Purple indicates that the policy response was implicitly mentioned in the policy document for invasive species in general, but the intention is that it would cover marine species. Teal indicates that the policy response was explicitly mentioned in the policy document as it relates to marine species. * The CEP five-year work plan has identified the highlighted policy responses for inclusion in future work.

	Policy Responses						
	Prevention			Monitoring		Control	
Informal plans, strategies, and guidelines	**Restoration of disturbed sites**	**Biosecurity**	**Antifouling and ballast water management**	**Monitoring and surveillance**	**Biosecurity**	**Management action plans**	**Monitoring and surveillance**
IAATO Guidelines for Visitors to the Antarctic	Teal	Black	Grey	Grey	Grey	Grey	Grey
IAATO Guidance for those Organising and Conducting Tourism and Non-Governmental Activities in the Antarctic	Orange	Grey	Grey	Orange	Grey	Grey	Grey
IAATO Yachting guidelines for Antarctic cruises	Grey	Teal	Teal	Grey	Grey	Grey	Grey
UNESCO Southern Ocean Action Plan 2021 - 2030	Orange	Purple	Grey	Purple	Purple	Purple	Grey
Anti-fouling and In-water Cleaning Guidelines	Grey	Teal	Teal	Grey	Teal	Grey	Grey
COMNAP Inter-continental checklists for supply chain managers for the reduction in risk of transfer of non-native species	Orange	Teal	Orange	Grey	Grey	Grey	Grey
CEP Non-native species manual	Purple	Purple	Teal	Orange	Orange	Purple	Purple
CEP Clean-up manual	Orange	Grey	Grey	Grey	Grey	Grey	Grey
CEP Practical guidelines for ballast water exchange in the Antarctic Treaty Area	Grey	Grey	Teal	Grey	Grey	Grey	Grey
CEP Five-year work plan *	Orange	Orange	Grey	Purple	Grey	Purple	Grey
IMO Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species	Grey	Teal	Teal	Grey	Teal	Teal	Grey

7.6 Discussion

Marine non-native species are an emerging threat to the Antarctic, but all of the identified potential policy responses appear in the current Australian institutional framework in at least a general sense. However, most of those responses are vague in application, such as stating that a monitoring plan is needed, rather than prescribing what a monitoring plan would look like. For all three locations, as well as in the informal documents, prevention of non-native species appeared in the highest proportion of documents when compared to monitoring or control. This is not unexpected, given that the field of MIS in the broader Antarctic region has only recently gained significant traction (This thesis; Committee for Environmental Protection, 2019; Hughes et al., 2020; McCarthy et al., 2019). Only one statutory document, the *Biosecurity Act 2015 (Cth)*, had responses in all of the invasion stages, though it was not concerned with restoration of sites as a means of increasing ecosystem resilience to invasions, nor the ongoing monitoring after a control response has been used to ensure its success or otherwise. Conversely, the formal management plans for both Macquarie Island and Heard Island both either implicitly or explicitly capture all of the potential policy responses for the threat of non-native species. Similarly, the CEP non-native species manual also captures all of the potential policy responses, though advice on monitoring for non-native species was lacking. However, given the manual is described as a living document and the intention of the CEP is to focus attention on non-native species (Hughes et al., 2018), including marine non-native species, it is likely that this document will be updated as that research is done.

Marine invasions are notoriously difficult, or impossible, to manage once they occur (Thresher and Kuris, 2004). However, around Antarctica and the Southern Ocean, the number of pathways for MIS to enter the region is lower than for terrestrial pathways. There are also no deliberate introductions of MIS occurring in the Antarctic, compared to elsewhere in the world where vectors can include deliberate introduction for mariculture and the aquarium trade (Bax et al., 2003). Introductions of MIS to Antarctica and the Southern Ocean are likely to occur as fouling on ships or equipment, meaning that the pathway is well defined (Lewis et al., 2005; Molnar et al., 2008). This means that prevention measures that specifically target fouling as a vector for MIS transfer will be the most effective way to keep the Southern Ocean MIS-free (Crall et al., 2006). Currently, prevention is the best represented of the policy responses in the current institutional framework, with explicit responses to marine biosecurity, and antifouling and ballast water management. The prevention and restoration of disturbed sites is less well-represented; however, this is because the policy responses are aimed with restoration to protect native species, rather than being aimed at preventing MIS from becoming established. Regardless, habitat restoration for any reason will contribute towards the goal of preventing MIS from becoming established in this region (Bergstrom, 2022). There are certain species, such as *Undaria pinnatifida* (Chapter 4), which will struggle to establish in undisturbed sites, so preventing the disturbance from occurring in Antarctic and subantarctic regions will help reduce the pool of potential MIS that could invade these areas (James and Shears, 2016b; Valentine and Johnson, 2003).

Australia has recently deployed its new icebreaker ship, *RSV Nuyina*. As part of Australia's commitment to the Madrid Protocol, the vessel underwent an Initial Environmental Evaluation which included a risk assessment for operation (Australian Antarctic Division, 2021b). Transfer of terrestrial and marine non-native species was highlighted as a risk of operation with a number of measures identified to limit the risk of non-native species transfer. These include pre-season hull inspections for fouling species, the implementation of a biofouling management plan with stipulations for monitoring, as well as practical measures to prevent entrainment of non-native species onto the vessel. Even though non-native species have been identified as a threat under this risk assessment, there are currently no monitoring plans for marine non-native species in the environments around the research bases or subantarctic islands. This is despite there being provisions for monitoring for non-native species under the *Biosecurity Act 2015 (Cth)*, as well as the formal management plans for each of the islands. So, while the institutional framework appears to cover all of the potential policy responses, the reality is that these are not necessarily implemented currently.

Historically there have been actions taken in conflict with the regulatory instruments of the time, including late arriving cargo evading quarantine or disposal of waste on Macquarie Island (Potter, 2007). This was attributed to the fact that the Australian Antarctic Division plays the dual role of operator and regulator and that the non-committal language of various policy instruments (i.e. may take an action versus will take an action) means that operational functions could take a higher priority than environmental concerns (Kriwoken et al., 1989). Since this time though, the Australian Antarctic Division has taken considerable steps to clean up legacy waste (Snape et al., 2001; Stark et al., 2006), to manage non-native species around the Antarctic bases and on the subantarctic islands (Shaw et al., 2011), and to enhance its biosecurity measures to a high standard which are recommended for emulation by other nations (Tsujimoto and Imura, 2013).

Whilst there is paucity of Australian statutory instruments which cover the specific threat of MIS to the Antarctic and subantarctic, there is an appreciation for the threat at the operational level. In 2002, a barge set for deployment at Macquarie Island was found to have a significant fouling community of at least 20 different species, including algaes, crustaceans, and molluscs (Lewis et al., 2006). Though there was an attempt to remove this fouling, ultimately a management decision was made to stop the deployment of the barge into the waters of Macquarie Island. Although the outcome of this instance was favorable, the fact is that most fouling will not be easily visible from onboard a vessel and will be largely inaccessible to operators on journeys already underway. This is a key factor that sets marine invasions apart from terrestrial invasions.

A compounding issue of implementation and compliance is the lack of statutory authority over the citizens and ships of foreign nations in Antarctica (but not Macquarie Island and Heard Island). Although Australia has a territorial claim over the Australian Antarctic Territory, there are no sovereign rights conferred and the claim is not recognized under the Antarctic Treaty. Therefore, while policy instruments like the *Protection of the Sea (Prevention of Pollution from Ships) Act* contain requirements and penalties for all ships in the Australian Antarctic Territory exclusive economic zone, the reality is that most nations do not recognize this area as being under the control of Australia. Therefore, enforcement of Australian statutory instruments to non-Australian citizens and ships in this region is unlikely and could be considered by the international community as a breach of international law, as was recently seen with Australia’s attempt to enforce a directive to stop whaling to the Japanese under the *EPBC Act* (Anton, 2008). This then sets a precedence going forward where even those countries which recognize the Australian territorial claim cannot be unfairly targeted by compliance with Australian legislation.

Without monitoring and surveillance of the marine environment near the Australian Antarctic research bases or subantarctic islands, the risk of negative impacts to these ecosystems increases from the moderate rating seen in the *RSV Nuyina* IEE to potentially catastrophic (Chapter 6). Identification of introduction and establishment events of non-native species to these regions will be left to chance, and oftentimes this only occurs once a species has increased to a sizeable population (Crooks, 2005). Early detection is crucial to ensure that eradication is a feasible solution, such as with the black-striped mussel, *Mytilopsis sallei,* from Darwin, Australia (Giakoumi et al., 2019). The connectivity within the ocean, as well as the harsh environment of the Antarctic and subantarctic regions, means that eradication attempts would be incredibly difficult even in the case of early detection. This is particularly true for species with high dispersion capacity (Giakoumi et al., 2019), like *Asterias amurensis* (Chapter 4).

A key policy response that was deficient in the institutional framework was biosecurity monitoring of pathways for non-native species. Typically, biosecurity would happen before departure and then the recipient location would be monitored for non-native species introductions and establishment. However, the internal systems of a ship can pose a significant risk of non-native species transfer. Recent work has shown that internal seawater systems of polar ships, as opposed to external recesses like sea chests, can harbour significant, and reproductive, populations of many taxa of non-native species (Mccarthy, 2021). Further, the water from these internal pipes is often discharged overboard. Strategies for monitoring for non-native species on vessels should include methods to test for non-native species within the internal frameworks of the vessel whilst underway. That way, preventative measures can be implemented *en route* to stop non-native propagules being released into coastal and shallow water environments.

There is a clear ambition at the international, national, and operator level to protect the Antarctic and subantarctic marine environments from invasive species. Policy responses for all stages of an invasion are in place in the Australian institutional framework, though more could be done to further enhance and standardize policy responses and on-the-ground non-native species management. Collaboration between Antarctic and subantarctic scientists and policymakers, and invasion scientists from other Australian government branches, such as the Department of Agriculture, Water, and the Environment, should be encouraged to facilitate the development of best practice strategies for managing the non-native species threat to Antarctica and the subantarctic. If we are able to robustly and strategically manage for non-native species now, we may be able to avoid a future where native Antarctic and subantarctic coastal marine ecosystems are being impacted by marine invasive species.

Chapter 8: Thesis discussion

8.1 Summary

In this thesis I have used available data and a range of modelling tools to produce an overall risk assessment of the threat of MIS to the nearshore environments located near Australia's Antarctic bases and subantarctic islands. Further, I have explored Australia's current policy responses to the threat of MIS in the Southern Ocean. To our knowledge, there are no invasive species that have successfully established in the Southern Ocean. This gives us an opportunity to anticipate, and ideally prevent, marine invasions from occurring. Throughout this PhD program the issue of marine invasions in the Southern Ocean has grown from an addendum of the terrestrial invasive risk into an area receiving concerted attention with an increased awareness of the threat that native Southern Ocean marine ecosystems face in a changing world.

8.2 Marine invasive species in a changing world

Introductions of several MIS have already occurred in the Southern Ocean, and this is likely to continue into the future as shipping traffic in the region continues to increase (McCarthy et al., 2022). However, the Australian Antarctic research bases located in East Antarctica receive significantly less shipping traffic than the West Antarctic Peninsula region, where the first marine invasions are predicted to occur. Heard Island receives even less, with ships visiting maybe once every several years. Macquarie Island, on the other hand, receives the highest number of ship visits of the Australian sites and is often visited by tourist cruises with itineraries out of New Zealand.

A lack of fouling community data from Southern Ocean bound vessels limits our ability to predict which species may be introduced to the region. None of the species that were predicted to survive the Antarctic and subantarctic conditions near the study sites have been found on Southern Ocean bound ships so far, however that is more likely representative of a lack of studies, rather than a true indication that the species are not being carried on the outer surfaces of ships.

In Chapter 3 I explored the potential of MIS to spread around the Southern Ocean as planktonic larvae. The sparse arrangement of shallow coastal environments around the Antarctic continent (Southwell et al., 2021) works to protect the nearshore environments of the Australian Antarctic bases from passive introduction of non-native species from other regions of the Antarctic or beyond in biologically relevant time periods. Only spread from nearby stations was predicted for Davis and Mawson Stations, from the nearshore shallow coastal habitats of the Prydz Bay region. For that to occur, however, non-native species would need to have been introduced by anthropogenic means to shallow coastal environments within Prydz Bay. The subantarctic islands were predicted to be at a slightly higher risk of passive introductions. If non-native species were introduced to the northern Kerguelen Plateau, then there is the potential for spread southward to Heard Island. Macquarie Island's closest shallow water connection is Heard Island, which is a three-month journey in the plankton. This would limit the number of species that could be carried in the plankton to Macquarie Island to those with long pelagic larval durations in excess of three months, such as *Asterias amurensis*.

Overall, Macquarie Island represents the highest risk area for the introduction of MIS, while Heard Island represents the lowest risk. The Antarctic bases are at an intermediate level of risk as they receive more ships than Heard Island.

8.3 Establishment risk

The majority of non-native species would not be able to establish in the Antarctic marine environment due to the harsh conditions, such as seasonality of primary productivity and freezing ocean temperatures (Barnes et al., 2006; McCarthy et al., 2019). Only one species was consistently modelled in Chapter 4 as being able to survive in all five locations currently and in the future under two climate change scenarios: *Asterias amurensis*. Given our currently understanding of the thermal tolerance of the larvae of this sea star, it is unlikely that it would currently be able to successfully establish in the Antarctic . However, this is a species with a polar distribution in the Arctic, indicating that it has the ability to adapt to freezing conditions. This barrier to larval development though, does not extend into the subantarctic. Species distribution modelling of larval thermal tolerance has shown that conditions around Macquarie Island could support larval development (Byrne et al., 2016).

Overall, the risk of establishment is unlikely at the Antarctic sites or Heard Island due to the colder ocean temperatures experienced in those regions. Macquarie Island is the most likely place that establishment of non-native species would occur.

8.4 Impacts to native marine ecosystems

Predicting impacts of invasive species can be difficult, particularly when there is no information on impacts in similar environments. The lack of native community data and the computation cost of running invasion simulations means there is much uncertainty in the models. In Chapter 5 I used the ensemble ecosystem modelling (EEM) system to make predictions about a range of possible plausible future scenarios following the introduction, and successful establishment of a set of high-risk MIS into an Antarctic coastal marine environment. In most scenarios, the risk was negligible for native species; however there were simulations that resulted in significant native species losses. Given the precautionary principle, I classified the potential impact of species as catastrophic.

There is a lack of community data from the subantarctic islands, so impacts could not be modelled for Macquarie Island or Heard Island.

8.5 Marine invasive species risk to East Antarctic coastal environments

The Australian Antarctic marine environments are at a high risk of marine invasions. This is largely driven by the potential impacts if species were able to establish. This finding indicates that steps should be taken to mitigate the risk. The best way to mitigate the potential impacts of invasive species is to prevent them from reaching and establishing in the region. Further, it should bring attention to the need to develop monitoring and surveillance methodologies to identify any non-native species as early in the invasion process as possible, ideally before being transported to the Antarctic.

8.6 Marine invasive species risk to Macquarie Island

Macquarie Island is difficult to assess due to the lack of impact data. However, if we use the precautionary principle as a guide, we should hope for the best but prepare for the worst. In this case, with an almost certain risk of anthropogenic introduction and likely establishment risk, the highest category of overall risk is severe. As with the Antarctic sites, this should bring attention to the need to develop monitoring and surveillance methodologies to investigate whether there are any non-native species already in the region, and to ensure that there are sufficient policy responses in place to prevent introductions from occurring. Hull fouling community data should be a priority area of research, with a focus on ships that are travelling directly to Macquarie Islands and do not undergo periods of sea ice traversal.

8.7 Marine invasive species risk to Heard Island

Heard Island is difficult to assess due to the lack of impact data. However, as with Macquarie Island, the precautionary principle should be applied when assessing this risk. In this case, with an unlikely risk of introduction or establishment, the highest risk rating possible is high. Due to the lack of ship visits to the island from the Australian Antarctic Division, it would be difficult to implement a robust monitoring program in this region. However, establishing baseline data of shallow water ecosystems should be a priority so that changes to the ecosystem can be monitored over longer time periods. Imperfect data is better than no data.

8.8 How to make decisions with imperfect prediction data?

Imperfect data or conflicting results should not prohibit the initiation of management action in terms of the threat of MIS around Antarctica. The test of whether to implement a management action based on imperfect data should be weighed against the potential cost if the management action is not undertaken (Bergstrom, 2022).

The cost to undertake monitoring missions to the Antarctic and subantarctic is considerable. Added to this is the potential cost of a monitoring program that may be selected for use, whether it is diver surveys, autonomous underwater vehicles, or genetic techniques. The gamble is that all of those resources are used but no non-native species are ever detected. Conversely, if we fail to implement a monitoring program and a non-native species establishes it could have a negative impact on native species and ecosystems. In this case there will be a considerable lag time between the introduction of the species and its detection. In that time the invasive species population would have grown to a significant size. At this point the trade-off would likely be between investing significant resources to try to remove or manage the species, which may or may not succeed, and may need to continue indefinitely, and doing nothing. The potential cost of not implementing a monitoring program is far more than spending the resources upfront to prevent introductions and establishment of invasive species.

8.9 Are Australian policies ready for Southern Ocean marine invasions?

There is a clear intention by Australia, and the international community to prevent marine invasions from occurring in the Antarctic and subantarctic. Potential policy responses to the various stages of an invasion event are all captured, at least in part, by the current Australian institutional framework. Currently lacking though, are prescriptive instruments which outline how each policy response should be undertaken. The threat of MIS has been neglected in policies that deal with environmental protection or terrestrial invasions, though appreciation of the risk has increased in recent years. Development of MIS-specific guidelines and policy should be a priority if Australia wishes to maintain its position as a global leader in the protection of the Antarctic and subantarctic environment.

8.10 Are there marine invasive species in the Southern Ocean hiding in plain sight?

Although there are several examples of non-native species being observed in the Southern Ocean, there are other non-native species that I found in the species occurrence databases that have never been further examined in the literature, except in one paper, for one of the species. Three species which were identified in occurrence databases are known to be invasive. A record of the polychaete *Alitta succinea* at Macquarie Island was located within the biodiversity.aq database (found during research for chapter 5 of this thesis). That record was part of a historical record from the 1911–1914 Australasian Antarctic Expedition and is the only record of the species in that region. A record of the ribbed mussel *Geukensia demissa* at King George Island (Antarctic Peninsula region) was located within the OBIS database (found during research for chapter 3 of this thesis). Finally, multiple records of the polychaete *Chaetopterus variopedatus* were found in the Global Biodiversity Information Facility database covering regions such as the subantarctic, the Antarctic Peninsula region, and the Ross Sea region. The identification of *C. variopedatus* in the literature was the first record of the species in Admiralty Bay, but it was not identified as a potential invasive species (Pabis and Sicinski, 2010), even though later work has flagged its potential as an invasive species in the Antarctic Peninsula region (Hughes et al., 2020).

8.11 Strengths

MIS are notoriously difficult to study. The openness of the ocean and our limited ability to access and research in these environments are compounded in the polar regions. Extremely remote locations like Heard Island may only be visited once every few years, and even then, there may be no marine work undertaken as part of that visit. Therefore, we need tools that can complement *in-situ* research to make good environmental decisions. The purpose of my thesis was to provide the first overall risk assessment of the MIS threat to nearshore marine ecosystems near Australia's Antarctic research bases and subantarctic islands. Of key importance was that this risk assessment could be completed at a desktop, given the difficultly in accessing the Antarctic fieldwork.

In Chapter 3 I examined the potential for planktonic species to be carried around the Antarctic and subantarctic (and in some cases, beyond) in order to understand the risk of non-native species being carried to the five study sites. It is widely accepted that non-native marine species are likely to first arrive in the West Antarctic Peninsula region and thus most research on MIS has focused on that region. Here, I provided the first assessment of the risk of MIS travelling as plankton to East Antarctica and Australia's subantarctic islands. This modelling used freely available ocean current, wind, and wave action data to determine the potential source locations of planktonic species arriving at the study sites. As far as I am aware, this is the first study to run simulations backwards in time to determine the source of planktonic species around the study sites. The Python-based software used in this chapter is also freely available and highly customizable enabling access to a range of end-users and is not limited to the Southern Ocean. This tool will also be useful if, but more likely when, a non-native species establishes in the Southern Ocean as it can be readily adapted to include species-specific information for a more precise prediction of non-native species spread around the region.

In Chapter 4 I used a popular machine learning algorithm to predict whether known global invasive species could survive near the five study sites given a range of environmental parameters. Mine was the first study to use this algorithm to predict invasive species survival in the Antarctic and subantarctic region, and only the second to use it for invasive species modelling worldwide. The ability to make accurate predictions with this algorithm means that it is used in areas such as credit card fraud detection. Even with sparse records of invasive species occurrences, I was able to make predictions with very high accuracy using this algorithm for both the presence of MIS as well as the absence of MIS. As with Chapter 3, I was able to make use of freely available data on species occurrences, current environmental conditions, and future environmental conditions. The analysis was done in the program R, another common and freely available program that allows access by a range of end-users. This modelling can be readily applied to other areas of the Antarctic and subantarctic to predict MIS survival in those regions.

In Chapter 5 I modelled a range of plausible future scenarios following the establishment of a MIS. This is the first study to use the ensemble ecosystem modelling technique to make predictions of the range of impacts that may be possible with the establishment of MIS to the Antarctic region. Antarctic benthic ecosystems are unlike other benthic ecosystems of the world, owing to the high level of endemism in the region. This makes it particularly difficult to predict impacts of non-native species in this region. I was able to use the limited data that is available for species interactions in the Antarctic to model a range of plausible future scenarios over a biologically relevant time period, given the long generation time of many native Antarctic species. This chapter also allowed me to see the potential fate of MIS being introduced into an Antarctic ecosystem, though it was not the original intention of the chapter. The algorithm lends itself to be adapted to include a secondary (or beyond) influence on the ecosystem, such as including declines in abundance of native species due to other threats like climate change. As with the previous chapter this modelling was done using the freely available R software. The developer of this method has also provided the code for free use and adaptation for other users.

In Chapter 6 I integrated the results of Chapters 3 through 5 with the existing literature to provide an overall risk assessment of MIS to the nearshore marine environments of Australia's Antarctic research bases and subantarctic islands. This is the first overall MIS risk assessment to all of these areas. My work has highlighted areas for future work to enhance our understanding of the MIS risk. For example, nearshore marine community surveys are needed at Macquarie Island and Heard Island as there is an insufficient understanding of the baseline ecosystems of these subantarctic islands. Further, we need a better understanding of the fouling species that are being carried to the Australian Antarctic and subantarctic sites on the hulls of ships.

Chapter 7 was the first review of Australian policy in respect to MIS in the Southern Ocean. While all components of the invasive species problem were found within the current policy instruments, there are still deficiencies when it comes to specific MIS responses. This information can help to strengthen the current policy landscape before MIS become a problem near the Australian study sites.

Whilst there has been a paucity of research on MIS in the Antarctic region, particularly in East Antarctica, I have shown that it is possible to create a robust, and repeatable, risk assessment using freely available software and data. This research has highlighted where the current knowledge gaps are, and how future work in this space can enhance the overall assessment of risk for the MIS threat to the Antarctic region.

8.12 Limitations and suggestions for future work

Uncertainty is the biggest limitation of using models to make predictions. All three of the risk assessment content chapters (Chapters 3 – 5) relied on data that is likely to be imperfect. Ocean current data is often interpolated from a range of data sources. Occurrence records of MIS are almost certainly missing from less surveyed regions. Future climate predictions could take a number of trajectories. Antarctic food web links are often determined by using surrogate species from other regions of the world. Invasive species' interactions with Antarctic species may be completely different than those recorded in other regions of the world. A number of plausible futures are possible with current modelling techniques. Every step of this process has had some degree of uncertainty.

Another key limitation to these types of analyses is the need for very large data sets. Some data sets are available to be accessed online during calculations, such as ocean current data; however others need to be downloaded and stored by the user. Further, the outputs of many models, like the particle tracking models, are very large data files, requiring significant storage capacity. To run the models in a reasonable time frame, access to high power computing was required. I appreciate that I have been incredibly privileged to have access to these resources, and that not everyone will have this same opportunity.

Specifically related to the ensemble ecosystem modelling, a more fundamental limitation is that in order to process the simulations in a reasonable time frame the food web had to be condensed into a smaller number of species groups, from the original 41 species of the survey data from the Windmill Islands near Casey Station. This makes it difficult to capture species-specific impacts and may hide some interesting interactions that are not observed in the reduced model.

Future work with particle tracking should look at ways of obtaining Stokes drift data via the OPeNDAP (Open-source Project for a Network Data Access Protocol) or THREDDS (Thematic Real-time Environmental Distributed Data Services) frameworks, as currently it is only available as a large downloadable netcdf file from an FTP server. To improve the accuracy of the particle tracking models, future work should look to include species-specific data such as larval durations and behaviours. In Chapter 3 I wanted to capture a general overview of the threat to each of the Australian Antarctic and subantarctic locations, rather than focus on any one specific species. Also, combining particle tracking data with environmental variables such as temperature would improve overall model predictions of the spread of invasive species.

Machine learning has shown real promise as a tool for making predictions of species distributions, and indeed my models showed high levels of accuracy for predicting species distributions from a training dataset. Future work in this space would benefit from the incorporation of the new IPCC CMIP6 datasets which have been released recently. These new datasets have increased in their resolution, better encapsulate sea ice dynamics, and include a number of new emissions scenarios which can enable modelling across a wide range of predicted futures. Future work using machine learning should also increase the spatial coverage of the predictions, rather than limiting them to a subset of sites.

Addressing a key gap in the knowledge – the lack of fouling community analyses – will help guide MIS management in the Antarctic space. Although very few of these analyses have been conducted on a wide range of species were found. Of particular concern for the Antarctic region are niche areas of the submerged surface of ships, like sea chests and other water intakes. Settlement plates are a common tool to assess biofouling communities (Gartner et al., 2016; Marraffini et al., 2017). This would involve placing magnet-backed tiles on areas of a ship's hull, with a particular focus on niche areas, or those areas which are more susceptible to biofouling, such as sea chests, propeller shafts, inlet gratings, etc.

New technologies will enable easier monitoring and surveillance for MIS in the Antarctic region. Detecting newly introduced species can be incredibly difficult, especially in remote marine environments. Often the best opportunity to control an invasion event is long gone by the time the invasive species is detected in the environment (Invasive Species Council, 2017). Traditional methods of detection, such as physical sampling and taxonomic identification, are often laborious, time-consuming, expensive, and ultimately inefficient (Bott et al., 2010). However, new molecular techniques now give us the opportunity to detect species at very early stages of introduction. The use of environmental DNA (eDNA) allows for the detection of a specific species of interest from a filtered sample of water, by capturing the remnants of DNA that have been excreted or shed by the species into its environment (Bott et al., 2010; Pilliod et al., 2013).

8.13 Conclusion

In this thesis I aimed to develop an understanding of the risk posed by MIS to Australia's Antarctic and subantarctic sites. Polar fieldwork is particularly difficult to undertake and is very limited in the number of participants that head south, which has been compounded by the global pandemic. The tools I have repurposed for an Antarctic context and the data used in these studies are all free to the public; however computational requirements currently limit the ability for equitable uptakes by all stakeholders.

I have shown that the risk of MIS is site specific, and even those sites with similar latitudinal locations vary in their risk profile. Perhaps most pertinent is that I have shown that there is a real possibility that MIS have the capacity to cause extinctions of native Antarctic species. The species of the Antarctic will be under increasing stress from climate change, and the incursion of an invasive species could have a synergistic negative effect on ecosystem functioning in this region.

After the complete failure of the 2021 United Nations Climate Change Conference (commonly referred to as COP26) summit to create meaningful plans for the management of climate change, we must prepare for the fact that the world is changing, and that change will create a myriad of new and complex environmental and sociological issues that extend around the globe.

The Antarctic is one of the last places on the planet that we can plan ahead to avoid and manage at least one of the threats arising from climate change; that of marine invasive species.

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Appendices

Appendix A: Supplementary Materials for Chapter 4

This appendix contains the supplementary material provided with the publication in *Diversity and Distributions* that makes Chapter 4 of this thesis.

Further, the data that was used in this study is available on the Dryad database and can be accessed at: https://doi.org/10.5061/dryad.3ffbg79hf .

A.1 Australian Antarctic environmental change

A.1.1 Mawson: RCP 4.5 & RCP 8.5

Average **SST** is expected to rise < 1˚C under both RCP scenarios (Fig A1.1). Changes in temperature are predicted to be consistent for all seasons, however under RCP 8.5 summer temperatures are predicted to increase faster towards the end of the century. **SSS** is expected to decrease under both RCP scenarios with greater decreases expected in summer and autumn under RCP 4.5, and in autumn for RCP 8.5. Similarly, **pH** will drop from over 8.0 currently to close to 7.9 by the end of the century. This change is expected to be consistent over all seasons under both RCP scenarios. **NO_3** is first expected to rise slightly, and then sharply decrease in the last quarter of the century under both RCP scenarios. This decrease will be more pronounced during winter and spring under both climate change scenarios.

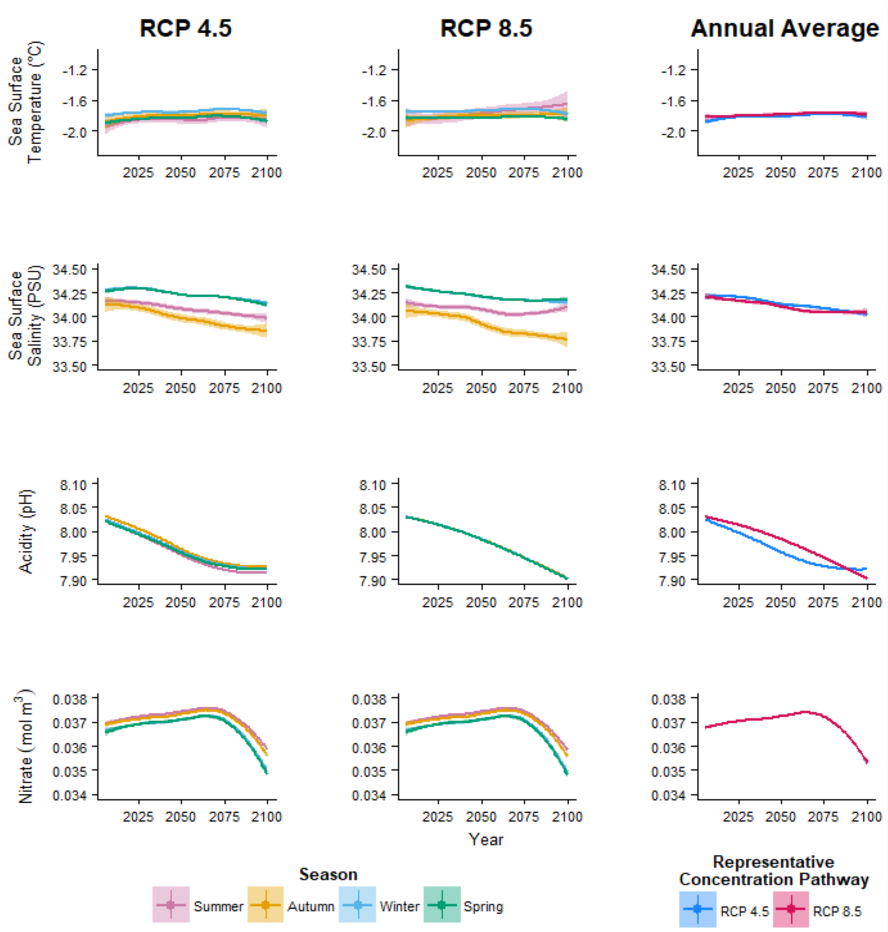

Figure A1.1 Mawson Station CanESM2 CMIP5 projections for seasonally averaged and annually averaged (RCP 4.5 and RCP 8.5) environmental variables in shallow marine habitats adjacent to Australia's Mawson continental research station.

A.1.2 Davis: RCP 4.5 & RCP 8.5

Average **SST** increase of < 1˚C is expected under both RCP scenarios, with summer and autumn temperatures driving the overall increase, particularly under RCP 8.5 (Figure A1.2). **SSS** is predicted to decrease under both RCP scenarios, with a greater decrease occurring under RCP 8.5. The decrease in SSS is expected to occur consistently for all seasons under both RCP scenarios. **pH** is also expected to decrease under both RCP scenarios, with a greater decrease occurring under RCP 8.5. **NO_3** will also decrease overall under both RCP scenarios, with a greater decrease occurring under RCP 8.5. These decreases in NO_3 are expected to occur consistently for all seasons under both RCP scenarios.

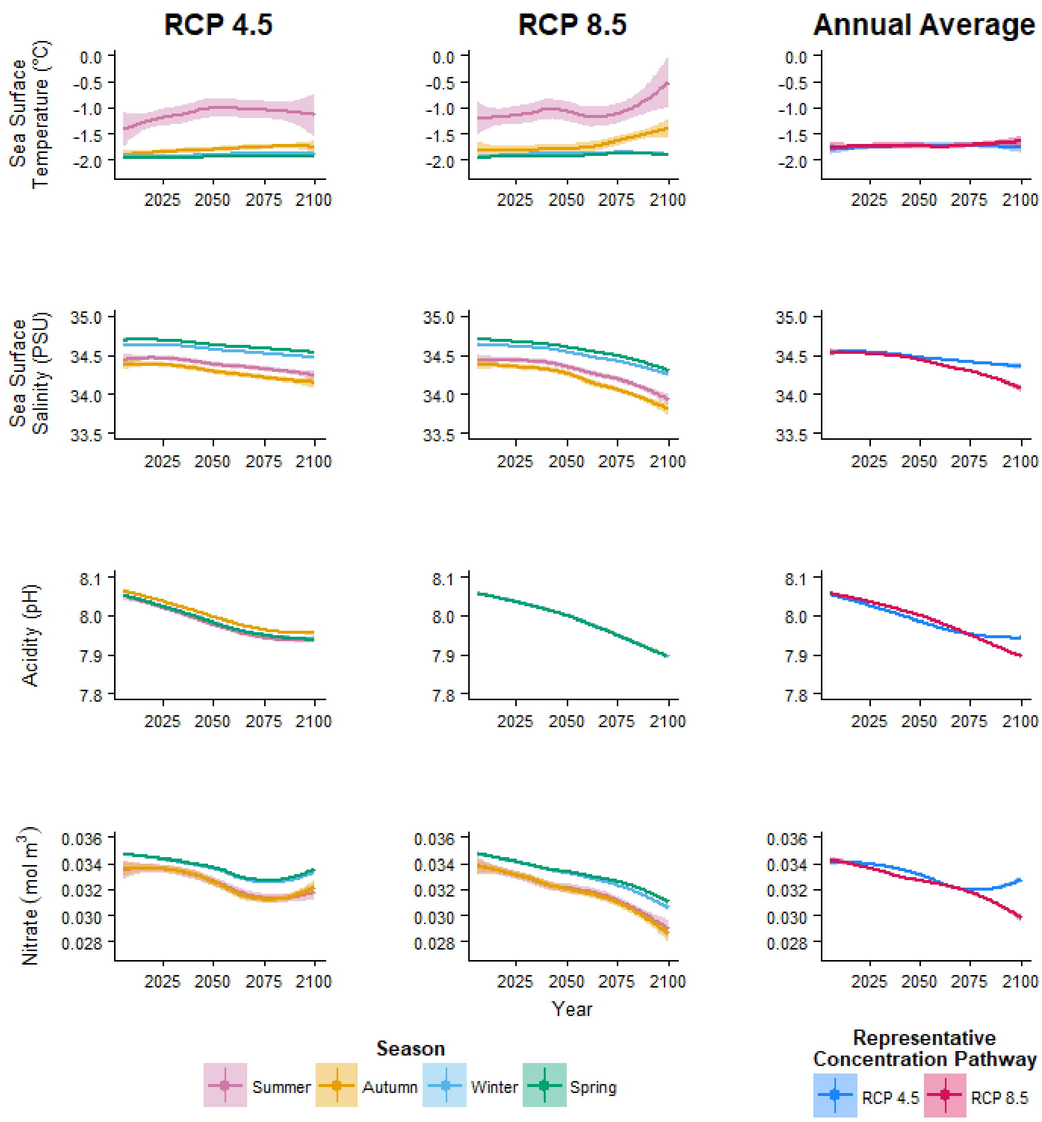

Figure A1.2 Davis Station CanESM2 CMIP5 projections for seasonally averaged (RCP 4.5 and RCP 8.5) and annually averaged environmental variables in shallow marine habitats adjacent to Australia's Davis continental research station.

A.1.3 Casey: RCP 4.5 & RCP 8.5

Average **SST** is expected to rise < 0.5˚C under both RCP scenarios, with winter and spring temperatures remaining stable through to the end of the century (Figure A1.3). More fluctuation is seen in summer and autumn temperatures, with summer SST under RCP 8.5 rising sharply in the latter half of the century. **SSS** is expected to decrease under both RCP scenarios, with a greater decrease under RCP 8.5. Decreases in SSS are predicted to occur consistently throughout all seasons under both scenarios. **pH** will decrease overall under both RCP scenarios; however, the pattern of decrease differs between the scenarios. There will be a consistent decrease in pH under RCP 4.5 through to the end of the century, however under RCP 8.5 the pH will decrease to a minimum around 2060, before rising again to near current pH. **NO_3** will decrease under both RCP scenarios, with greatest decreases occurring in autumn for both scenarios.

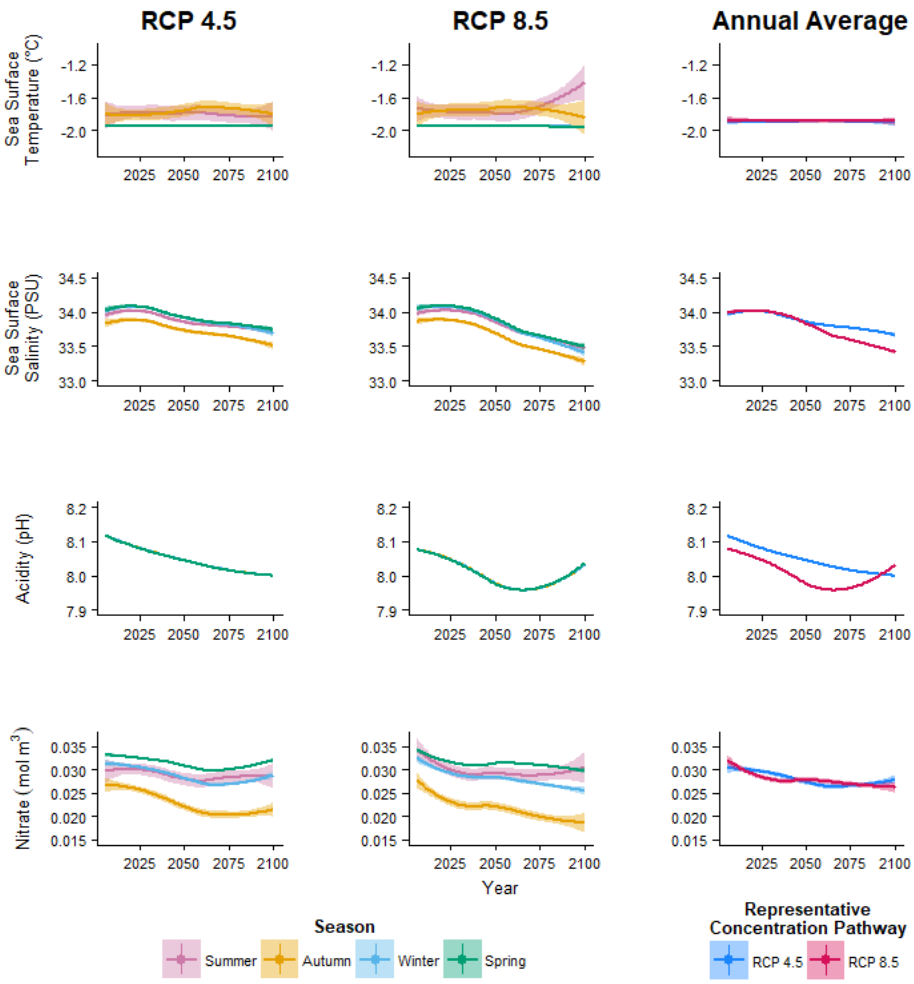

Figure A1.3 Casey Station CanESM2 CMIP5 projections for seasonally averaged (RCP 4.5 and RCP 8.5) and annually averaged environmental variables in shallow marine habitats adjacent to Australia’s Casey continental research station.

A.1.4 Macquarie Island: RCP 4.5 & RCP 8.5

Average **SST** is expected to rise approximately 1˚C under RCP 4.5 and 2˚C under RCP 8.5 (Figure A1.4). Unlike the continental stations, these increases in SST will occur consistently throughout the seasons. Both RCP scenarios experience similar rates of increase through the first half of the century, before RCP 8.5 SST sharply increases in the second half of the century. **SSS** is expected to decrease under both RCP scenarios, with fairly consistent decreases occurring in all seasons. **pH** is also expected to decrease under both RCP scenarios. **NO_3** is expected to decrease under both RCP scenarios, however only after initial increases in the first half the century. This decrease will plateau under RCP 4.5 but will continue to decline sharply under RCP 8.5. The NO_3 decrease in winter under RCP 4.5, and in autumn and winter under RCP 8.5 drive these overall decreases.

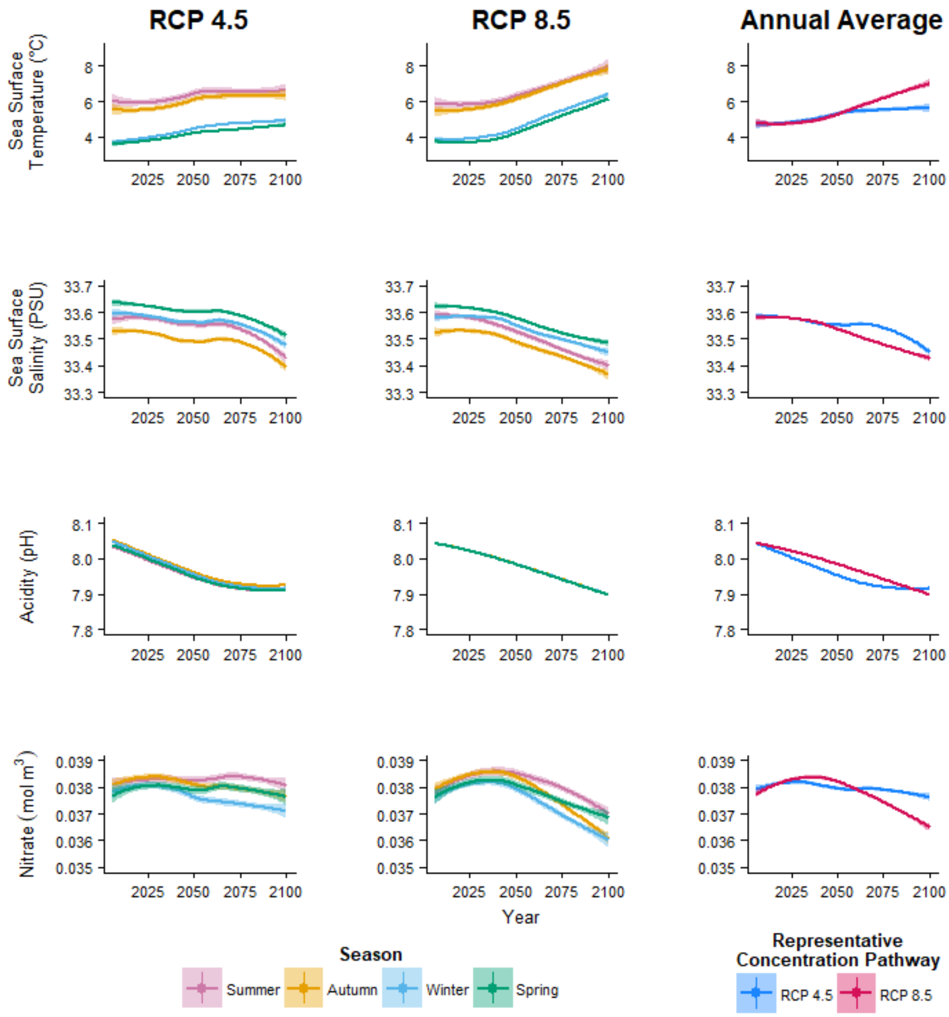

Figure A1.4 Macquarie Island CanESM2 CMIP5 projections for seasonally averaged (RCP 4.5 and RCP 8.5) and annually averaged environmental variables in shallow marine habitats adjacent to subantarctic Macquarie Island.

A.1.5 Heard and McDonald Islands: RCP 4.5 & RCP 8.5

Average **SST** is expected to rise approximately 2˚C under RCP 4.5 and 4˚C under RCP8.5 (Figure A1.5). The increases in SST under both scenarios are consistent under both scenarios for the first half of the century, before RCP 8.5 increases sharply during the latter half of the century. The increases in SST will be fairly consistent across seasons, however under RCP 8.5, spring SST is expected to increase more sharply than the other seasons. **SSS** is expected to decrease under both RPC scenarios, however under RCP 8.5 SSS will decrease to a minimum around 2060 before rising to near-current SSS by the end of the century. **pH** is also expected to decrease under both RCP scenarios, with a greater overall decrease seen in RCP 8.5 by the end of the century. The decrease is pH is consistent between seasons under both RCP scenarios. **NO_3** is expected to remain fairly stable under both scenarios, with only a slight decrease under RCP 4.5, and a slight increase under RCP 8.5 by the end of the century. Changes in NO_3 are consistent between seasons under both RCP scenarios.

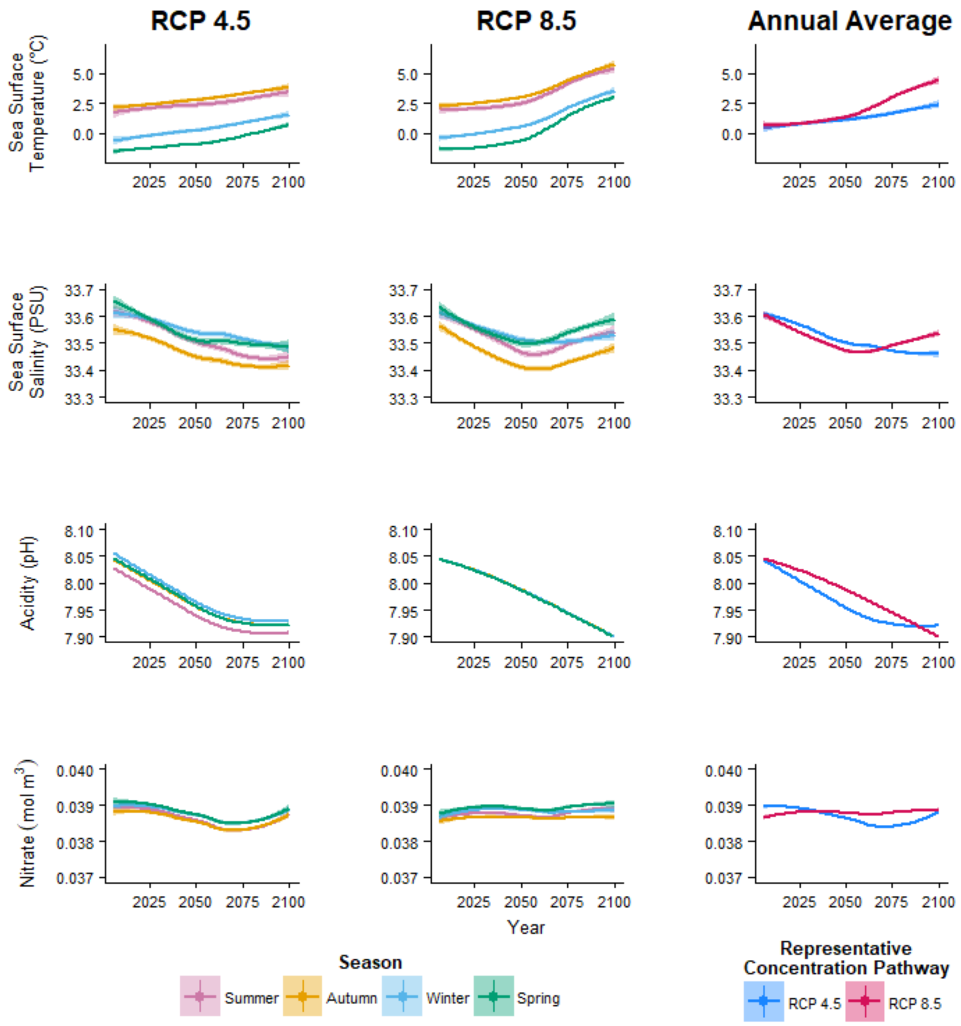

Figure A1.5 Heard and McDonald Island CanESM2 CMIP5 projections for seasonally averaged (RCP 4.5 and RCP 8.5) and annually averaged environmental variables in shallow marine habitats adjacent to subantarctic Heard and McDonald Islands.

A.1.6 Model performance by species

Table A1.6a This table provides details of the Annual Model performance for each species by the type of variable aggregation. The resampling technique which created the best model is identified, as is the most important variable to the model.

	SST Only				SST, SSS, NO3				SST, SSS, NO3, PH			
	Resamp.	Predict Absence	Predict Presence	Most imp var	Resamp.	Predict Absence	Predict Presence	Most imp var	Resamp.	Predict Absence	Predict Presence	Most imp var
Alitta succinea	Over	95.55	86.83	Avg SST	Over	96.56	88.29	Avg SST	Over	0.9686	0.9138	Min SST
Ascidiella aspersa	Over	94.09	94.83	Max SST	Over	96.7	95.98	Avg SST	Both	0.9504	0.9185	Max SST
Asterias amurensis	Under	86.88	92.86	Avg SST	Under	88.62	95.24	Min SST	Under	0.8855	0.9778	Max NO3
Bugula neritina	Over	94.56	81.54	Avg SST	Both	93.78	90.77	Max SST	Under	0.8743	0.8905	Max pH
Carcinus maenas	Under	92.62	93.84	Avg SST	Under	94.07	95.73	Avg SST	Over	0.9769	0.9425	Avg SST
Charybdis japonica	Over	99.89	82.35	Avg SST	Under	93.33	100	Avg SST	Under	0.8633	0.8889	Max pH
Ciona intestinalis	Over	96.03	91.41	Avg SST	Over	96.45	96.46	Avg SST	Under	0.941	0.9147	Avg SST
Codium fragile	Over	95.16	94.79	Min SST	Over	96.15	91.67	Min SST	Under	0.9131	0.951	Avg SST
Crassostrea gigas	Over	96.12	88.4	Min SST	Both	94.73	90.06	Max SSS	Under	0.9287	0.9795	Max SST
Crepidula fornicata	Both	94.05	96.65	Max SST	Both	95.85	96.09	Avg SST	Both	0.9586	0.9246	Max SST
Didemnum vexillum	Over	97.29	96.15	Avg SST	Both	98.12	98.08	Avg SSS	Over	0.9825	0.8871	Max NO3
Elminius modestus	Over	98.14	84.04	Avg SST	Under	95.54	89.36	Min SST	Both	0.9759	0.9394	Max SST
Ficopomatus enigmaticus	Both	95.61	83.05	Avg SST	Both	97.39	86.44	Avg SST	Under	0.8635	0.9242	Min SST
Geukensia demissa	Both	96.77	96.97	Avg SST	Both	97.25	96.97	Avg NO3	Under	0.9309	0.8987	Avg NO3
Gracilaria vermiculophylla	Over	97.81	91.18	Max SST	Over	99.08	91.18	Max NO3	Under	0.904	0.9024	Max SSS
Hemigrapsus sanguineus	Over	98.47	92.16	Avg SST	Both	98	94.12	Max SST	Under	0.9634	0.963	Avg SST
Hypnea musciformis	Both	95.17	81.48	Avg SST	Over	98.11	81.48	Max SSS	Under	0.8184	0.8621	Avg pH
Littorina littorea	Over	97.55	94.26	Avg SST	Over	97.98	96.17	Avg SST	Over	0.986	0.9698	Avg SST

Musculista senhousia	Both	96.21	86.21	Min SST	Over	98.1	89.66	Max NO3	Both	0.9814	0.831	Avg SST
Mya arenaria	Over	96.09	96.95	Avg SST	Under	93.44	97.71	Max SST	Both	0.9749	0.8508	Avg NO3
Mytilopsis leucophaeta	Over	97.9	84.44	Min SST	Under	92.65	100	Max SST	Under	0.9234	0.8548	Min SST
Mytilus galloprovincialis	Over	95.99	93.8	Min SST	Under	93.54	93.02	Avg SST	Under	0.918	0.9291	Avg SST
Polysiphonia brodiei	Both	96.9	98.41	Max SST	Under	95.23	100	Avg SST	Under	0.9611	0.9692	Max SST
Rangia cuneata	Over	98.95	97.56	Avg SST	Both	98.84	97.56	Avg SSS	Under	0.9452	0.9615	Min NO3
Rapana venosa	Over	98.25	88.37	Avg SST	Over	98.95	97.67	Max SSS	Under	0.9331	0.9783	Min SST
Rhithropanopeus harrissii	Both	96.48	92.31	Max SST	Both	97.33	93.59	Min SSS	Under	0.9439	0.9247	Min SST
Sabella spallanzanii	Both	93.36	87.84	Min SST	Over	95.65	87.84	Avg SST	Both	0.9448	0.9351	Max SST
Schizoporella errata	Over	98.62	79.41	Max SST	Both	98.96	82.35	Max SSS	Both	0.9801	0.8889	Max pH
Schizoporella unicornis	Over	95.23	90.55	Avg SST	Over	96.65	91.34	Avg SST	Under	0.9265	0.8993	Max SST
Styela clava	Over	96.64	95.54	Avg SST	Over	97.45	92.99	Max SSS	Over	0.9674	0.8908	Max SST
Styela plicata	Both	94.93	87.04	Max SST	Under	90.57	98.15	Min SST	Both	0.9787	0.9138	Avg NO3
Undaria pinnatifida	Over	97.76	96.36	Max SST	Both	96.81	92.73	Min SST	Under	0.9204	0.9683	Avg SST
Watersipora subtorquata	Under	87.56	88.24	Min SST	Both	99.19	79.41	Max SSS	Over	0.9943	0.7179	Avg SST

Table A1.6b This table provides details of the Seasonal Model performance for each species by the type of variable aggregation. The resampling technique which created the best model is identified, as is the most important variable to the model.

	SST Only				SST, SSS, NO3				SST, SSS, NO3, PH			
	Resamp.	Predict Absence	Predict Presence	Most imp var	Resamp.	Predict Absence	Predict Presence	Most imp var	Resamp.	Predict Absence	Predict Presence	Most imp var
Alitta succinea	Over	96.5	97.91	Summer SST	Over	97.36	98.96	Summer SST	Both	0.9433	0.9318	Summer SST
Ascidiella aspersa	Over	95.82	98.77	Autumn SST	Over	96.94	99.26	Spring SST	Under	0.9462	0.9278	Autumn SST
Asterias amurensis	Over	98.21	100	Summer SST	Over	99	100	Winter SST	Under	0.8832	0.9111	Winter NO3
Bugula neritina	Over	96.51	99.67	Summer SST	Over	98.34	99.67	Summer SST	Under	0.9007	0.9185	Autumn NO3
Carcinus maenas	Over	96.47	98.99	Autumn SST	Over	97.77	98.58	Autumn SST	Under	0.9341	0.9541	Summer SST
Charybdis japonica	Over	99.85	100	Autumn SST	Over	99.9	100	Summer SSS	Over	0.9968	0.7647	Winter pH
Ciona intestinalis	Over	97.75	98.48	Summer SST	Over	97.99	99.35	Autumn SST	Under	0.9445	0.9804	Winter SST
Codium fragile	Over	95.11	99.12	Autumn SST	Over	97.45	99.12	Summer SST	Both	0.9593	0.8824	Autumn SST
Crassostrea gigas	Over	95.96	99.06	Summer SST	Over	97.74	99.29	Autumn SST	Under	0.9005	0.9412	Spring SST
Crepidula fornicata	Over	95.44	99.28	Autumn SST	Over	97.28	99.28	Summer SST	Both	0.9689	0.9577	Winter SST
Didemnum vexillum	Over	98.24	99.19	Summer SST	Over	97.63	100	Winter SSS	Under	0.958	1	Winter SSS
Elminius modestus	Over	98.52	99.55	Autumn SST	Over	99.36	100	Autumn SST	Under	0.9374	0.9388	Summer SST
Ficopomatus enigmaticus	Over	97.61	100	Spring SST	Over	98.53	100	Autumn SST	Both	0.8906	0.9732	Spring SSS
Geukensia demissa	Over	98.62	100	Winter SST	Over	98.97	100	Winter SSS	Under	0.9593	0.8831	Autumn NO3
Gracilaria vermiculophylla	Over	96.05	100	Autumn SST	Over	98.32	100	Autumn SSS	Under	0.9514	0.9722	Summer SSS
Hemigrapsus sanguineus	Over	97.78	100	Winter SST	Over	98.39	100	Winter NO3	Both	0.9824	0.9444	Spring pH
Hypnea musciformis	Over	98.84	98.45	Autumn SST	Over	99.34	98.45	Autumn SST	Under	0.8783	0.7368	Spring pH

Littorina littorea	Over	98.64	98.98	Autumn SST	Over	99.13	99.18	Winter SST	Under	0.9663	0.9635	Summer SST
Musculista senhousia	Over	98.12	100	Spring SST	Over	98.53	100	Autumn SST	Under	0.8919	0.9531	Autumn SST
Mya arenaria	Over	98.46	97.88	Spring SST	Over	98.86	98.86	Autumn SST	Under	0.9461	0.967	Winter SST
Mytilopsis leucophaeta	Over	98.15	100	Spring SST	Over	98.9	100	Winter SSS	Under	0.8974	0.9818	Spring pH
Mytilus galloprovincialis	Over	96.9	99.34	Summer SST	Over	98.28	99.67	Spring SST	Under	0.9299	0.9254	Spring SST
Polysiphonia brodiei	Over	98.37	99.32	Winter SST	Over	98.98	99.32	Winter NO3	Both	0.9821	0.9846	Spring SST
Rangia cuneata	Over	99.55	100	Autumn SST	Over	99.6	100	Winter NO3	Over	0.9956	0.9184	Spring pH
Rapana venosa	Over	98.11	100	Spring SST	Over	98.95	100	Autumn SST	Over	0.9836	0.9778	Autumn SST
Rhithropanopeus harrissii	Over	96.93	100	Summer SST	Over	97.97	100	Winter SST	Under	0.9348	0.9195	Summer SST
Sabella spallanzanii	Over	94.36	99.43	Spring SST	Over	97.1	100	Autumn SST	Under	0.871	0.9091	Autumn SST
Schizoporella errata	Over	97.48	100	Autumn SST	Over	99.11	98.77	Autumn SST	Under	0.8573	0.9444	Spring pH
Schizoporella unicornis	Over	96.41	98.66	Spring SST	Over	98.12	99.67	Spring SST	Over	0.9722	0.9323	Autumn SST
Styela clava	Over	97.18	99.46	Autumn SST	Over	98.33	99.46	Autumn SST	Under	0.9309	0.9818	Autumn SST
Styela plicata	Over	97.53	100	Winter SST	Over	99.04	100	Spring SST	Under	0.8627	0.8966	Winter SST
Undaria pinnatifida	Over	98.08	99.24	Spring SST	Over	99.19	99.24	Spring SST	Under	0.8843	0.9194	Winter NO3
Watersipora subtorquata	Over	99.41	100	Autumn SST	Over	99.85	100	Autumn SST	Under	0.9049	0.8056	Spring pH

A.2 Predicted species information

Details for which variable aggregation set lead to the prediction can be found in the manuscript Results – Table 2 and Table 3.

Charybdis japonica was predicted to be environmentally suited by the annual model to:

a. Casey

The largest driver of occurrence in the model was average annual pH, followed by maximum pH and maximum salinity.

The occurrence records of this species on the Ocean Biogeographic Information System show that it has been found mainly in northern China and Korea, along with Japan, Australia, and New Zealand. The species status as invasive is well-established in the literature (e.g. Fowler, Gerner, and Sewell 2011; Wong et al. 2016).

Geukensia demissa was predicted to be environmentally by the annual model to:

a. Mawson

b. Davis

c. Casey

The occurrence of this species is driven by maximum nitrate, followed by maximum pH and average nitrate.

The occurrence records of this species on the Ocean Biogeographic Information System indicate that this species is found in the United States, northern Spain, and on the Antarctic Peninsula (King George Island). There is no information in the literature regarding the Antarctic finding of this species. The OBIS record makes reference to a published study (Tatián et al., 2008) as a bibliographic reference, however this paper mentions *G. demissa* as a comparative species for carbon flux in suspension feeders. Therefore, this species is either: a) incorrectly identified as *G. demissa,* or b) the first record of this MIS in the greater Antarctic region.

Hypnea musciformis was predicted to be environmentally suited by the annual model to:

a. Davis

The occurrence of this species is driven by average annual pH, followed by minimum pH and maximum pH. The occurrence records of this species on the Ocean Biogeographic Information System indicate that this species is found on every continent, except Antarctica. It is particularly common in temperate and tropical regions, however it is also found in regions with cooler temperatures, such as the United Kingdom, Russia, Tasmania, and Île Amsterdam. The species status as invasive is well-established in the literature (e.g. Smith, Hunter, and Smith 2002).

A.2.1 Asterias amurensis

Asterias amurensis is an apex predatory star fish with a native range spanning the Northern Pacific from the Arctic to Japan (Byrne et al., 2013). It was introduced to the Port of Hobart, Derwent River, Tasmania in the 1980s through ballast water release (Buttermore et al., 1994), and has been secondarily introduced to Port Phillip Bay, Victoria in the 1990s (Murphy and Evans, 1998). *Asterias amurensis* is currently subject to a National Control Plan due to its "*...having significant and potential future impacts on Australia's marine environment, social uses of the marine environment and the economy*" (Australian Government, 2008b, p. 11).

This species displays considerable phenotypic plasticity and can alter spawning times to coincide with local conditions (Buttermore et al., 1994; Byrne et al., 1997; Ling et al., 2012). The planktonic larvae stage is long-lived (120 days) and larval concentrations recorded in the Derwent River, Tasmania, are among the highest ever recorded (>1,100 larvae/m^3) for echinoderms (Bruce, 1998). The spawning periods for the Tasmanian population occurs between April to October, coinciding with the Austral winter period (Byrne et al., 1997; Hayes et al., 2004; Ling et al., 2012). Concerningly, this coincides with the overwinter harbouring of the Australian Antarctic research vessel *Aurora Australis*, with travel to the Antarctic and subantarctic recommencing in October each year. In its native range in Japan, *A. amurensis* also spawns in winter and early spring; however, in its other native range in Russia it spawns in summer and autumn (Byrne et al., 1997).

Photoperiod has been suggested as the trigger for spawning to occur, which corresponds to roughly 9 – 10 hours of daylight in Japan and Tasmania, during the winter solstice. How this could translate in the Antarctic and subantarctic realm remains to be understood, as the continental stations experience months of twilight through the winter months. The same level of daylight hours is reached in the Antarctic around September each year; a month before the shipping season recommences. This mismatch of day length may confer some protection against the species becoming established in the Antarctic region, however the fact that it shows spawning plasticity in relation to region may nullify this protection. This species also has an introduced range through Canada and Alaska, indicating the species ability to tolerate cold conditions and the presence of sea ice (Byrne et al., 2016).

A.2.2 Undaria pinnatifida

Undaria pinnatifida is a commercially harvested kelp species in its native Japan, Korea and China (Shibneva et al., 2013; Uwai et al., 2006) . The introduced range encompasses Australia, New Zealand, Europe, Argentina, and California (James et al., 2015). In Australia, it is also subject to a National Control Plan (Aquenal Pty Ltd, 2008b). *U. pinnatifida* is a poor competitor that struggles to establish in stable environments but thrives in disturbed environments (James and Shears, 2016a; Valentine and Johnson, 2003). Disturbance is predicted to occur more often in Antarctic and subantarctic benthic ecosystems via increased iceberg scouring (Barnes and Souster, 2011; Peck et al., 2005), less sea ice opening up new areas to iceberg scour, and increased winds leading to larger waves in coastal areas (Stark et al., 2019).

U. pinnatifida undergoes a seasonal life cycle in its native range; alternating a microscopic gametophytic stage from late summer until winter, and a macroscopic sporophytic stage from early winter to mid-summer (Campbell et al., 2005; Choi et al., 2007). Spores from mature sporophytes are typically released in spring (Choi et al., 2007). In the invaded range, however, the life cycle has been shown to be either biannual, as seen in California (Thornber et al., 2004), or year-round, as seen in Australia (Campbell et al., 2005; Schaffelke et al., 2007), Argentina (Casas et al., 2008), France (Floc'h et al., 1991), North America (Fletcher and Farrell, 1999; Zabin et al., 2009), and New Zealand (James and Shears, 2016b). The release of spores in Tasmania can occur year-round, however the rate of spore release is seasonal, with maximum releases expected in March and September (Campbell et al., 2005).

As with *A. amurensis*, the timing of spore release corresponds to a period when the *Aurora Australis* was overwintering in Hobart. Sporophytes form roughly 14 to 30 days after fertilization, and can grow to over one metre in length within 3 – 5 months (Campbell et al., 2005). *U. pinnatifida* employs differing dispersal mechanisms for short- and long-range spread. Spores in the water column have been shown to only support short-range dispersal (~ 100 m yr^{-1}) and lose their ability to attach by day 14, whilst long-range (many kilometres) dispersal occurs when whole or fragments of sporophytes are carried by water currents that can subsequently release spores at more distant locations (Forrest et al., 2000). It has also been suggested that the microscopic life stage of this species can form an analogous 'seed bank' over the long term (Campbell et al., 2005). As disturbance via iceberg scour can occur throughout the year and recovery from a scour event can take many years, there is potential for this species to exploit areas of suitable habitat if it is transported to the region (Stark et al., 2019).

A.3 Machine learning with imbalanced data

A.3.1 Issues with imbalanced datasets

Class imbalance in data occurs when the proportion of positive (minority) classes is significantly less than those of the negative (majority) class (Leevy et al., 2018). In m study, this meant that the number of occurrences of marine invasive species in global ports was significantly less than the number of absences. The majority of marine invasive species were found in less than 10% of ports worldwide, and 11 species were found in less than 1% of ports. This could lead to a classifier achieving a very high overall accuracy in prediction, simply by only predicting the majority class (Díez-Pastor et al., 2015; Leevy et al., 2018). There are two general methods to overcome the problem of imbalanced data: at the data-level and at the algorithm-level.

A.3.2 Resampling techniques used in this study

Four data resampling techniques were used in this study: over-sampling, under-sampling, combined over- and under-sampling, and ROSE (synthetic minority oversampling). These were conducted using the R packages ‘caret’ and ‘ROSE’(Kuhn, 2019; Lunardon et al., 2015). It is important to note, these four techniques do not represent the full suite of resampling techniques that are available for handling imbalanced data.

Over-sampling (with replacement) increases the number of minority class samples by creating random copies of those found in the original data, such that samples from the original data can be copied more than once. The number of minority class samples is increased to equal the number of majority class samples in the dataset. This can help identify the most important variables and improve overall classification accuracy (Leevy et al., 2018). A recent review of methods to address imbalanced datasets found that over-sampling was often the best performing resampling method to improve classification accuracy, however overfitting of the minority class may occur (Leevy et al., 2018). In my study over-sampling was the best performing resampling technique for both the annual and seasonal models. We did not suffer from overfitting as we maintained high accuracy in predicting presences and absences.

Under-sampling (without replacement) decreases the number of majority class samples by randomly removing majority class samples down to a number equal to the minority class samples. Similarly to over-sampling, the aim of this technique is to improve overall classification accuracy (Leevy et al., 2018). This technique performed relatively well in this study for the annual model. This technique should be used with caution however, as you may lose important information by reducing the majority class, so that it is no longer indicative of the distribution of variables in the majority class (Leevy et al., 2018).

Both over- and under-sampling combines the two techniques mentioned previously, with random over-sampling with replacement and random under-sampling without replacement. In this case the total dataset retains the same number of records by randomly reducing the number of the majority class to half the original dataset size and replicating minority class samples up to half the original dataset size. As for over-sampling and under-sampling, the aim is to improve classification accuracy. It has the benefit of not removing as many potentially important samples from the majority class, as well as reducing the chance of over-fitting by the minority class (Leevy et al., 2018). This technique also performed relatively well in my study when using the annual model.

ROSE resampling is a type of over-sampling technique which creates synthetic minority samples. This method generates new minority samples using a smoothed bootstrap technique in order to better classify the minority class. New samples correspond to the kernel density estimate of kernel K, which defines the contour of the neighbourhood shape, and a smoothing matrix (for further detail see Lunardon et al., 2015). Generation of new artificial samples, rather than exact copies of existing samples, is designed to help avoid the potential overfitting of the minority class and increase model generalization (Leevy et al., 2018; Lunardon et al., 2014). However, in my study, there were no species for which this resampling technique worked best. In most cases, overfitting of the minority class occurred with relatively (and sometimes extreme) low accuracy in the majority class.

A.4 Sample code to run analysis with XGBoost and caret

XGBoost for classification

XGBoost - or Extreme Gradient Boosting - is an Optimized Gradient Boosting algorithm that builds on Random Forest and Gradient Boosting algorithms.

Extreme gradient boosting is an ensemble machine learning technique that makes predictions based on combinations of multivariate predictor data producing a specific outcome. For this study we elected to use environmental variables at ports (the predictor variables) to predict the presence or absence of an invasive species (the response variable). This method creates an ensemble of decision trees which aims to create a strong classification (or regression) model based on a set of 'weak' classifiers of the response variable.

Here we use the 'xgboost' and 'caret' packages in R (Chen et al., 2019; Kuhn, 2019). Some advantages of XGBoost over other machine learning techniques are regularization to avoid overfitting, parallel processing capabilities, and inbuilt cross validation capabilities – particularly when paired with the 'caret' package in R.

Despite the paucity of use in invasive species research, XGBoost is a popular modelling algorithm in many other fields; such as: critical care management (Chang et al., 2019; Zhang et al., 2019); financial fraud detection (Zhou et al., 2018) and credit scoring (Munkhdalai et al., 2019); and satellite image (Just et al., 2018) and astronomical feature classification (Tamayo et al., 2016). The XGBoost system has consistently outperformed other machine learning algorithms and often features in winning entries in data science competitions (Chen and Guestrin, 2016). Other machine learning algorithms have been used to predict invasive species distributions with random forest, support vector machine, and other gradient boosted algorithms performing well (Früh et al., 2018; Shiferaw et al., 2019).

Required libraries

These are the libraries required to run XGBoost with resampling and the tuning parameters in ‘caret’.

```
library(gdata)
library(caret)
library(doSNOW)
library(ROSE)
library(xgboost)
```

Data Import

This assumes that your spreadsheet has one row for each location (in my study, each global port) and that the columns include each of the environmental variables (e.g. mean winter temperature, annual average salinity, etc) and that there is a column for each of your species indicating that there is either ‘YES’ for presence at that location or ‘NO’ if it is absent from this location. Modify the file path and the sheet number to correspond to your data spreadsheet of interest.

```
# Modify to your file path and sheet number within the
spreadsheet file

import <- read.xls("path/to/my/file.xlsx", sheet = 3, header =
TRUE)

# Import the datasheet you wish to make prediction from

data2 <- read.xls("path/to/new/data.xlsx", sheet = 2, header =
TRUE)

# Input your list of species if you are looking at multiple
species

species <- c("Species A", "Species B", "Species C")
```

Caret parameter tuning

See the ‘caret’ documentation for more options to tune these parameters. Note, the more parameters you have, the more times the model will run, and the longer it will take. You can use `View(tune.grid)` to see how many iterations of the model will be run.

```
train.control <- trainControl(method = "repeatedcv",
                              number = 10,
                              repeats = 3,
```

```
                                      search = "grid")
tune.grid <- expand.grid(eta = c(0.2, 0.3),
                                nrounds = c(100, 250, 500),
                                max_depth = 6:9,
                                min_child_weight = c(2.0, 2.5),
                                colsample_bytree = c(0.3, 0.5),
                                gamma = 0,
                                subsample = c(0.5, 1.0))
```

Set up parallel processing

The number in the makeCluster function is the number of cores of your computer/server. You do not have to use all your cores, however if you input a number greater than the number of cores available it will cause an error.

```
cl <- makeCluster(4, type = "SOCK")
registerDoSNOW(cl)

# To close parallel processing place this at the very end of your
code - OUTSIDE of any loop you create below.

stopCluster(cl)
```

XGBoost loop for each species

If you have multiple species, wrap this loop code around all the code in the following sections. Note that you will need to change the code in the features to features <- c("Variable_1", "Variable_2", "Variable_n", i).

```
for(i in (species)){
}

# You would place the "stopCluster(cl)" here if you run a loop.
```

Specify variables

Your original dataset could have superfluous columns, so use this to specify which subset of variables you are interested in. This will create a new subset of data with only the variables of interest and the single species name for analysis. As mentioned previously, this can be run within a loop by changing "Species_name" to i.

```
data <- import

features <- c("Variable_1", "Variable_2", "Variable_n",
"Species_name")

model <- data[, features]
```

Data PreProcessing

This imputes missing data by fitting a bagged model of all the other explanatory variables.

```
# Impute missing values via a bagged model based on all the other
explanatory variables (i.e. environmental variables)

model.pre.process <- preProcess(model, method = "bagImpute")
model.imputed.data <- predict(model.pre.process, model)

# Insert imputed values into dataset - number is the reference to
the column in the dataset

model$Variable_1 <- model.imputed.data[,1]
model$Variable_2 <- model.imputed.data[,2]
model$Variable_n <- model.imputed.data[,n]
```

Train and test split

We divided the model building dataset into training (70%) and testing (30%) datasets.

```
# Change species column name to 'y' - number corresponds to the
column number of your species presence/absence

names(model)[13] <- "y"

# You can set a seed for reproducibility

set.seed(123)

# Create training and test datasets
# Using the createDataPartition function keeps the ratio of
presence:absence consistent between training and testing datasets

indexes <- createDataPartition(model$y, times = 1, p = 0.7, list =
FALSE)

train <- model[indexes,]
```

```
test <- model[-indexes,]

names(train)[13] <- "y"
names(test)[13] <- "y"
```

Resampling imbalanced datasets

Due to imbalanced data we analysed the performance of four resampling techniques: over-sampling, under-sampling, both over- and under-sampling, and synthetic over-sampling (see below). The resampling technique which was best able to predict the presence of a species, whilst also maintaining a high accuracy in predicting the absence of a species, was deemed the best model. Higher accuracy in predicting the presence of species was given a higher preference as it is likely that the presence data is accurate, whereas absence data are likely to be less accurate as the data is more prone to Type II errors.

```
no <- sum(train$y == "NO")
yes <- sum(train$y == "YES")

# Over-sampling of the minority class by duplicating minority
class entries

over <- ovun.sample(y ~ Variable_1 + Variable_2 + Variable_n,
                    data = train,
                    method = "over",
                    seed = 234,
                    N = (no * 2))$data

# Under-sampling of the majority class by random removal

under <- ovun.sample(y ~ Variable_1 + Variable_2 + Variable_n,
                     data = train,
                     method = "under",
                     seed = 345,
                     N = (yes * 2))$data

# Both over- and under-sampling of both classes to produce
# a dataset which is of equal size to the original dataset

both <- ovun.sample(y ~ Variable_1 + Variable_2 + Variable_n,
                    data = train,
                    method = "both",
                    seed = 456,
                    N = (no + yes))$data

# Create new artificial samples of the minority class by over-
```

```
sampling in k-space
# so that the total number of samples is equal to twice the
majority class

rose <- ROSE(y ~ Variable_1 + Variable_2 + Variable_n,
             data = train,
             seed = 567,
             N = (no * 2))$data
```

Run the XGBoost model on each resampled dataset

Including importance = TRUE will allow you to see the importance of each variable in the final model. This code will run through multiple iterations of the model and the output will be the best model result.

```
# The code is the same for each of the resampled datasets, except
that you change the "data =" to match the resampled dataset names
above
# This code can take a long time to run, particularly if you have
included multiple levels within the tune.grid and train.control
functions
# For example, taking the above code as is, the model will run
192 times for each unique set of tune.grid parameters and took
several hours to complete one species - and more than a week to
complete all 33 species in my study.

cv.over <- train(y ~ .,
                 data = over,
                 method = "xgbTree",
                 tuneGrid = tune.grid,
                 trControl = train.control,
                 importance = TRUE)
```

Test the model with unseen test dataset

This section compares how well our model predicted presence/absence when exposed to the 30% test dataset that we created.

```
# The code is the same for each of the resampled datasets, except
you change the "cv.over" to match the model created in the
previous step. It is also a good idea to change the name it is
saved as to match the resampling method, e.g. "preds.under".

preds.over <- predict(cv.over, test)
```

```
# See the confusion matrix which will tell you how well your model performed.

confusionMatrix(preds.over, test$y)
```

Predictions with your new data

You may have only one, or many, new datasets that you wish to make a prediction about. In our study we looked at multiple time periods (Current, 2030, 2050, 2100) as well as two Representative Concentration Pathway (RCP 4.5 & RCP 8.5), so in total we had 7 newdata sets. They were imported individually as data2, data3, and so on.

```
# These are the predictions you want to make on your new area, new time period, etc. Change the name to suit your prediction dataset/s.

year_2050.over <- predict(cv.over, newdata = data2)
year_2030.under <- predict(cv.under, newdata = data3)

# Find the variable importance for your model

varImp(cv.over)
```

Environmental variable importance was recorded for each model to show which environmental variables contributed most to the model. The best model as described here was used to predict the invasive species environmental suitability at each the five Australian Antarctic and subantarctic locations currently; and at 2030, 2050, and 2100 under the two representative concentration pathways RCP 4.5 and RCP 8.5. Only species which experience 11°C or less were included in the study, to correspond to the maximum predicted temperatures in the subantarctic sites. Only species which experience sub-zero temperatures were considered for the Antarctic sites due to the physiological constraints of organisms below freezing.

Variable Importance

This section shows you how important the various variables were to the final model. If you do not include scale = FALSE, it will scale the most important variable as 100, and show the other variables in respect to that. When you do include it, it will show variable importance as value out of 100 for all variables.

```
imp.over <- varImp(cv.over, scale = FALSE)
```

Print results to an external file

If you want to print results to an external file (which I recommend after losing so much time to random computer freezes or reboots) you can use the code below. I put this inside the loop so that the results are printed as they are processed and if you need to restart, you know which species you were up to.

```
# Adding append = TRUE means that new results will be added to
the existing file. This opens up the dataset for printing only to
file and not to screen - be sure to close it off within a loop
when you have finished outputting results.

sink("file/path/to/your/output/file.txt", append = TRUE)

# Print the confusion matrix for each of your resampled models so
that you can easily compare all models
# Run several in sequence to print out results from all resampled
models by changing " - Oversampling" to " - Undersampling" AND
preds.over to preds.under, etc

print(paste0(i, " - Confusion Matrices"))
print(paste0(i, " - Over-sampling"))
print(confusionMatrix(preds.over, test$y))

# Print predictions for the new data
# On the second line I like to include a legend of what each row
represents, for example in my newdata sets each row represented 1
of 5 Antarctic and subantarctic locations
# Add as many lines as you need to print all your outputs

print(paste0(i, " - Oversampling Predictions"))
print("1 - Mawson; 2 - Davis; etc for all rows")
print("What your newdata set represents (RCP 4.5 2030
Undersampling)")
print(year_2100.under)

# Print the variable importance

print(paste0(i, " - Variable Importance for Oversampling"))
print(imp.over)

# Close the open file
sink()
```

Appendix B: Supplementary Materials for Chapter 5

R Code for this chapter is available on github:
https://github.com/OakesHolland/Antarctic_EEM

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B.1 Species interaction matrices

Asterias amurensis is a predatory sea star native to the northern Pacific Ocean which has been introduced to southern Australia (Aquenal Pty Ltd, 2008a). It is a generalist predator with a preferences for bivalves which will be the assumed prey for *A. amurensis* in this study (Ross et al., 2003b). Several Antarctic species are known to eat predatory star fish, such as *Polynoidae sp., Odontaster validus, Urticinopsis antarctica,* and *Trematomus bernacchii*, and will be the assumed predators for *A. amurensis* in this study.

Table B1.1 Species interaction matrix with *Asterias amurensis* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Asterias amurensis	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Asterias amurensis	-1	0	1	-1	0	0	0	0	0	0	-1	0	-1
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	-1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	0	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	0	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	0	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	1	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	1	0	1	-1	1	2	1	2	2	1	2	-1	-1

Botryllus schlosseri is an invasive colonial ascidian, originally thought to be native to the Mediterranean and Black seas (Cima et al., 2015). It is a strong competitor against native filter feeders, and are positively impacted in disturbed environments (Cima et al., 2015; Nydam et al., 2017). Echinoderms, gastropods, and fish are known to predate on *B. schlosseri* and will be the assumed predators for this study (Giachetti et al., 2020).

Table B1.2 Species interaction matrix with *Botryllus schlosseri* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Botryllus schlosseri	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Botryllus schlosseri	-1	0	1	-1	-1	0	0	0	-1	-1	0	0	0
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	-1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	1	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	0	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	1	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	1	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	0	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	0	0	1	-1	1	2	1	2	2	1	2	-1	-1

Charybdis japonica is a generalist crab with a native range in the north west Pacific Ocean (Townsend et al., 2015). It mainly feeds on suspension feeders, crustaceans, bivalves, and echinoids (Townsend et al., 2015). There is a lack of information on the predators of this species, and as such for this study I have used the predators of similar sized crabs from the same family: *Carcinus maenas* and *Ovalipes catharus* (Garside et al., 2015; McLay, 1988). The assumed predator for *C. japonica* in this study is the benthic fish group.

Table B1.3 Species interaction matrix with *Charybdis japonica* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Charybdis japonica	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Charybdis japonica	-1	0	1	-1	0	1	1	1	0	0	0	0	0
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	-1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	0	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	-1	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	-1	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	-1	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	0	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	0	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	0	0	1	-1	1	2	1	2	2	1	2	-1	-1

The *Mytilus sp.* group are bivalves that are native to the Mediterranean with an vast invaded range (Branch and Nina Steffani, 2004). They have shown a competitive advantage over sessile organisms in their invaded range (Branch and Nina Steffani, 2004). The assumed predators in this study were assumed to be the same as predators of Antarctic bivalve species, namely benthic fish, omnivorous and predatory echinoderms, predatory anemones, predatory pycnogonids, and predatory polychaetes.

Table B1.4 Species interaction matrix with *Mytilus sp.* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Mytilus	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Mytilus	-1	0	1	-1	-1	0	0	0	-1	0	-1	-1	-1
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	-1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	1	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	0	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	1	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	1	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	1	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	1	0	1	-1	1	2	1	2	2	1	2	-1	-1

Hypnea musciformis is a red algae native to North America, Central America and South America (Global Invasive Species Database, 2006). This species outcompetes native algae to produce monocultures (Alidoost Salimi et al., 2021). For this study, I assume that native species which feed on macroalgae will also consume this invasive species.

Table B1.5 Species interaction matrix with *Hypnea musciformis* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Hypnea musciformis	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Hypnea musciformis	-1	1	-1	0	-1	-1	0	0	0	0	0	0	0
Macroalgae	-1	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	0	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	1	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	1	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	0	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	0	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	0	0	1	-1	1	2	1	2	2	1	2	-1	-1

Halicarcinus planatus is an omnivorous/detritivore crab native to South America and the subantarctic (Dietz et al., 2018). This species consumes carrion and sediment particulate organic matter, both of which do not appear in my network as nodes (López-Farrán et al., 2021). This is because carrion and particulate organic matter are not limiting in the food web, and as such, the inclusion of these would provide no additional benefit to the network analysis (Baker et al., 2017). In its native range it is eaten by fish, birds, crabs and sea stars (López-Farrán et al., 2021). For this study I assume that benthic fish, and omnivorous and predatory echinoderms will predate on *H. planatus.*

Table B1.6 Species interaction matrix with *Halicarcinus planatus* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Halicarcinus planatus	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Halicarcinus planatus	-1	0	0	-1	-1	0	0	0	-1	0	0	0	0
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	0	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	1	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	0	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	1	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	0	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	0	0	1	-1	1	2	1	2	2	1	2	-1	-1

Carcinus maenas is an omnivorous crab native to the Atlantic region of Europe (Young and Elliott, 2020). This species primarily feeds on bivalves, though it will incidentally feed on other small organisms, with the exception of echinoderms (Grosholz and Ruiz, 1996). Predators of this crab include large fish, large crabs, and cephalopods (Garside et al., 2015). For this study I assume that predators of *C. maenas* would be the benthic fish group.

Table B1.7 Species interaction matrix with *Carcinus maenas* as the invasive species. -1 indicates that there is a negative impact on row species by the column species, 1 indicates there is a positive impact on row species by the column species, 0 is no interaction, 2 indicates there is a mutual predator/prey relationship where both the row species and columns species have a positive and negative on each other. Letters in brackets refer to trophic level – O omnivore, P predator.

	Carcinus maenas	Macroalgae	Herbivores	Benthic fish	Echinoderms (O)	Echinoids (O)	Crustaceans (P)	Amphipods (P)	Echinoderms (P)	Gastropods (P)	Anemones (P)	Pycnogonids (P)	Polychaetes (P)
Carcinus maenas	-1	0	1	-1	0	0	0	0	0	0	0	0	0
Macroalgae	0	-1	-1	0	-1	-1	0	0	0	0	0	0	0
Herbivores	-1	1	-1	-1	-1	-1	0	0	-1	-1	-1	-1	-1
Benthic fish	1	0	1	-1	0	0	1	1	0	0	0	0	1
Echinoderms (O)	0	1	1	0	-1	1	1	1	2	1	0	0	-1
Echinoids (O)	0	1	1	0	-1	-1	0	1	0	0	-1	-1	2
Crustaceans (P)	0	0	0	-1	-1	0	-1	0	0	0	0	0	-1
Amphipods (P)	0	0	0	-1	-1	-1	0	-1	0	0	0	0	2
Echinoderms (P)	0	0	1	0	2	0	0	0	-1	0	-1	0	2
Gastropods (P)	0	0	1	0	-1	0	0	0	0	-1	0	-1	-1
Anemones (P)	0	0	1	0	0	1	0	0	1	0	-1	-1	2
Pycnogonids (P)	0	0	1	0	0	1	0	0	0	1	1	-1	1
Polychaetes (P)	0	0	1	-1	1	2	1	2	2	1	2	-1	-1

Appendix C: Supplementary Materials for Chapter 7

C.1 Scoped policy instruments

Table D1.1 International agreements signed by Australia. Green indicates the spatial coverage of the agreement to the case study sites. Note that Heard Island refers to both Heard Island and McDonald Island.

	Policy Instrument	Antarctica	Heard Island	Macquarie Island
Antarctic Treaty System	*Antarctic Treaty 1959*			
	Protocol on Environmental Protection (Madrid Protocol) 1991			
	Convention on the Conservation of Antarctic Marine Living Resources 1980			
International Maritime Organization	*International Convention for the Prevention of Pollution from Ships (MARPOL) 1973*			
	International Code for Ships Operating in Polar Waters (MARPOL - Polar Code) 2014			
	International Convention on the Control of Harmful Anti-fouling Systems 2001			
	International Convention for the Control and Management of Ships' Ballast Water and Sediments 2004			
United Nations Environment Programme	*Convention on Biological Diversity 1992*			
United Nations Educational, Scientific and Cultural Organization	*World Heritage Convention 1972*			
United Nations Division for Ocean Affairs and the Law of the Sea	*Convention on the Law of the Sea 1982*			

Table D1.2 Australian national and state level policy instruments and formal management plans. (Cth) indicates federal level policy and (Tas) indicates Tasmanian state level policy. Green indicates the spatial coverage of the policy to the case study sites.

<table>
<tr><th>Policy Instrument</th><th>Antarctica</th><th>Heard Island</th><th>Macquarie Island</th></tr>
<tr><td>Antarctic Treaty Act 1960 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Treaty Regulations 1993 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Treaty (Environment Protection) Act 1980 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Treaty (Environment Protection) (Environmental Impact Assessment) Regulations 1994 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Treaty (Environmental Protection) (Waste Management) Regulations 1994</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Marine Living Resources Conservation Act 1981 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Antarctic Marine Living Resources Conservation Regulations 1994 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Protection of the Sea (Prevention of Pollution from Ships) Act 1983 (Cth)</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Protection of the Sea (Prevention of Pollution from Ships) (Orders) Regulations 1994 (Cth)</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Protection of the Sea (Prevention of Pollution from Ships) Amendment (Polar Code) Act 2017 (Cth)</td><td style="background:#00b050"></td><td></td><td></td></tr>
<tr><td>Biosecurity Act 2015 (Cth)</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Biosecurity Regulations 2016 (Cth)</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Protection of the Sea (Harmful Anti-fouling Systems) Act 2006 (Cth)</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Environment Protection and Biodiversity Conservation Act 1999 (Cth)</td><td></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Environment Protection and Biodiversity Conservation Regulations 2000 (Cth)</td><td></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Heard Island and McDonald Islands Act 1953 (Cth)</td><td></td><td style="background:#00b050"></td><td></td></tr>
<tr><td>National Parks and Reserves Management Act 2002 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>National Parks and Reserves Management Regulations 2019 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Nature Conservation Act 2002 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Threatened Species Protection Act 1995 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Threatened Species Protection Regulations 2016 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Biosecurity Act 2019 (Tas)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Australian Antarctic Division Environmental Policy 2018 – 2022</td><td style="background:#00b050"></td><td style="background:#00b050"></td><td style="background:#00b050"></td></tr>
<tr><td>Heard Island Fisheries Management Plan (2002)</td><td></td><td style="background:#00b050"></td><td></td></tr>
<tr><td>Heard Island and McDonald Island Marine Reserve Management Plan 2014 – 2024</td><td></td><td style="background:#00b050"></td><td></td></tr>
<tr><td>Macquarie Island Marine Park Management Plan (2001)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Macquarie Island Nature Reserve and World Heritage Area Management Plan (2006)</td><td></td><td></td><td style="background:#00b050"></td></tr>
<tr><td>Macquarie Island Toothfish Fishery Management Plan (2006)</td><td></td><td></td><td style="background:#00b050"></td></tr>
</table>

Table D1.3 Informal plans, strategies, and guidelines.

Issuing Organization	Plans, strategies, and guidelines
International Association of Antarctic Tour Operators	Guidelines for Visitors to the Antarctic
	Guidance for those Organising and Conducting Tourism and Non-Governmental Activities in the Antarctic
	Yachting guidelines for Antarctic cruises
United Nations Educational, Scientific and Cultural Organization	United Nations Decade of Ocean Science for Sustainable Development (2021-2030): Southern Ocean Action Plan (2021-2030)
Australian Government Department of Agriculture and Department of the Environment	Anti-Fouling and In-Water Cleaning Guidelines (2015)
Council of Managers of National Antarctic Programs	Inter-continental checklists for supply chain managers for the reduction in risk of transfer of non-native species
Committee for Environmental Protection to the Antarctic Treaty	Non-native species manual
	Clean-up manual
	Practical guidelines for ballast water exchange in the Antarctic Treaty Area
	Five-year work plan
International Maritime Organisation	Guidelines for the Control and Management of Ships' Biofouling to Minimize the Transfer of Invasive Aquatic Species

C.2 Keywords for gap analysis

The following keywords were used for the policy gap analysis in Chapter 7.

These keywords were used to identify policy instruments that included mention of invasive species and apply as keywords to all policy responses:

non-native	pest	transfer
invasive	non-indigenous	impact*
invasion	introduce*	activit*
alien	exotic	

These keywords were used in relation to the policy response *Prevention and restoration of degraded habitats*:

restor*	disturb*	recovery*
rehabilitat*	degrad*	clean

These keywords were used in relation to the policy response *Management action plans*

manage*	respond	resilience
action	remov*	adverse
response	eradicate*	

These keywords were used in relation to the policy response *Monitoring and surveillance*

monitor*	surveillance

These keywords were used in relation to the policy response *Biosecurity*

biosecurity	quarantine	prevent*

These keywords were used in relation to the policy response *Antifouling*

foul*	biofilm
ballast	attach*