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# Integrating past and present connectivity into the design of marine protected areas to plan for persistence in the Great Barrier Reef

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# Abstract

The recent decline in coral cover worldwide has led to a global concern about the ability of coral reefs to withstand the impacts of climate change. One of the tools developed to protect coral reefs is the design of marine protected areas (MPAs) through systematic conservation planning.

In this context, our work used the Great Barrier Reef (GBR) as its study area. The aim was to identify which reefs would be most resilient in the face of global warming and therefore strategic to consider as part of MPAs. In order to do so, a retrospective method was adopted. We constructed a timeline based on the most important mass bleaching events that hit the GBR over the past decades. The reefs which maintained a certain level of resilience throughout the timeline were considered most fit to ensure persistence on the GBR.

Three main steps led to identifying these reefs. The first step involved calculating the coral cover for the $\sim$ 3800 reefs making up the GBR and this, for each event of our timeline. Coral cover was used as it is an important indicator of reef health. In addition to coral cover, we used the connectivity of corals in the second step to identify important hotspots within the network. This connectivity was weighted with the coral cover to identify temporal variations in the strength of connections between coral reefs. Finally, we integrated the information from the previous steps into Marxan, a decision support tool, to identify a set of priority areas across the GBR through different scenarios. Priority areas that were consistently selected throughout the timeline were considered best able to answer our call for resilience.

Our results show that resilient areas on the GBR only truly started standing out once connectivity was integrated into the design. The areas that showed the highest selection frequencies over the years were located mostly in the south of the GBR, indicating this might be a region worth securing for the future. Our maps also indicate that the current level of protection of the GBR seems to already satisfy a certain level of resilience which could nevertheless be improved by expanding some areas. Our findings also indicate that the approach of considering both healthy and degraded reefs rather than focusing solely on healthy reefs when designing MPAs is more appropriate in today's context. Overall, this work highlights the importance of integrating connectivity into spatial design and more specifically, how combining past and present connectivity may help plan for persistence on the GBR.

# Acknowledgements

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# Contents

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# List of acronyms

AIMS: Australian Institute of Marine Science
$CaCO_3$: Calcium carbonate
COTS: Crown-of-Thorns Starfish
GBR: Great Barrier Reef
GBRMP(A): Great Barrier Reef Marine Park (Authority)
LTMP: Long-Term Monitoring Program (AIMS)
ppt: Parts per thousand
SBS: Shifting baseline syndrome
SST: Sea surface temperature
IDW: Inverse Distance Weighting
MPA: Marine Protected Area
HQI: Habitat Quality Index
BLM: Boundary Length Modifier
SPF: Species Penalty Factor
CSM: Connectivity Strength Modifier

# Introduction

The Great Barrier Reef (GBR), located on the eastern coast of Queensland, Australia, is the world's largest coral reef ecosystem. Known to be one of the most complex natural systems on Earth, it is home to many different species and attracts tourists from around the world which makes it a powerful economic driver (GBRMPA, 2018g). As its name implies, the GBR is most famous for its range of diverse and colourful coral reefs, which provide food and shelter for many species as well as serve as a coastal protection structure.
However, coral reefs around the world are now under constant pressure from anthropogenic sources and have become one of the most vulnerable ecosystems to climate change (Hoegh-Guldberg, 1999; West and Salm, 2003; Donner et al., 2005; Jones et al., 2009; Wolff et al., 2015). Their protection has been recognized as a global priority for effective marine conservation (Strain et al., 2018). To survive, corals depend on a symbiotic relationship with a single-cell algae known as zooxanthellae. These small organisms provide the corals with up to 95% of their carbon requirements. This relationship is sensitive to a variety of factors known as "bleaching stressors". These stressors induce the zooxanthellae to separate from the coral, taking away its photosynthetic pigments which then leads to a white, bleached coral (West and Salm, 2003). A rise in sea surface temperature is an important stressor which can lead to mass coral bleaching events. With the current trends in global temperature, it is likely that we will witness more of these mass bleaching events. Bleaching is not directly associated with mortality but corals may die if the recovery time of their symbionts is prolonged (Hughes et al., 2018a). For the corals that are able to recover from such events, they do not necessarily recover with the same resilience they had before the bleaching, leading to a fragilized ecosystem (Hughes et al., 2010). The Great Barrier Reef Marine Park (GBRMP) was created under the *Great Barrier Reef Marine Park Act* in 1975 and covers 344 400km$^2$ of the GBR. At the time, only 4,5% of the Park was under protected areas making it inadequate to protect its ecological diversity. Therefore, a rezoning plan was established in 2004 in order to configure a more large, comprehensive, adequate and representative network of marine protected areas. This network covers 33% of the GBRMP and corresponds to the current zoning plan (Fernandes et al., 2005). Despite these conservation efforts, coral cover is still continuing to diminish, begging the question of what more can be done to save the coral reef ecosystems?

Many scientists have focused on this issue and are trying to come up with alternatives to help preserve the coral reefs, and although progress is made in research, most of them have come to a common conclusion that further damage on coral reefs cannot be avoided if global action on climate change fails to limit to 1,5-2°C above the pre-industrial baseline (Donner et al., 2005; Hughes et al., 2017, 2018a; Wolff

et al., 2018). Thus, the best way to go about protecting the GBR today is to plan for its persistence in the future, and therefore focus conservation or restoration efforts on the reefs most fit to succeed. This cannot be done anymore by focusing solely on biodiversity attributes to integrate into systematic conservation planning. A more holistic view must be adopted, favouring connectivity-driven processes (i.e larval dispersal patterns) in order to boost the ecosystem's resilience (Beger et al., 2010; Kininmonth et al., 2011; Magris et al., 2014, 2016; Hock et al., 2017).

This thesis finds itself in that same line of thinking. While some authors have focused on how climate change may directly impact larval dispersal in order to predict future connectivity (Álvarez-Romero et al., 2018), this work does the opposite by looking into past connectivity in order to identify the larval pathways which always seem to withstand tough conditions. More specifically, this work extends itself along a timeline reflecting the change of coral cover on the GBR, from a pre-disturbance state where the conditions are considered ideal up to a current state affected by important bleaching events related to global warming. Coral cover further affects the connectivity between reefs since their reproductive output is proportional to their coral cover. The time events used to build this timeline are the mass bleaching events of 1998, 2002, 2016 and 2017 which are considered to have been the most important ones on the GBR (Baird and Marshall, 2002; Berkelmans et al., 2004; Osborne et al., 2011; De'ath et al., 2012; Hughes et al., 2017; Kerry and Hughes, 2017; Mellin et al., 2019). All these events are marked by spatial variations in coral cover levels. The idea is to try and assess, for every one of these periods, how the change in coral cover has affected the connectivity of corals and how this, in turn, affects the spatial distribution of conservation priorities. This will be done using "Marxan" (Ball et al., 2009), a decision support software used worldwide for spatial planning. Because the global reef crisis is above all a governance crisis (Hughes et al., 2010; Sheppard, 2014), using a decision support tool is important in order to convey a clear message which can help decision makers take on effective actions. Ultimately, beyond understanding the relationship between coral cover and connectivity, the aim is to see if some reefs -regardless of the intensity and spatial pattern of bleaching- consistently remain important connectivity hubs. The results from Marxan will allow us to highlight the reefs which are consistently selected for protected areas -if there are any- throughout the whole timeline. In other words, whatever the Great Barrier Reef may have to overcome in the future in terms of global warming, some reefs may prove to be sufficiently resilient for us to protect them due to their performance in the past. These reefs would be an insurance for the future and it would therefore be worth having conservation efforts redirected on them.

This thesis is divided into four chapters. The first one will introduce the study area and the problem surrounding it. The second chapter will develop the notion of coral cover and its assessment in the GBR after every major mass bleaching event, that is after the events of 1998, 2002, 2016 and 2017. The results from this second chapter will be used in the third chapter to weigh connectivity relative to coral cover on each reef. In this third chapter, we will explain in detail the concept of connectivity, why it is crucial in today's context and further discuss the results. The fourth and last chapter will introduce Marxan. We will integrate the results from the previous chapters into Marxan to identify priority areas on the GBR. Several

scenarios were implemented in order to evaluate the sensitivity of the results to a change in input data. We will then identify the reefs selected by Marxan throughout the timeline and further process the results to try and establish if there is any common ground throughout that timeline. A overview of the key steps involved in this thesis are provided in Fig. 1.

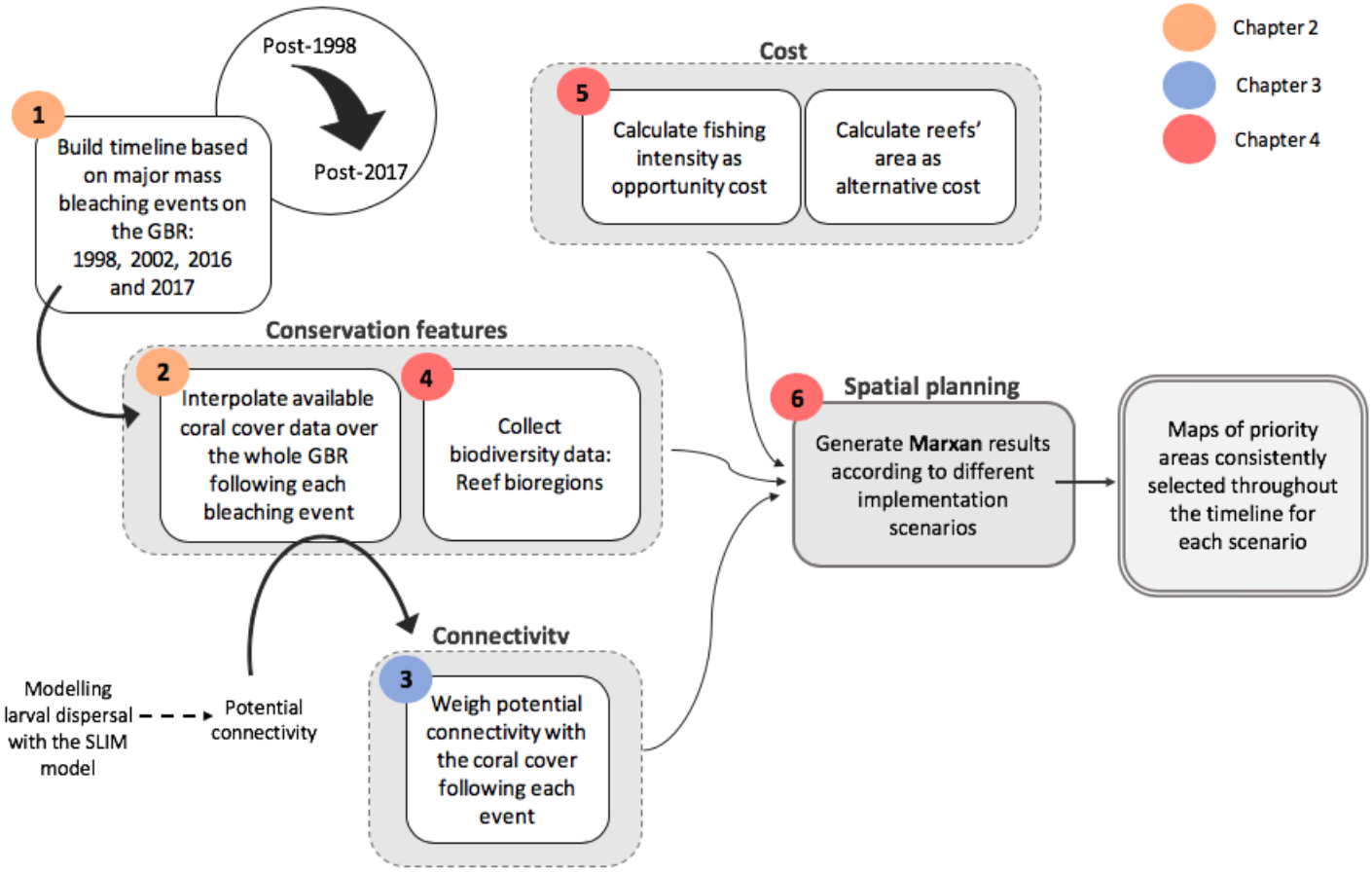


**Figure 1:** The general workflow of this thesis with a colour code indicating in which chapter each step will be implemented.

# Chapter 1

# Characterizing the study area: The Great Barrier Reef

The study area used in this work is the GBR, but this section could in fact apply to any other coral reef system of the world as the issues they face are universal. This section helps understand what is at stake, by going back to the start and acquiring knowledge on corals themselves and how they come about forming coral reefs, the stressors that threaten their stability, and finally the way forward endorsed by many scientists to tackle this issue.

## 1.1 Physical aspects

The GBR extends from 24°30'S in the south to 9°30'S in the north, covering a distance of about 2300km along the north-east coast of Australia, in the Coral Sea (Hopley et al., 2007). From the northern tip of Queensland to just north of Bundaberg, the Great Barrier Reef Marine Park (GBRMP) covers an area of 344 000 $km^2$, with a width between 60 and 250km (Fig. 1.1). This area, as large as 70 million football fields, makes the GBR the world's largest coral reef system. Ranging from shallow estuarine areas with an average depth of 35m, to 2000m deep oceanic waters, the GBR is home to a fascinating ecological diversity (GBRMPA, 2019c).

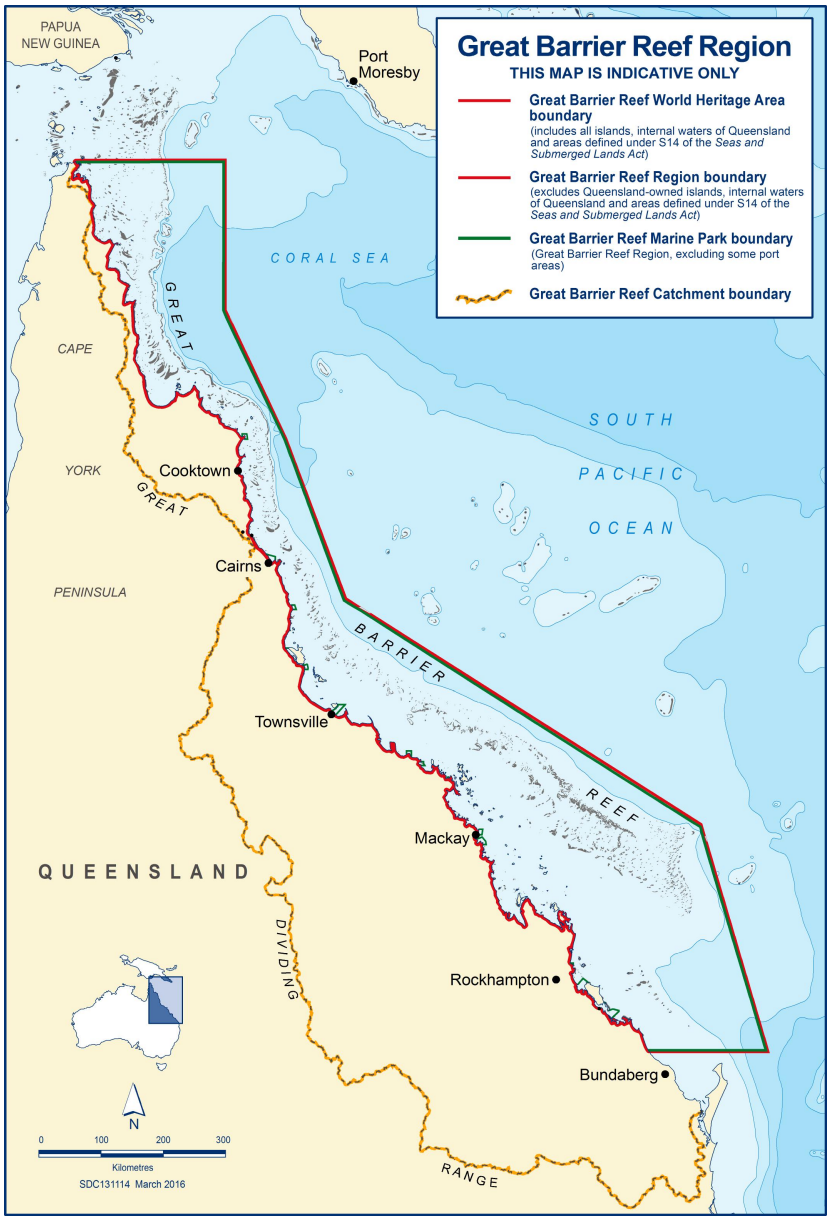


**Figure 1.1:** Map of the study region with the boundaries of the GBR Marine Park (GBRMPA, 2019c).

## 1.2 Socio-economic aspects

The GBR was the first coral reef ecosystem to be listed as World Heritage Area by UNESCO in 1981, given its 'outstanding universal value'. In May 2007, it was then placed on the National Heritage List, because of its contribution to national identity (GBRMPA, 2018g). It is strongly valued by Australians for it provides a coastal defence from natural hazards, a source of food on which depend fisheries to thrive, recreation and income (O'Mahony et al., 2017). It is safe to say that the GBR has now become an iconic global destination, attracting millions of tourists every year (GBRMPA, 2018g). Although tourism operators benefit from the reefs' extraordinary biodiversity, they also benefit the GBR by sensitizing people to the issues facing the coral reefs.

However, the GBR is more than just coral reefs and to prove this, the management consulting company *Deloitte Access Economics* has tried to quantify its value (O'Mahony et al., 2017). Since the GBR is already listed as one of the world's seven natural wonders, making it priceless, putting a value on it may seem unnecessary or controversial as it is not an object which one could try and trade with. Even though it is increasingly seen as a resource of incalculable value for current and future generations (Spalding et al., 2001), according to Deloitte, however, putting a value on something as pristine as the GBR will allow to understand its full contribution to the economy and society. The aim is to ensure the GBR is considered at its true and rightful value during decision making and to better inform policy settings in its regard. The report's findings revealed that the GBR has a socio-economic and icon asset value of $56 billion, it supports 64 000 jobs and contributes $6.4 billion to the Australian economy.
It is clear that the GBR is worshipped by Australians and people all around the world, and the need for its protection is raising increasing awareness as to what is at stake.

## 1.3 Corals

When one thinks of the GBR, corals is probably the first thing that comes to mind, and quite rightfully. Corals are the reef-building structures that shape the GBR and give it its palette of colors. In this section, information is provided on their general biology, their reproduction cycle and how they grow.

### 1.3.1 Distribution

Among the environmental conditions necessary for corals to thrive are warm temperatures (optimal between 23°and 29°C, high salinity (32 to 42 parts per thousand -*ppt*) and sufficient water clarity to permit light penetration (NOAA, 2014). This restricts them to a narrow band of low latitudes covering tropical and semitropical waters (Fig.1.2). The diversity diminishes as we move away from these latitudinal clines (Spalding et al., 2001).

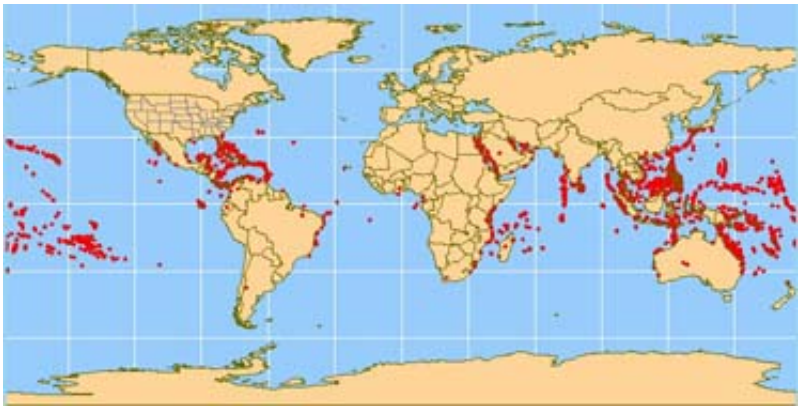

**Figure 1.2:** Worldwide distribution of corals located between 30°north and 30°south latitudes (NOAA, 2014)

### 1.3.2 General biology

Corals, which are often mistaken with plants, are actually invertebrate animals that belong to the class *Anthozoa*, the largest class within the phylum *Cnidaria* (NOAA, 2014), the same group as jellyfish and sea anemones. Corals consist of small individual animals called polyps. They have a very small cylindrical body with tentacles at the top enabling them to capture food from the surrounding waters (Spalding et al., 2001). Polyps associate each other to form colonies of hundreds to thousands of individuals, thereby forming corals (ICRI, 2018). Most corals are involved in a symbiotic relationship with single-cell dinoflagellates called zooxanthellae. Thanks to their photosynthetic ability, they provide their host with most of its nutrition requirements and in return the host provides them with metabolic waste products (CO2, inorganic nutrients) and a sheltered environment (Spalding et al., 2001; NOAA, 2014). It is also this symbiosis with zooxanthellae that gives most corals their vibrant colors.

There are 600 different types of corals in the GBR divided into two main categories: hard corals and soft corals. Hard corals act as reef builders because of the hard limestone skeleton they produce (GBRMPA, 2018d). Soft corals on the other hand are flexible due to their lack of a calcareous skeleton. Instead, they use spike resembling structures called *spicules* for support (ICRI, 2018). Soft corals also tend to be more brightly coloured and so, one is more likely to contemplate bright pinks and mauves on a soft coral than it is on a hard one, as shown on Fig. 1.3 below.

**(a)** Soft coral of the genus *Dendronephthya* common in the Indo-Pacific

**(b)** The rigid surface of a brain coral (hard coral) of the genus *Platygyra*

**Figure 1.3:** Images of (a)a bright colorful soft coral and (b)a hard coral (Spalding et al., 2001)

Corals also act as shelter for many species (Hoegh-Guldberg, 1999; GBRMPA, 2018g). But the welcoming features of soft corals don't always attract friendly guests, and they are often more vulnerable to predators such as fish, snails or crustaceans. Producing chemicals and displaying their spike-like spicules are their only defence mechanisms (GBRMPA, 2018d).

### 1.3.3 Reproduction

Corals can reproduce asexually and sexually. Asexual reproduction is most of the time achieved through "budding", where the young coral grows out from the adult polyp and participates to the colony growth. All asexual pathways induce the formation of clones (GBRMPA, 2018c).

However, most corals are hermaphrodites and thus undergo sexual reproduction (ICRI, 2018). The majority of species rely on broadcast spawning which involves massive amounts of eggs and sperm being released simultaneously into the water. Once these two meet, they form a larva called *planula*. Conversely, when the sperm fertilizes the egg within the polyp and the larvae are only released once relatively well developed, we then refer to it as "brooding". While brooded larvae may be immediately ready for settlement, spawned larvae may reside as plankton for up to several weeks, covering a broad geographic area. Once the planula has settled in a particular area, it develops into a new coral colony (NOAA, 2017; GBRMPA, 2018c). From this, it is clear that sexual reproduction is important to maintain genetic diversity and to have corals settle in new areas, establishing connectivity between separated reefs.

Sexual reproduction happens through a yearly synchronised spawning event. It occurs between October and December (depending on location) after a full moon and after water temperatures have increased sufficiently to stimulate the maturation of gametes. Resulting in clouds of larvae being released of many different colors, it is an impressive sight for the eyes (Fig. 1.4). Different species release their eggs and sperm on different days to avoid the formation of hybrid species (GBRMPA, 2018c). While the great majority does not survive this event due, in part, to predation, the remaining planulae participate in gene flow between distant reefs or in recovery of damaged reefs. Reefs located in areas with unfavourable conditions to their development may depend entirely on larval recruitment from better located reefs (Spalding et al., 2001).

**(a)** Spawning of an *Acropora* species (SECORE, 2016)

**(b)** *Fungia mushroom* coral spawning a cloud of sperm (Thomas, 2002)

**(c)** *Boulder star* coral releasing its gamete bundles (ReefResilienceNetwork, 2018)

**Figure 1.4:** Spawning among different coral species

### 1.3.4 From corals to coral reefs

Although corals themselves are very simple organisms, the coral reefs they form once they grow into colonies are not only among the most complex of all ecosystems, but in the case of the GBR, they are also the largest living structure on Earth (Spalding et al., 2001; GBRMPA, 2018g; ICRI, 2018). According to Spalding et al. (2001), coral reefs can be defined as shallow marine habitats, defined both by a physical structure and by the organisms found on them. Well-developed reefs can reflect up to millennia of history, which makes them all the more valuable and worth fighting for. Above all, they provide food and shelter on which an entire community of species depend.

As mentioned in subsection 1.3.2, hard corals are able to produce a limestone skeleton. They do this by extracting calcium from the surrounding waters and then using it to secrete calcium carbonates ($CaCO_3$) for protection and growth (ICRI, 2018). Massive reef structures are created when millions of tiny polyps secrete $CaCO_3$. However, not all corals that produce limestone are reef-builders, as some do not produce enough of it and others are solitary corals which would rather grow separately than to form a colony with other species (NOAA, 2014). The major contributors to reef growth are the *scleractinia* corals and coralline algae (Hopley et al., 2007).

The reef-building corals, also known as "hermatypic" corals, come in all shapes and sizes. The growth rate and patterns depend on the coral species, the environmental conditions, the geographic location and the density of surrounding species (NOAA, 2014). Ideal conditions for coral reefs growth include areas below the breaking line of waves, where light is abundant for photosynthesis, sedimentation is low, the salinity is sufficiently high (as of 31 ppt) and the temperatures range from 20°to 29°C (Sheppard et al., 2017). Still, even in ideal conditions, these hermatypic corals are slow growing with massive corals expanding their skeleton at slow rates of just a few millimetres per year. Branching corals on the other hand have much higher growth rates and can have their tips grow as much as 150 mm per year (Spalding et al., 2001).

## 1.4 Threats to coral reefs

No different from any other living organism, coral reefs are fragile and the GBR is no exception to that. Natural hazards have hit the reefs in the past and will hit again. This is not a bad thing in the sense that the occurrences of such events may actually contribute to strengthening the Reef on the long term. Indeed, albeit inflicting physical damage, these short-term events create a change in conditions which may give the opportunity for some new species to settle in, thus enhancing diversity on the reefs (Spalding et al., 2001). But the problem is reefs are not only facing natural hazards, they are also having to bear chronic human stressors. These slow drivers of change generate a constant background level of disturbance. And rather than allowing for some new species to settle and encourage diversity, these anthropogenic stressors cause some reefs to undergo a dramatic phase-shift of their coral assemblage, with reduced biodiversity (Hughes et al., 2010, 2005, 2018a). The

following subsections summarize the natural and anthropogenic threats that menace the stability of coral reefs and develop the concept of coral bleaching.

### 1.4.1 Natural

One of the major natural disturbances are extreme weather events such as tropical cyclones of category four and five. According to the GBRMPA (2019d), cyclones were responsible for 48% of coral loss recorded by the Australian Institute of Marine Science (AIMS)'s Long-Term Monitoring Program between 1985 and 2012. Between 2004 and 2018, 10 cyclones of category three or more hit the GBRMP, causing significant damage to coral habitats. Category five cyclones, such as the 2011 cyclone *Yasi* (Fig. 1.5), can cause significant damage to the reefs' structure, and can take centuries to fully recover (GBRMPA, 2019d). Although these extreme weather events are rare, they are predicted to increase with climate change (GBRMPA, 2019d).

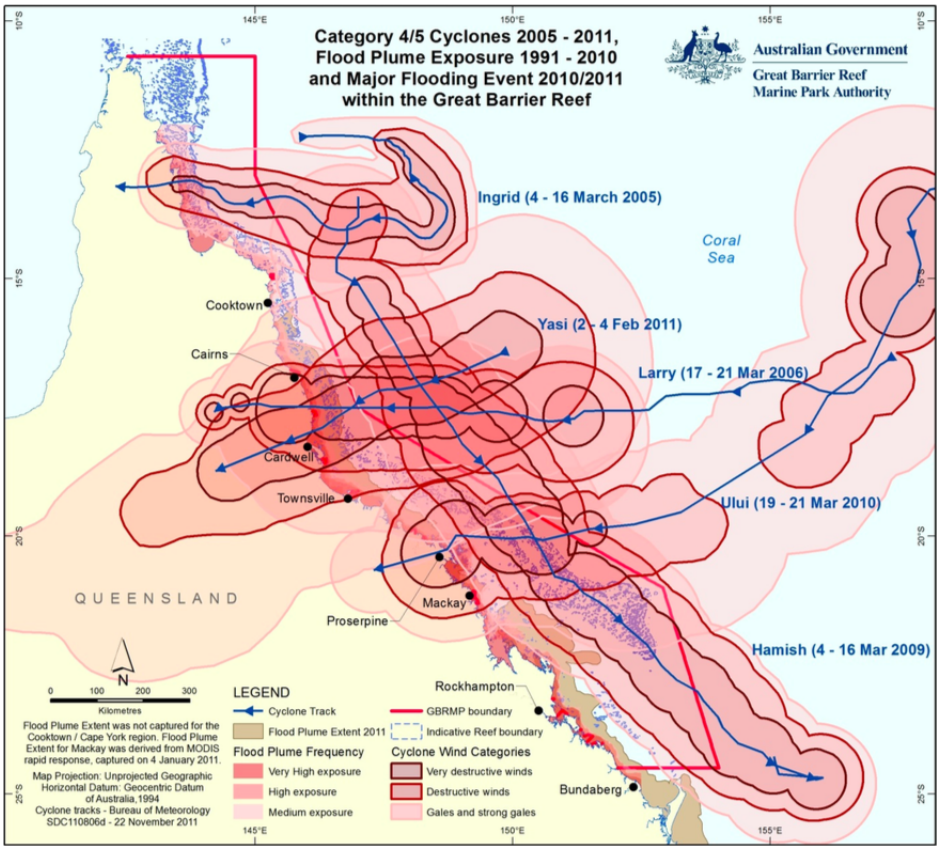


**Figure 1.5:** Extent of category 4 and 5 cyclones that hit the GBR between 2005 and 2013 (GBRMPA, 2013)

Weather events can also include major floods during wet seasons. The excess amount of freshwater decreases salinity and can lead to coral mortality, affecting inshore corals (GBRMPA, 2013). Intense floods have started to occur more frequently since the late 19th century with one every six years, compared to one every 20 years before then (GBRMPA, 2019d).

These weather disturbances lead to injured corals which can then become more vulnerable to coral diseases. Pathogens and parasites will take advantage of the coral's lowered immune system to thrive onto its damaged tissues. Once infested, there is little the coral can do to survive (Lamb et al., 2015). The most common diseases include the *white syndrome*, the *black band* and the *brown band* disease. The exact mechanisms behind these diseases are not yet fully understood (AIMS, 2018b).

Corals also have to face coral-eating predators, such as the Crown-of-Thorns-Starfish (COTS), *Acanthaster planci*, whose population outbreaks can decimate entire hard coral communities (AIMS, 2019b) (Fig. 1.6). These voracious creatures are difficult to manage because of their high reproductive potential, their high growth rates and their important outbreak densities. These outbreaks usually follow major flooding events that bring excessive amounts of nutrients from land to inshore waters (De'ath et al., 2012; GBRMPA, 2019a). But once breeding occurs within these outbreaks, the COTS larvae can then spread to other reefs which were not even affected by runoff in the first place (AIMS, 2019b). This is just one of the examples of the intrinsic complexity that lies within these coral reefs' ecosystem. There have been four massive waves of COTS occuring since the 1960's, each lasting approximately 10 years (AIMS, 2019b). The GBRMPA has established a COTS control program to try and mitigate outbreak damage, next to the cumulative damages from other threats, such as climate change, which leads us to the next subsection.

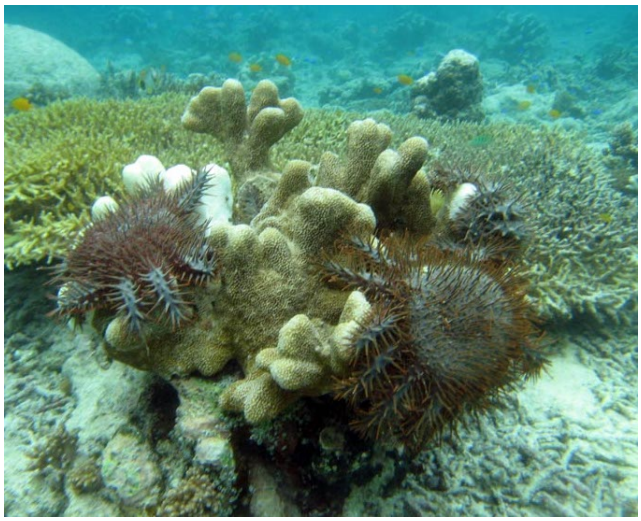

**Figure 1.6:** Image of two COTS feeding off a coral (ARC, 2018)

### 1.4.2 Anthropogenic

Major anthropogenic stressors include water pollution from terrestrial runoff, unsustainable fishing, coastal development, ocean acidification, and most of all rising temperatures due to global warming (Glynn, 1993; West and Salm, 2003; Levin and Lubchenco, 2008; Hughes et al., 2010; De'ath et al., 2012; Sheppard, 2014; Hughes et al., 2015). These human stressors not only add on to the natural pressures, but they also tend to amplify them. Therefore making a distinction between natural and anthropogenic stressors might be a bit radical, because they can be closely related

to one another (De'ath et al., 2012).

Intensive agriculture causes excessive amounts of nutrients, sediments and chemicals such as fertilisers and pesticides to be released in the water during flood plumes. This leads to reduced salinity and increased turbidity which is harmful to corals and the species surrounding them (GBRMPA, 2018e). Excessive nutrients in the water induces eutrophication, benefiting organisms such as algae which will proliferate rapidly. *In fine*, a radical shift in composition occurs, from a coral-dominated system to an alternate degraded one dominated by macroalgae or other weedy species (Hughes et al., 2010). Despite there being many human-induced threats, the biggest one of them all is climate change (GBRMPA, 2018a). From climate change follows a myriad of consequences: increase in sea surface temperatures (SST), more frequent occurrences of storms and cyclones, sea level rise, change in ocean currents, as well as unprecedentedly high levels of $CO_2$ emissions, which lead to ocean acidification (Bryan et al., 1998; GBRMPA, 2018a). All of which induce a level of stress unbearable for some coral species.

Among all of the above threats (natural and anthropogenic), tropical storms and COTS have been considered the most damaging ones (Osborne et al., 2011; De'ath et al., 2012). However, this statement might not be valid in the future where rising temperatures constantly challenge the reefs' ability to recover. In addition to that, even though cyclones and COTS constitute major threats, global warming acts as a constant background chronic stress, generating negative consequences for the reefs' resilience (Hughes et al., 2010).

### 1.4.3 Coral bleaching

When the conditions exceed a tolerable threshold, the symbiotic relationship between the zooxanthellae and the coral polyp ends as the zooxanthellae are expelled from the coral tissue. The coral looses all of its pigments, a phenomenon known as coral bleaching (Bryan et al., 1998) (Fig.1.7a). One of the most important environmental parameter for coral reefs is temperature (Berkelmans, 2002) and severe heatwaves can lead to mass bleaching events (Hughes et al., 2018a). Once bleached, corals are not necessarily dead but are at their most vulnerable state (NOAA, 2010). Corals that have bleached can still recover given sufficient time and minor, if not any, additional disturbances. A bleached coral will be white (Fig. 1.7a) whereas a dead coral will have a brown slimy aspect because it will be covered in algae (Fig. 1.7b). However, if temperatures are excessively high, it can literally burn the corals on the spot, and recovery is then no longer possible. In addition to this, the disturbance-free scenario for recovery is very unlikely nowadays, except for remote, pristine coral reefs far from human pressures. But however far from anthropogenic sources, no reef is safe from bleaching triggered by scorching temperatures. Confronted with such statements, scientists are continuously challenged to come up with alternatives before it is too late.

**(a)** A white field of bleached corals in the GBR (Sharwood, 2017)

**(b)** Dead corals covered in algae in the GBR (Goni, 2016)

**Figure 1.7:** Images of (a)bleached corals and (b)dead corals.

## 1.5 Conservation actions: Marine Protected Areas

Given all the potential threats, whether natural, anthropogenic, local or global, reef managers have the responsibility to take action in order to protect coral reefs. Despite providing theories that help understand processes which maintain and affect reef ecosystems, marine scientists cannot act directly on processes which are out of their control, such as climate change. Nevertheless, in addition to influencing politicians to set more ambitious policies to help mitigate climate change, scientists can already mitigate the impacts of processes on which they do have some control. One way of achieving this is by designing marine protected areas (MPAs) through conservation planning strategies (Margules and Pressey, 2000; Fernandes et al., 2005; Pressey et al., 2007; Micheli et al., 2012; Lamb et al., 2015; Strain et al., 2018). Marine spatial planning can be defined as "a framework that informs the spatial distribution of marine activities to support current and future uses, and maintain delivery of ecosystem services to meet ecological, economic and social objectives" (Ban et al., 2013). More specifically, when using the term "MPAs" in this work, we will refer to areas prohibiting any extractive or destructive practices (Game et al., 2008) (also referred to as 'no-take' areas).

This concept was already well understood by the GBRMPA twenty years ago when they undertook a rezoning of the GBR (Fernandes et al., 2005), which began in 1999 and ended in 2004 with the current zoning plan as a result. The aim was to increase the extent of *no-take* areas to better protect biodiversity of the GBR. In 1975, when the GBRMP was first created, *no-take* areas constituted only 4,5% of the Park's area, and this number increased to 33% after the rezoning of 2004 (Fernandes et al., 2005). The current zoning is divided into eight different types of zones, with varying levels of restrictions. From the least to the most restrictive zone (GBRMPA, 2014) :

- **General Use Zone** (light blue): General use, some activities require a permit.
- **Commonwealth Islands Zone**: They are the only land component of the Great Barrier Reef Marine Park. They can be used or entered without permission for low impact (non-extractive) activities, including photography, filming, sound recording and limited educational programs subject to certain limitations.

- **Habitat Protection Zone** (dark blue): No trawling, some activities require permits.
- **Conservation Park Zone** (yellow): No trawling, limited crabbing and line fishing.
- **Buffer Zone** (olive green): No aquaculture, bait netting, crabbing, harvesting fishing, collecting, spearfishing, line fishing, netting and trawling. Trolling for pelagic fish is allowed.
- **Scientific Research Zone** (orange): Research areas primarily around scientific research facilities. Same as Buffer zone but with no trolling.
- **Marine National Park Zone** (Green): *no-take* area. The following are allowed: boating, diving, photography and limited impact research. Some other activities are allowed with permits.
- **Preservation Zone** (Pink): *no-go* area. No activities are allowed except research activities with a permit.

As we can see on Fig.1.8, the major zones are the General Use, Marine National Park, and Habitat Protection zones.

Zoning has proven itself effective in increasing fish numbers and size, such as with the *coral trout* fish species whose numbers have increased of about 50% on offshore reefs and with larger average size (AIMS, 2018c). This increase in the amount of fishes, especially herbivorous, means higher grazing intensity, leading to a decrease in macro-algae surrounding the corals and thus less competition for the latter over space and light, which enables the establishment of more coral recruits (Hughes et al., 2010; Strain et al., 2018). Some of these herbivorous fishes (the bioeroders) also remove dead corals, freeing place for new corals to settle (Bellwood et al., 2004). More fish in protected areas usually means more fish in other zones as well due to the spillover effect. This replenishment of fish stocks is crucial to maintaining ecosystem structure as well as sustaining the fisheries which rely on the reefs for their activities (Micheli et al., 2012; Strain et al., 2018). As well as replenishing fish stocks, MPAs play a role in disease mitigation (Lamb et al., 2015), provide refuge for endangered species, protect spawning areas or nursery grounds, minimise damage to important habitats and most importantly, they help strengthen the resilience of the reefs against threats such as climate change (Hughes et al., 2010; Mumby et al., 2011; Micheli et al., 2012; Hughes et al., 2017; Álvarez-Romero et al., 2018).

The GBR is considered the best managed coral reef ecosystem in the world (Bruno and Selig, 2007), and despite promoting the persistence of some functional groups of corals (Strain et al., 2018), its coral cover is declining nevertheless, which leads to thinking more must be done to increase the reefs' resilience. Many scientists working on the matter agree that enhancing important connectivity patterns when designing MPAs is one way to go about it (West and Salm, 2003; Pressey et al., 2007; McCook et al., 2009; Botsford et al., 2009; Almany et al., 2009; Jones et al., 2009; Beger et al., 2010; Mumby et al., 2011; Kininmonth et al., 2011; Magris et al.,

2014; White et al., 2014; Beger et al., 2015; Schill et al., 2015; Magris et al., 2016; Hock et al., 2017; Krueck et al., 2017; Magris et al., 2017, 2018).

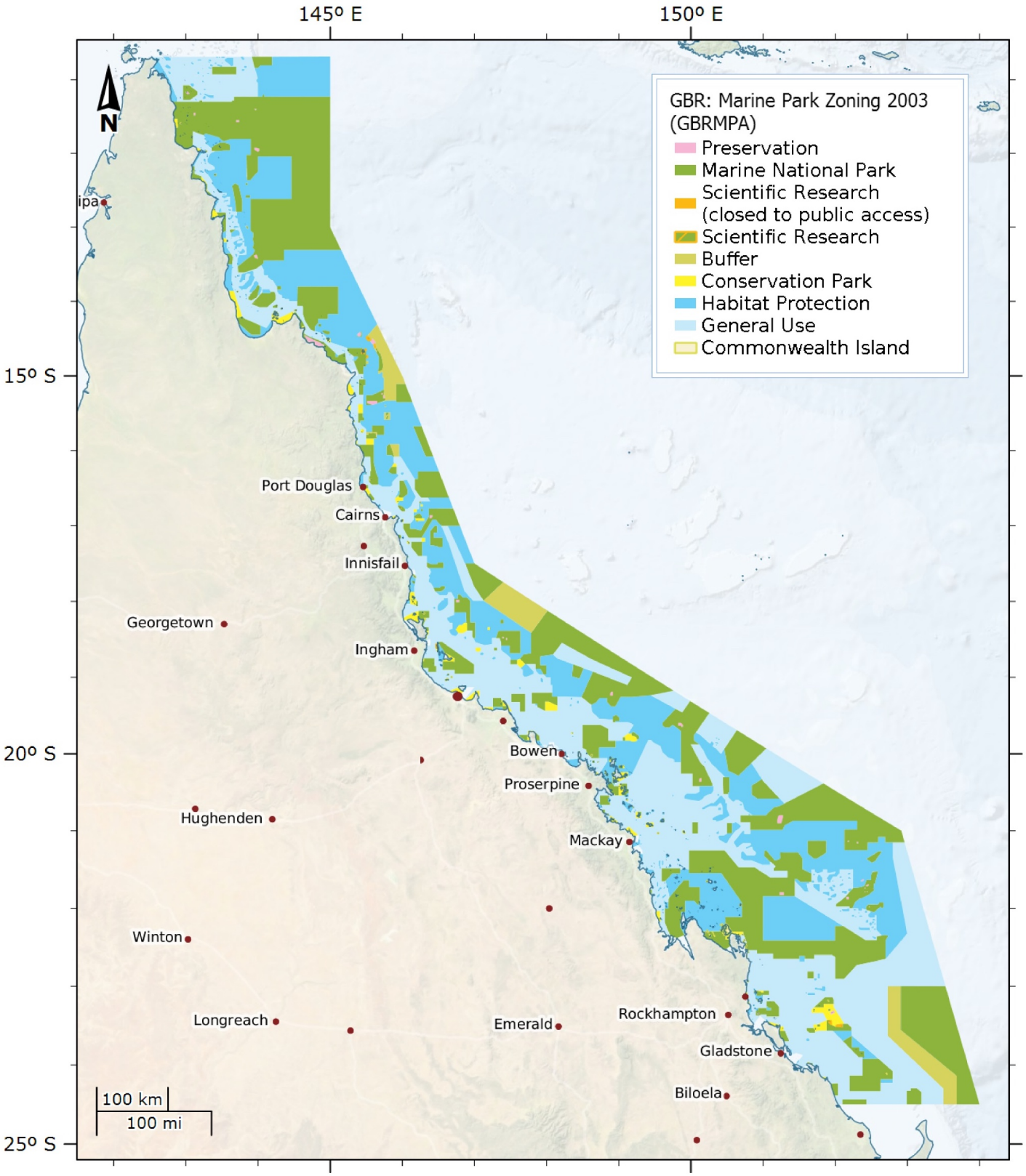


**Figure 1.8:** Great Barrier Reef Marine Park zoning (GBRMPA, 2014)

# Chapter 2

# Assessing coral cover over time

"In ecological studies of reefs, we aspire to understand past and future processes and events sufficiently well that we can make sensible predictions about the state of unseen reefs, now and in the future." (Wooldridge and Done, 2004). While the previous chapter led to understand why coral reefs are threatened and undergo a loss of coral cover, this chapter will now focus on depicting the spatial and temporal variations in that loss of coral cover in the GBR. The timeline used to study the temporal variations in coral cover is based on the four most important bleaching events recorded to have hit the GBR. These four bleaching events occurred in the years of 1998, 2002, 2016 and 2017. The aim of this section is to produce, for each year, a map of the coral cover's spatial patterns across the GBR. Once the coral cover has been assessed for each of those year cases, that information will be used in the next chapter to evaluate how connectivity is affected in turn.

## 2.1 Understanding the coral cover indicator

### 2.1.1 Reef health indicator

Hard coral cover is the single most reported measure of coral reef health and is commonly used by scientists to inform on management and government policies (Bellwood et al., 2004; Bruno and Selig, 2007; Osborne et al., 2011; Sweatman et al., 2011; De'ath et al., 2012; BIP, 2016). Thus besides just answering questions on the health status of reefs, it can further help answer questions such as "what impacts has had a specific management intervention or a new type of policy on the ecosystem?" (BIP, 2016). This indicator measures the percentage of reef surface covered by live hard corals, main contributors to the reefs' structure, and is therefore key in distinguishing a coral community from another taxa (e.g algae) dominated one (Healthy Reefs, 2018; BIP, 2016). When a coral reef shifts away from coral domination, it is usually a sign of ecosystem stress. When referring to a "healthy" reef, one generally implies relatively high percentage coral cover and low percentage of fleshy macroalgae. (Healthy Reefs, 2018).

However, when using coral cover as a health indicator one must keep in mind the resilience of reefs and this aspect is not communicated by this indicator. Indeed, as explained by Hughes et al. (2010), a healthy reef that is recovering towards a coral-dominated phase can have a much lower percentage coral cover than a reef

that is slowly degrading towards a macroalgae-dominance phase. Thus, in order for this indicator to integrate the resilience of reefs, the sites where coral cover is monitored should ideally be the same ones over the years.

That consistent monitoring method was adopted by De'ath et al. (2012) in their attempt to explain the decline in coral cover that has occurred on the GBR, specifically between 1985 and 2012. Throughout this 27-year period, coral cover on individual reefs varied between 1,50 and 79,7% spatially, with the highest values in the far northern and southern part of the GBR. Over the entire period, coral cover averaged 22,9%. Although coral cover did increase on some reefs (32,2% of the total), this was not sufficient to counter-balance the loss that occurred on the other reefs, leading to an overall decline in coral cover, from an average of 28,0% in 1985 down to 13,8% in 2012 (De'ath et al., 2012). This corresponds to a loss of 50,7% of intial coral cover. Their results found that this loss was in most part due to tropical cyclones (48%), followed by COTS outbreaks (42%) and then coral bleaching (10%) (Fig.2.1). These disturbances occurred frequently and scarce were the reefs that remained impact-free (only 3 out of the 214 reefs monitored).

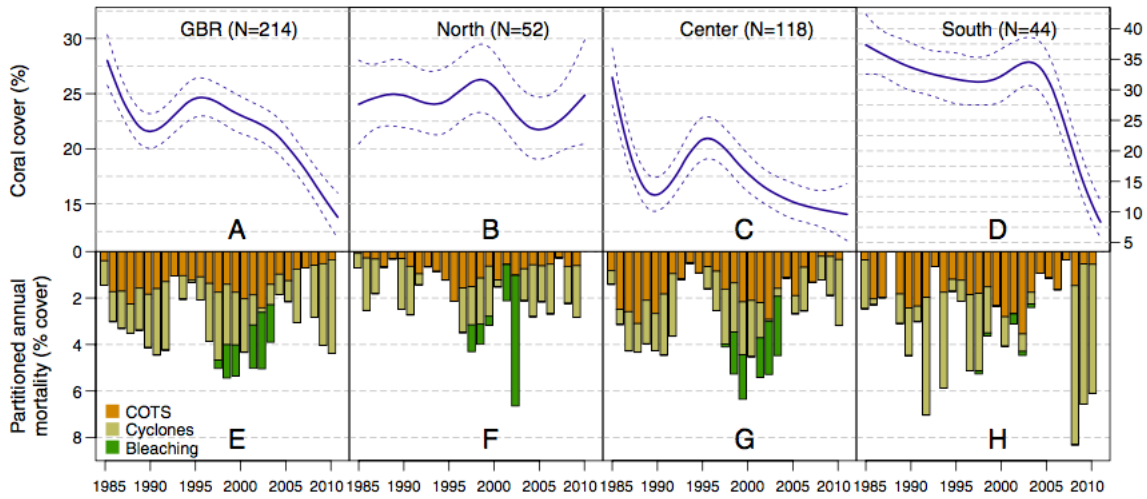


**Figure 2.1:** Temporal and regional trends in coral cover with associated mortality causes (De'ath et al., 2012)

### 2.1.2 Shifting baseline syndrome

This loss of 50,7% of initial coral cover is quite a gloomy statement and yet this 50% loss is deduced from the 1985 coral cover level, and by then, the GBR had already experienced important human impacts. In fact, surveys of coral reef ecosystems only began once significant damage had already been observed (Bruno and Selig, 2007; Knowlton and Jackson, 2008). So how can we assess the true extent of coral cover loss due to human disturbance without knowing exactly what to refer to for the pre-disturbance level ? This is problematic because without information on historical conditions of coral cover, it is difficult to understand how reefs behaved in the absence of human impacts and, from there, fixing a target for reef recovery becomes relatively subjective. Management practices rely on these targets for conservation and as the level of ecosystem health decreases over the years, targets also tend to

follow that trend. This issue is referred to as the "shifting baseline syndrome" (SBS) and has been pointed out by many scientists implicated in marine research (Sheppard, 1995; Bellwood et al., 2004; Bruno and Selig, 2007; Knowlton and Jackson, 2008; Hughes et al., 2011; Sweatman et al., 2011; Sheppard, 2014).

Far back in history (100–1000 y.b.p), average coral cover in the Indo-Pacific was probably about 50% (Bruno and Selig, 2007) but such statements are always characterized with uncertainty. While some argue that the decline in coral cover since historical levels has been overestimated due to changes in scale and type of surveying methods used (Sweatman et al., 2011), others, on the other hand, assert it has been underestimated (Bruno and Selig, 2007) and use that preceding argument as a foolproof example of a shifting baseline (Hughes et al., 2011). Studies such as the ones conducted by Bellwood et al. (2004) or Bruno and Selig (2007) illustrate that coral cover has indeed declined since at least the 1960s by an annual rate of 0.3-0.4% (Hughes et al., 2011). At the time when Bruno and Selig (2007) issued their study, they stated that "even though we may never know the precise Indo-Pacific coral cover baseline, we do know that regionally, cover is at least 20% below the best historical reference points." This statement is from 2007 which only leaves us with low expectations as to how far below these reference points coral cover has currently descended to.

## 2.2 Establishing the periods of interest: mass bleaching events

As explained in section 1.4.3, bleaching is referred to as the phenomenon by which corals become white and eventually die if no recovery is made. This can manifest itself at local scales such as parts of reefs, or at larger scales affecting entire reef systems. In the latter case, we refer to it as "mass bleaching" (Hoegh-Guldberg, 1999). Although bleaching of corals can be caused by different factors such as poor water quality, low salinity or little exposure to light, only heat stress, generated by abnormally high sea-surface temperatures (SSTs), can lead to important mass bleaching events (GBRMPA, 2017a), with a clear link established between increased temperatures and geographic footprint of bleaching (Osborne et al., 2011; Hughes et al., 2018a).

The worst temperature related mass bleaching events reported to have occurred on the GBR are the mass bleaching events of 1998, 2002, 2016 and 2017 (Baird and Marshall, 2002; Berkelmans et al., 2004; Osborne et al., 2011; De'ath et al., 2012; Hughes et al., 2017; Kerry and Hughes, 2017; Mellin et al., 2019). These events were therefore used to build the timeline along which changes in coral cover will be studied in this thesis. A short summary of each of these mass bleaching events will be presented in the following subsections, illustrating the spatial extent and intensity of each of them. An assessment of the coral cover following each of those bleaching events will then be presented in the following section.

When monitoring these bleaching events, there is a small window of action in order to capture the extent of the event as accurately as possible. This is due to

the fact that if the surveys are done too soon in the early stage of the bleaching event, some reefs may not have bleached yet and conversely, if the surveys are done too late, some reefs may have died already, covered in brown algae, making them unnoticed to the eyes of the monitors on the lookout for white, bleached corals. In both cases, the results will largely underestimate the true extent of the event even though, overall, the aerial surveys generally do tend to underestimate the intensity and extent of the bleaching (Berkelmans et al., 2004).

### 2.2.1 1998 bleaching event

In the summer of 1998, the GBR experienced its 6th recorded mass coral bleaching event since 1980 which was the first most intensive and extensive one ever recorded until then (Berkelmans and Oliver, 1999). Indeed, the GBR had already experienced three bleaching events in the 1980's and two in the early 1990's, however, due to the lack of reporting and documentation covering those events, it is difficult to assess the spatial extent or intensity of these bleaching episodes (Berkelmans and Oliver, 1999; Berkelmans et al., 2004). The 1998 bleaching began in late January and intensified by late February/March (Berkelmans and Oliver, 1999; AIMS, 2018a). Extensive aerial surveys of 654 reefs conducted at an altitude of 160m (Berkelmans and Oliver, 1999) revealed that $\sim$ 42% of reefs had bleached to some extent and 18% had strongly bleached (Berkelmans et al., 2004). The intensity of bleaching was more important on inshore reefs with 74% of them showing moderate to high levels of bleaching against 'only' 21% for the offshore reefs (AIMS, 2018a). Furthermore, these surveys showed that 25% of inshore reefs had extreme levels of bleaching (>60% of coral cover bleached) while none of the offshore reefs showed such levels of bleaching (Berkelmans and Oliver, 1999).

Results from the aerial surveys were confronted with observations from ground surveys conducted on 23 reefs having endured varying levels of bleaching, and this showed that the results from the former are likely to underestimate the true extent of the bleaching event (Berkelmans and Oliver, 1999). However, the damage was not permanent and most reefs made a good recovery with a mortality limited to less than 5% of reefs (AIMS, 2018a). The extent and intensity of the 1998 bleaching event can be seen on Fig.2.2, with a close-up on the Palm Island area where the worst affected reefs were located (AIMS, 2018a).

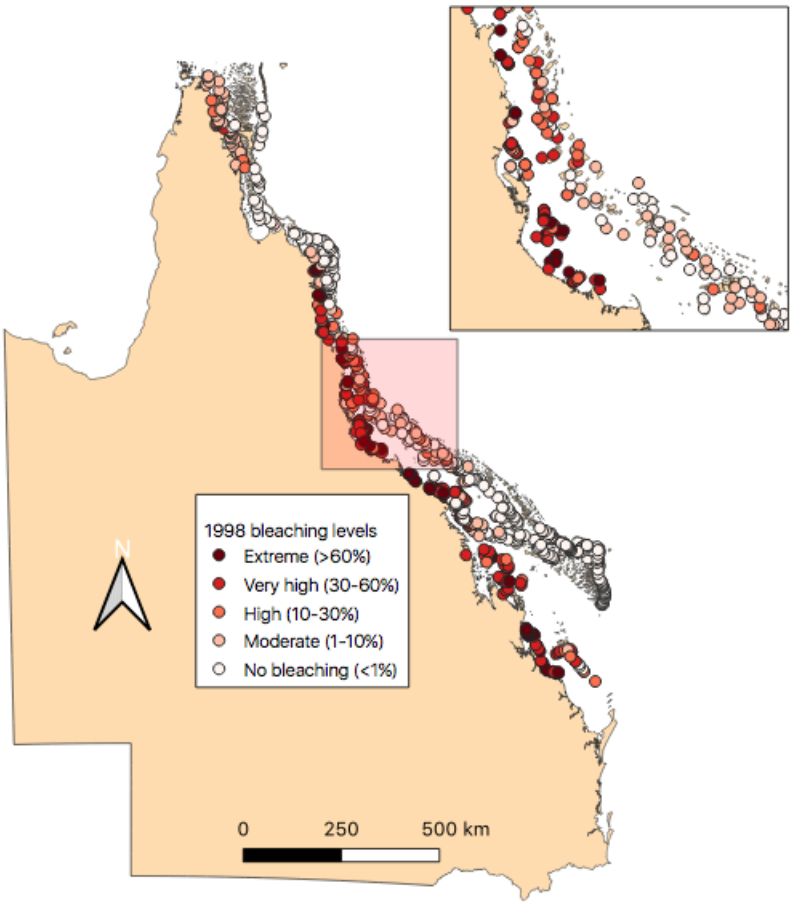


**Figure 2.2:** Extent and intensity of the 1998 mass bleaching event in the GBR, with a close-up on the worst affected area (Palm Island area)

### 2.2.2 2002 bleaching event

In the months of January and April 2002, another mass bleaching event took place in the GBR, more severe even than the one of 1998 (AIMS, 2018a), with 54% of reefs having bleached to some extent (against 42% in 1998) and 18% strongly bleached (Berkelmans et al., 2004; AIMS, 2018a). The same monitoring method was used as the one for the 1998 event, with surveys of 641 reefs conducted at a flying height of 160m. For logistic reasons, the same reefs were not necessarily surveyed in both years (Berkelmans et al., 2004).

This time, the offshore reefs were not spared as much as during the event of 1998. Indeed, although 72% of inshore reefs had moderate to high levels of bleaching, making it similar to 1998 (74%), the proportion of offshore reefs having experienced moderate to high bleaching had doubled from 21% in 1998 to 41% in 2002 (AIMS, 2018a; Berkelmans et al., 2004). Once again, most corals made a relatively good recovery with less than 5% mortality (AIMS, 2018a).

The extent and intensity of the 2002 bleaching event can be seen on Fig.2.3, with a close-up on the Bowen area where the most severely affected reefs were located (AIMS, 2018a). The 1998 and 2002 data of bleaching status of whole reefs

were made available by the Australian Institute of Marine Science (AIMS) on the "eAtlas" platform (AIMS, 2015).

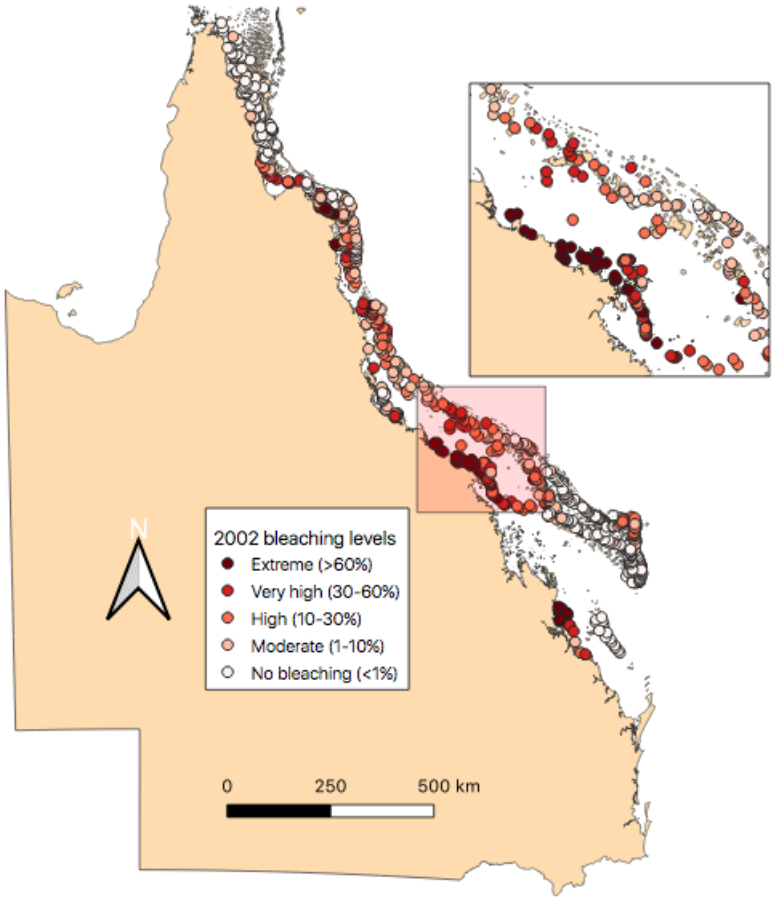


**Figure 2.3:** Extent and intensity of the 2002 mass bleaching event in the GBR, with a close-up on the worst affected area (Bowen area)

### 2.2.3 2016 bleaching event

The summer of 2015-2016 triggered record-breaking SSTs in tropical regions around the world, leading to widespread coral bleaching (GBRMPA, 2017a; Hughes et al., 2017, 2018b; AIMS, 2018a).This global heatwave was correlated with climate change induced ocean global warming as well as a strong El Niño event reducing the amounts of clouds or strong winds that could have helped mitigate the heat (GBRMPA, 2017a). The GBR suffered from highest temperatures on record between February and May (Fig. 2.4), and experienced its worst bleaching event ever recorded (GBRMPA, 2017a; AIMS, 2018a; Hughes et al., 2017, 2018b).

A similar protocol than that of 1998 and 2002 was adopted for the assessment of the bleaching. The monitoring of the coral bleaching took place in March-April 2016 when bleaching was at its highest peak, with extensive aerial surveys of 1156 reefs conducted at a flying altitude of 150m. Underwater surveys of 104 reefs were used to ground-truth the results of the aerial surveys, and a strong alignment was noticed between the two (Hughes et al., 2018b). The surveys indicated that less

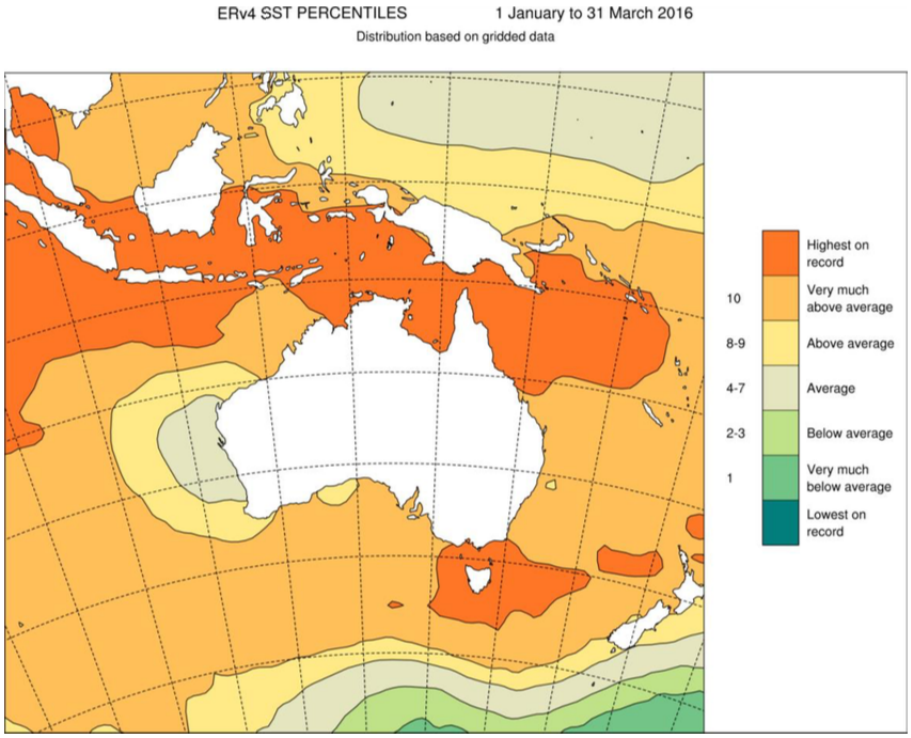


**Figure 2.4:** Distribution of record-breaking sea surface temperatures around Australia during the summer of 2016 (GBRMPA, 2017a)

than 9% of surveyed reefs had avoided bleaching and the proportion of reefs that were severely bleached (i.e. >60% of coral cover) was four times higher in 2016 than it was in the two previous events (Hughes et al., 2018b). The most extensive and intensive damage occurred in the northern part of the GBR (AIMS, 2018a). The extent and intensity of the 2016 bleaching event can be seen on Fig.2.5.

Whereas the death toll remained below 5% in 1998 and 2002, it increased to 29% for the entire GBR in 2016 (GBRMPA, 2017c). The mortality decreased as the surveys moved southwards with 67% mortality in the northern region, considered pristine until then, 6% in the central region and 1% mortality only in the south (ARC Centre of Excellence for Coral Reef Studies, 2016). The bleaching also varied longitudinally with 47% mortality occurring on mid-shelf reefs, 32% on inshore reefs and 21% on offshore reefs. The 2016 bleaching data was made available by Hughes et al. (2017) in their Supporting Information section (Hughes et al., 2017).

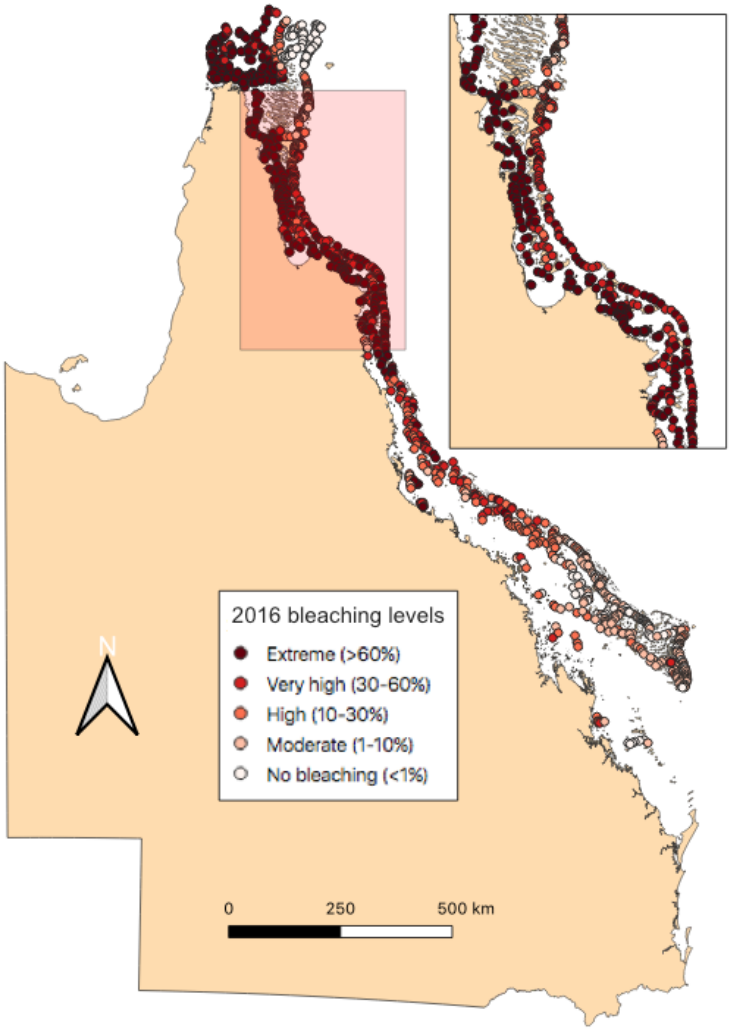


**Figure 2.5:** Extent and intensity of the 2016 mass bleaching event in the GBR, with a close-up on the northern tip which was the worst affected area

### 2.2.4 2017 bleaching event

The 2017 bleaching event marks another terrible hit in the GBR's bleaching history. This event was particularly damaging because it took place just the following year of the 2016 event, leaving very short time for corals to recover from the past bleaching event (Kerry and Hughes, 2017; GBRMPA, 2017a). Aerial surveys were conducted to assess the damage and indicated that most of the bleaching had occurred between Cooktown and Townsville, in the central region of the GBR (GBRMPA, 2017b). This means that when combining the damage of the two consecutive years, two-thirds of the GBR have undergone severe bleaching (Kerry and Hughes, 2017; GBRMPA, 2017b) (Fig.2.6).

The unusually high SSTs of 2016 were triggered by a strong El Niño event. However, this year, the above-average temperatures could not be explained by such event implying global warming is the culprit behind such elevated SSTs. This situation already occurred in 2002 where the abnormally high SSTs were not related to an El Niño event either (Kerry and Hughes, 2017).

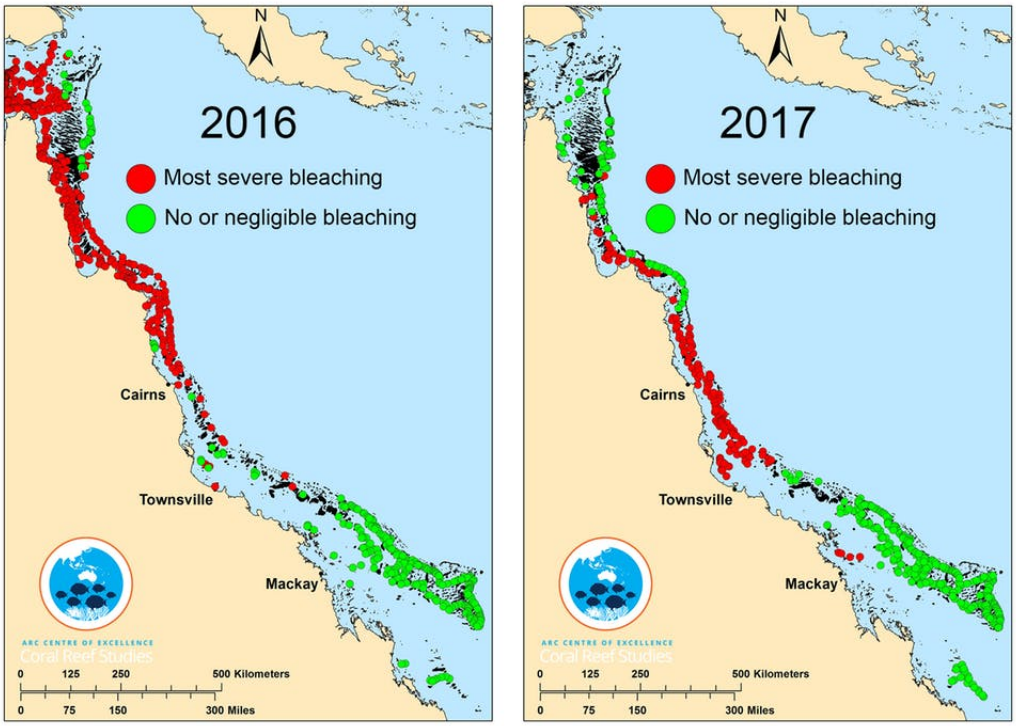


**Figure 2.6:** Extent of the 2017 bleaching event compared to the event of 2016 (Kerry and Hughes, 2017)

## 2.3 Finding the best available data: AIMS monitoring program

After having established the specific periods of interest from the bleaching events, the next step involved finding the best available data to use as a reference point for coral cover assessment. For this, we used the data from the Australian Institute of Marine Science (AIMS). This institute was established in 1972 and has since then been providing important research to help guide decision-making related to the management of Australia's marine estate (AIMS, 2000). Their comprehensive and impartial datasets have already been used by many scientists in different studies (Berkelmans et al., 2004; Wooldridge and Done, 2004; De'ath et al., 2012; Mellin et al., 2019), and their large-scale as well as long-term aspects make it ideal for this particular case study. This data was kindly provided to us by the AIMS Long-Term Monitoring Program (LTMP) on personal request. Two different methods are used in this program to assess coral cover: broadscale manta tow surveys and intensive photographic surveys (Fig. 2.7).

In the context of this work, the coral cover data used was the one assessed through the broadscale manta tow suveys. AIMS began these broadscale surveys in the 1980s with as first objective to monitor COTS populations. These surveys cover eleven sectors of the GBR, from Cape Grenville to Capricorn Bunkers (Fig. 2.8).

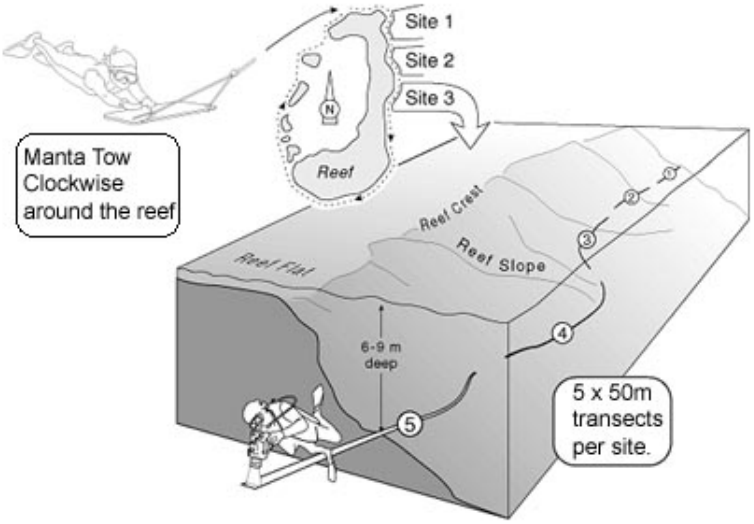


**Figure 2.7:** AIMS manta tow and intensive photographic surveying techniques (AIMS, 2018e)

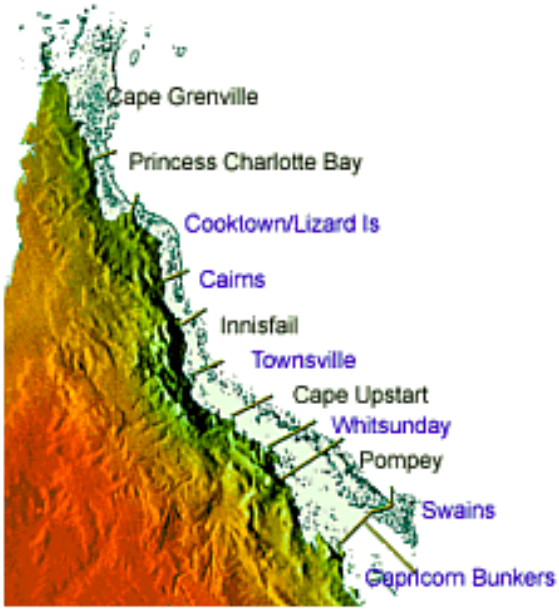


**Figure 2.8:** Sectors covered by AIMS for broadscale and intensive photographic surveys (in blue) and sectors covered only by broadscale surveys (in black) (AIMS, 2018e)

With the manta tow technique, a snorkel diver holds on to a manta board attached to a small boat and is towed around the entire perimeter of each sampled reef at a constant speed. Every two-minutes, the boat stops, allowing the snorkeller to report its results (among which, estimated coral cover) on a data sheet. The boat navigators follow "towpaths" marked on an aerial photograph of each reef in order to try and monitor the same parts of each reef during each survey. To be efficient, this survey technique requires at least 6m visibility underwater and because this often is not the case in inshore waters, inshore reefs are therefore poorly represented in this type of monitoring program (Sweatman et al., 2011). Still, this method has

the advantage of covering large areas rapidly and of requiring minimal equipment (AIMS, 2018e).

Because the manta tow method reports the mean coral cover by sampling the entire reef, which often include sandy bank reefs, it will have lower values than the intensive photographic method which occurs on smaller sections of the reefs and always in an area conducive to coral growth [G. Colemans, *pers. communication*]. Despite this shift, when comparing manta tow estimates with intensive scuba surveys undertaken by the GBRMPA and estimates from video footage by Reefwatch Australia, the resulting level of agreement suggested that broadscale surveys estimated coral cover with sufficient accuracy as to be able to identify moderate to large-scale disturbances and recovery (AIMS, 2018e).

## 2.4 Interpolating the data over the whole GBR

For obvious reasons, AIMS did not monitor all existing reefs in the GBR and an interpolation was thus necessary to obtain coral cover information for each reef. Because of the narrow shape of the GBR, spatial interpolation methods were quite limited and the Inverse Distance Weighting (IDW) method was finally considered most adequate. This technique computes an average value for unknown points from nearby weighted points. This weighting is proportional to the distance to the unknown point, assuming closer values to the unknown point exert more influence than further locations (QGIS, 2019). This spatial influence makes sense in our context since reefs tend to bleach (or not) in spatial clusters (Berkelmans et al., 2004; Osborne et al., 2011). This level of influence can be parametrised through the IDW power coefficient. A "leave-one-out validation" was performed to fine tune this power coefficient and its value was set for the lowest Root Mean Square Error (RMSE) computed (RMSE=9,23%; 10,93%; 11,42%; 10,58% for the baseline, post-1998, post-2002 and pre-2016 events respectively).

### 2.4.1 Approximations

Several approximations were made in order to produce the results developed in section 2.4.2.

For the post-1998 and post-2002 events, the assessment method involved IDW interpolation. Seeing that IDW produces best results when sampling is sufficiently dense and points relatively even (QGIS, 2019), a first approximation was made in order to include more data points as input for the interpolation, making it more reliable. This involved including data from previous years to the dataset of the current year. More specifically, the methodology to construct the input points was as follows: for a specific year (i.e 1998), when post-bleaching data was available (i.e the data from 1999), it would be used as input points and then, to acquire additional input data, coral cover values from before the bleaching event were used by targeting only the ones corresponding to reefs that were not located within the zones impacted by bleaching. The difference in the amount of data points available

can be seen on Fig. 2.9. With this method, the amount of sampled reefs reached 270 for 1998 and 220 for 2002.

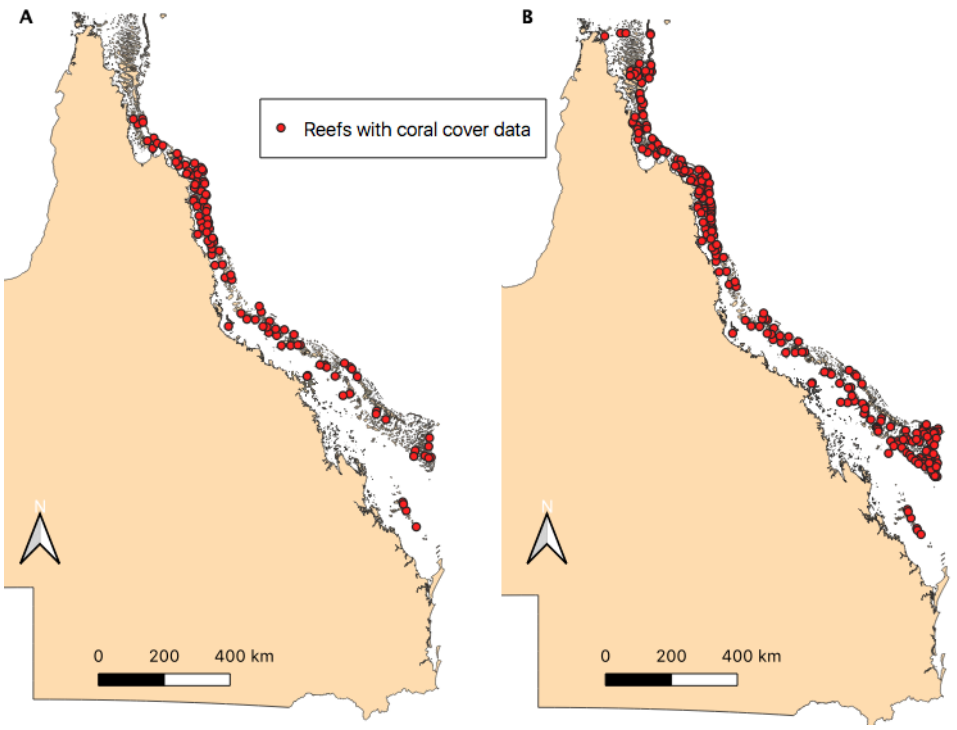


**Figure 2.9:** Comparison of the amount of data points for the 1998 event when **(A)** only post-1998 data is used, **(B)** both relevant pre-1998 data and post-1998 data are used

This is a big approximation but it remains fairly reasonable considering that AIMS focuses on monitoring areas where disturbances have occurred such as COTS outbreaks, cyclones and bleaching events (AIMS, 2018d), which means that where no monitoring has been conducted, it is likely that no major disturbance has occurred and that coral cover values remain relatively similar to the last most recent monitored ones. However, this hypothesis thus also rules out the possibility of an increase in coral cover which is more than likely if conditions have been favourable to growth in areas where no bleaching has occurred. This must be kept in mind when analysing the results as these likely underestimate the true coral cover values.

Since corals are able to recover from disturbances and can reach estimated growth rates of 2,85% $y^{-1}$(De'ath et al., 2012) to 3,69% (Osborne et al., 2011) in the absence of disturbances, this makes recovery an important aspect of coral reef health. This is why the 2016 event was divided into two coral cover levels, one "post-2016" and another one "pre-2016" in order to reflect the potential recovery that has manifested itself over the 14 years separating the 2016 event from the 2002 event. Recovery was only considered between those two events and not between the 1998 and 2002 events because of the shorter timespan between them. For this reason, 2016 is the only year in the timeline to have "post-" as well as "pre-" coral cover values. To assess the pre-2016 coral cover, data points available from 2012 to end of 2015 were used. The reason for having began in 2012 and not solely in 2015 is because using the year 2015 alone would not have provided sufficient data and in addition to that, AIMS does not monitor the same reefs every year which means that, by covering

more years, a larger amount of monitored reefs is covered. The amount of sampled reefs used for the IDW interpolation was 170.

Due to the smaller amount of data available for the post-2016 and post-2017 events, and because of the very short timespan separating these two events, the assessment of their corresponding coral cover was done through a different method than that for the years preceding the 2016 bleaching event. An alternative was thus found through the figure provided by Hughes et al. (2018a), illustrating the loss of coral cover following the 2016 event (Fig. 2.10). This percentage of coral cover loss was thus deduced from the already established pre-2016 level to assess the post-2016 coral cover. Because such similar information was not available specifically for 2017 and because the data available for 2017 was so scarce, Fig. 2.10 was also used as a reference to deduce the amount of coral cover lost during that year. Thus, the same percentage of loss corresponding to the most severe bleached areas in 2016 was also used for the most severe bleached areas in 2017 (Fig. 2.6).

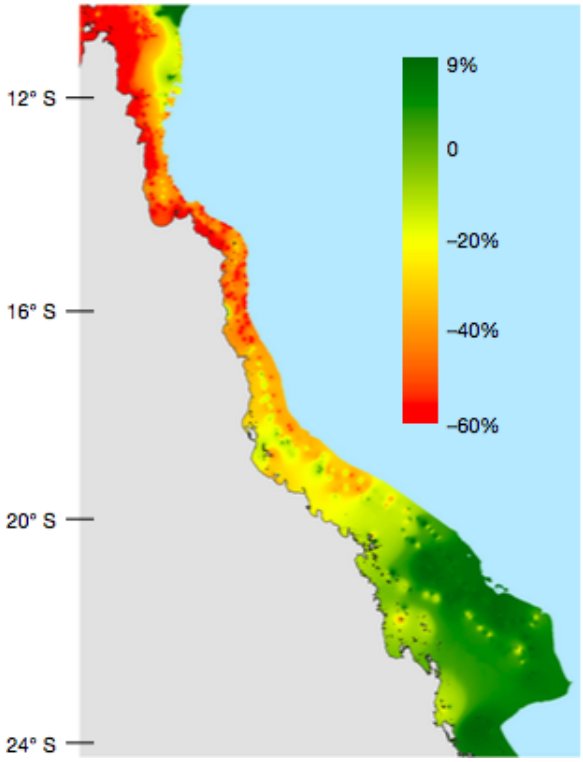


**Figure 2.10:** Change in coral cover on the GBR between March and November 2016 (Hughes et al., 2018a)

Coral cover decline tends to be lower on reefs experiencing low disturbance, and vice versa (Mellin et al., 2019). Yet, in the past -that is when human activities were not as developed as they are today (i.e pre-european settlement times)- coral reefs had fewer threats to face from anthropogenic sources, especially in terms of water quality (Bellwood et al., 2004; Hughes et al., 2011; De'ath et al., 2012). This lower disturbance level therefore implies coral cover was higher at the time than it is in this 21st century. It was therefore considered relevant to add a "pre-disturbance/baseline" coral cover level to the timeline. However, as explained in section 2.1.2 regarding the shifting baseline syndrome, no historical data on coral cover is available, making it difficult to assess this "undisturbed" coral cover state.

Although the dataset provided by AIMS goes as far back as 1985, data from that year could not be used as a reference point because its coral cover averaged 30%. Yet it is almost certain that historical coral cover was much higher than that (Bellwood et al., 2004; Bruno and Selig, 2007; Hughes et al., 2011; De'ath et al., 2012). An alternative thus had to be found if this baseline coral cover was to be added to the timeline in order to have a reference point to compare the different post-bleaching coral cover levels with.

The method that was finally adopted involved identifying, for each reef included in the dataset, its highest monitored coral cover value, whatever year it corresponded to. This approximation was made based on the fact that if a reef had high coral cover at some point between 1985 and 2019 (time scale of the AIMS LTMP), it was likely to have reached that same value at some point in the past. Naturally, these peaks in coral cover are unlikely to be the highest values these reefs have ever reached but they do give an idea of the potential a reef has in terms of coral cover. Also, even though maximum values were extracted, some of them remained relatively low and may still reflect important disturbances. To mitigate this drawback, maximum coral cover values below 30% were excluded since this was the average in the mid 80's according to the LTMP dataset and any coral cover below that amount thus most probably translates some sort of disturbance. With this method, 290 sampled reefs were used as input data for the interpolation.

Finally, although this is not an approximation *per se*, it is nonetheless also important to keep in mind the method involved behind the data acquisition. Indeed, because the data used in this work comes from broadscale manta tow surveys, as mentioned in section 2.3, the coral cover values won't be as high as if we had done the same work but with intensive photographic data.

The data manipulations described above were for the most part achieved using the QGIS geographic information system (QGIS, 2002).

### 2.4.2 Results and discussion

Once the approximations were set, coral cover for the different events was assessed. This was done using the open source RStudio software (RStudio Team, 2016). Only the coral cover for the baseline, 1998, 2002 and pre-2016 events were interpolated in RStudio seeing that the post-2016 and 2017 coral cover were assessed through a different method (see previous section).

Our results found the average coral cover on the whole GBR for the baseline scenario is 43,42% (Fig.2.11). This does not correspond to the 50% coral cover estimated by Bruno and Selig (2007) for the historical level, and is instead closer to the average 40% coral cover they estimated for the Indo-Pacific in the early 1980's. Also, specifically for the GBR, Bellwood et al. (2004) performed a meta-analysis of the literature to produce historical coral cover results. Their results are visible on Fig. 2.12. From that figure, Hughes et al. (2011) redraw a frequency distribution of coral cover prior to 1980 (Fig. 2.13). This shows the reefs had a modal amount of 41-50% coral cover during that time.

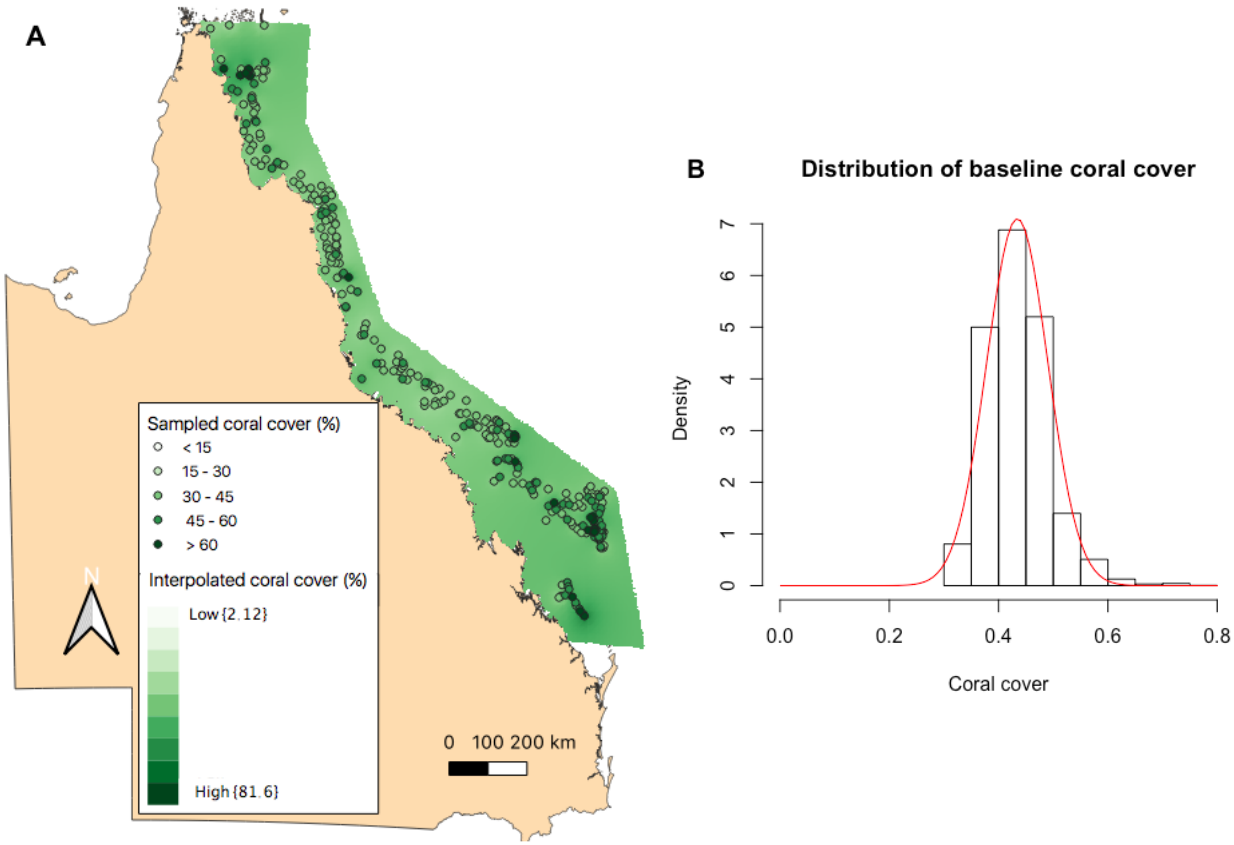


**Figure 2.11:** Results for the baseline scenario showing **(A)**Interpolated coral cover on the GBR with the dots representing the sampled points used as input for the interpolation; **(B)**Density plot showing the distribution of coral cover data

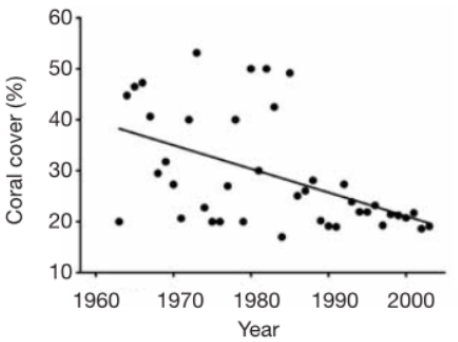


**Figure 2.12:** Results of coral cover based on a meta-analysis of the literature (Bellwood et al., 2004)

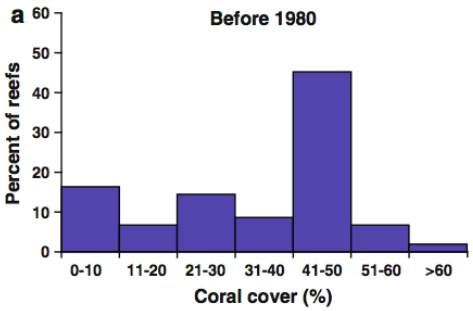


**Figure 2.13:** Frequency distributions of coral cover on the GBR in 1963-1980 based on a meta-analysis of the literature (Hughes et al., 2011)

When combining information from these different studies (Bellwood et al., 2004; Bruno and Selig, 2007; Hughes et al., 2011), it seems the results produced in this work for the baseline scenario should not be associated with the pre-European settlement but rather with the middle of the 20th century if they are to be relevant. This happens to coincide with the period just prior to when the fossil fuel industry started to grow rapidly in Queensland, which began in the 1970s (Hughes et al., 2015). Therefore, although this baseline scenario does not reflect an "anthropogenic

disturbance-free" level, it is still potentially representative of the level of coral cover that may have existed before unprecedented amounts of mining, greenhouse-gas emissions, shipping, port developments and dredging started to burden the GBR (Hughes et al., 2015).

It is however important to keep in mind that metadata analyses performed in the context of coral cover decline on the GBR have caused debate, with some scientists challenging the accuracy of such metadata analyses claiming their estimates are most likely overestimated (Sweatman et al., 2011). Even though there is indeed an unknown amount of variance associated with such type of analyses (Hughes et al., 2011), they nevertheless provide valuable insight into what coral cover could have been like in the past and that is already in itself a step forward.

Fig. 2.14 and Fig. 2.15 show the coral cover results for the post-1998 and post-2002 bleaching events respectively. The average coral cover estimated on the whole GBR for these two events is 31,01% and 27,76% respectively. A comparison with the literature was made to evaluate how relevant these results were.

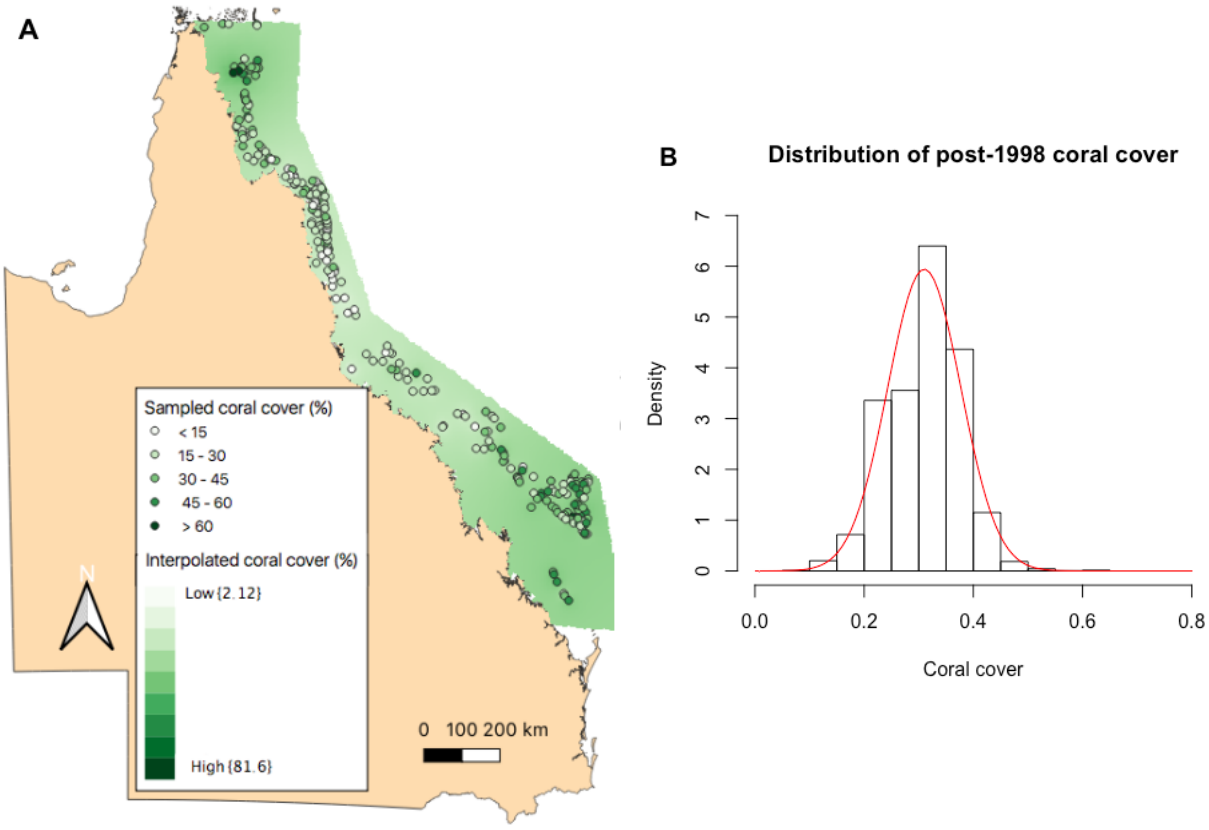


**Figure 2.14:** Results for the post-1998 event showing **(A)**Interpolated coral cover on the GBR with the dots representing the sampled points used as input for the interpolation; **(B)**Density plot showing the distribution of coral cover data

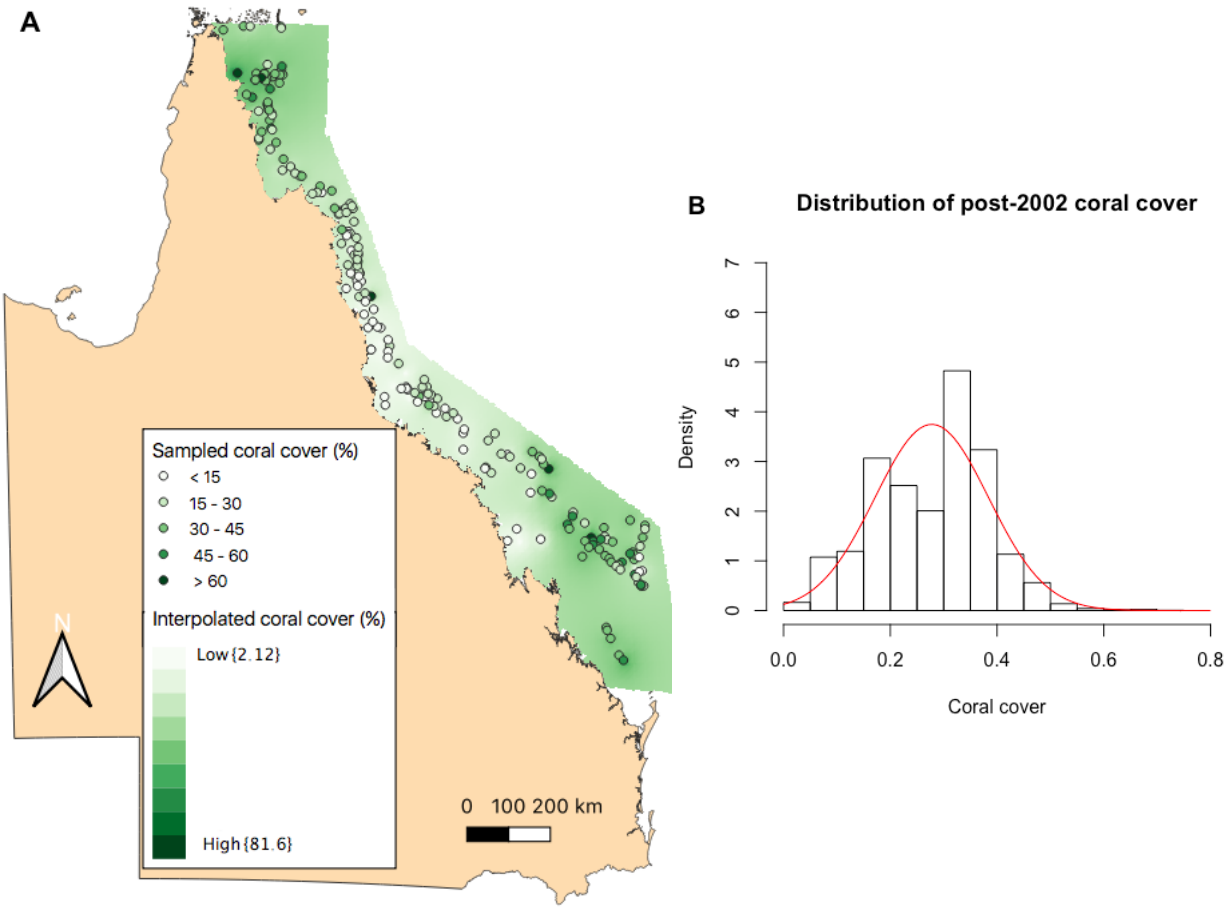


**Figure 2.15:** Results for the post-2002 event showing **(A)**Interpolated coral cover on the GBR with the dots representing the sampled points used as input for the interpolation; **(B)**Density plot showing the distribution of coral cover data

Mellin et al. (2019) studied the spatial resilience of the GBR under cumulative disturbances and showed there was a noticeable decline in coral cover (-0.75%/year) between 1996 and 2002 due to the 1998 and 2002 bleaching events as well as a major COTS outbreak. Their level of coral cover remained however fairly consistent between the post-1998 and post-2002 events with only a slight decrease (see red dotted lines on Fig.2.16).

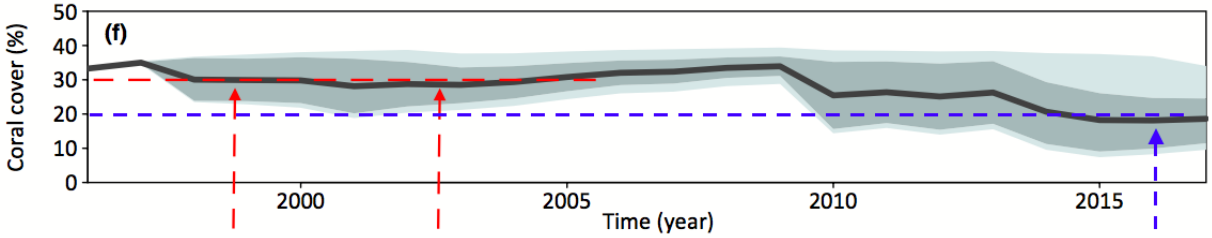


**Figure 2.16:** Mean predictions of coral cover averaged across the entire GBR (Mellin et al., 2019). The dotted lines were added to indicate the levels at which our coral cover results should be compared with.

Osborne et al. (2011) showed similar results in their study on the dynamics of coral cover on the GBR between 1995 and 2009. They established no net decline in coral cover had occurred in that period despite the presence of disturbances. They estimated an average 30% coral cover over that time frame, with a peak of 33% in 1999 and a low of 27% in 1995 (thus slightly contradicting the results of Mellin et al. (2019) which placed their peak prior to 1999).

This apparent overall stability, illustrated on Fig. 2.17, was the result of a balance between local increases and decreases of coral cover at subregional scales (Osborne et al., 2011). Seing that Osborne et al. (2011) contend that 27-33% cover is a meaningful and accurate representation of the average coral cover on the GBR since 1995, and that Mellin et al. (2019) show a similar estimate averaging 30%, our results thus seem fairly relevant.

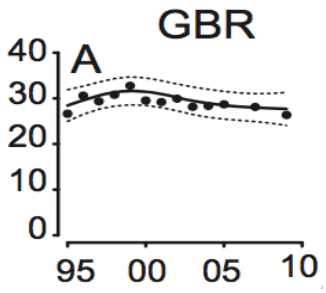


**Figure 2.17:** Average coral cover (%)(y-axis) for the whole GBR between 1995-2009 (x-axis) (Osborne et al., 2011)

However, Osborne et al. (2011) conducted their own sampling of reefs using a different method than that used by AIMS. Indeed, they sampled sites located on the flank of reefs which is where reef accretion is most active whereas AIMS included large areas of non-coral habitat using the mantatow technique (as explained in section 2.3). Mellin et al. (2019) on the other hand combined data from both methods. The results of Osborne et al. (2011) thus likely illustrate a 'best case scenario' and the results from AIMS a 'worst case scenario'. Based on AIMS monitoring method, the average coral cover on the whole GBR declined from 28 to 22% between 1986 and 2004 (Sweatman et al., 2011; De'ath et al., 2012). Because the data used in this work comes from AIMS, the results should be closer to the 28-22% coral cover. Instead, our results seem to be closest to the ones of Mellin et al. (2019). Nevertheless, because the data from AIMS likely underestimates the true extent of coral cover (cfr. section 2.3), our results being slightly superior to their estimates is thus perhaps not such a bad thing.

Fig. 2.18 shows the results for the coral cover preceding the 2016 bleaching event, illustrating the potential recovery that occurred since 2002. These results show an average coral cover on the whole GBR of 22,37%. Although this 'pre-bleaching' coral cover was considered in our timeline in order to highlight the recovery process of corals, instead of seeing an increase in average coral cover on the GBR, there is a noticeable decrease from 27,76% post-2002 to 22,37% pre-2016. Our timeline has been constructed based on mass bleaching events having hit the GBR, however, these are not the only threats to coral reef habitats and other disturbances such as cyclones or COTS predation could explain why the coral cover decreased between 2002 and pre-2016 even though no mass bleaching occurred.

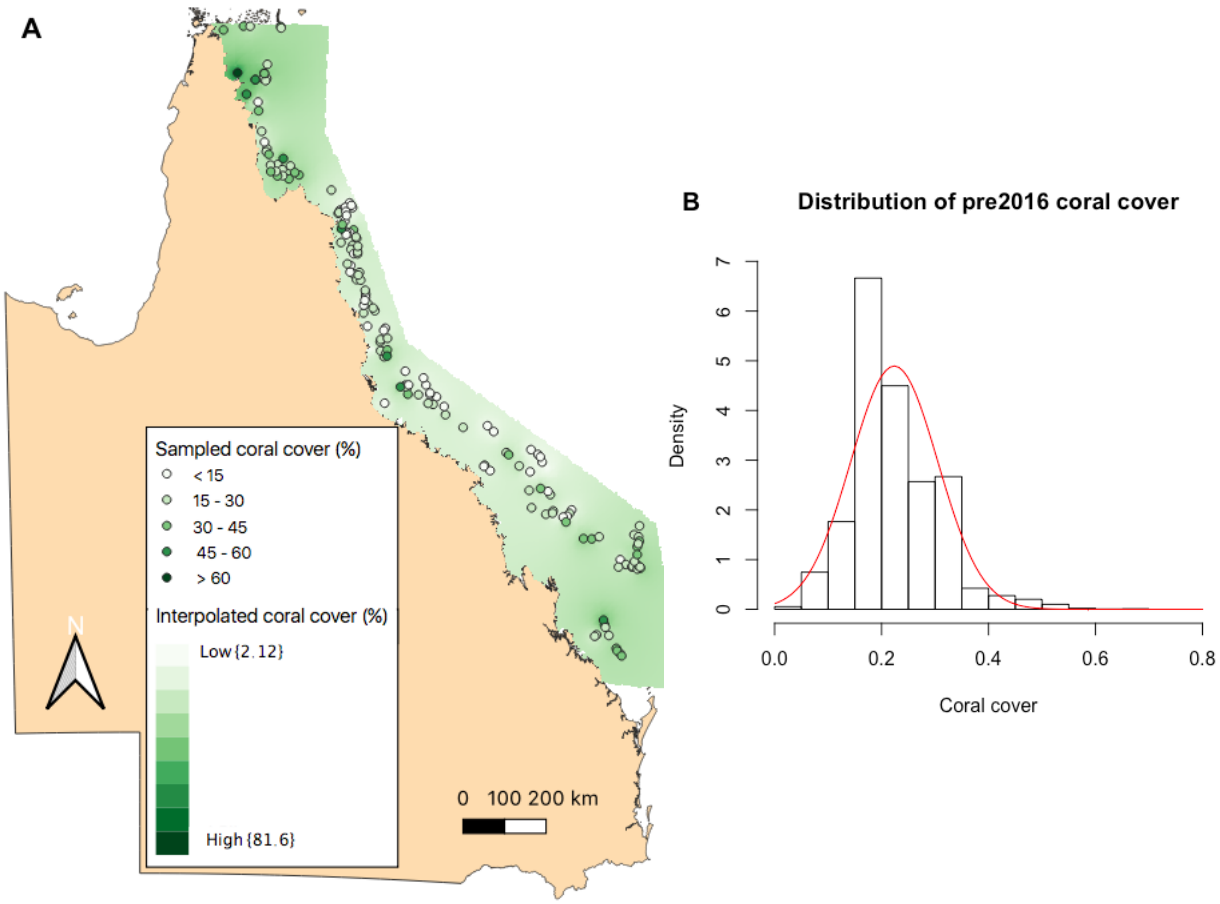


**Figure 2.18:** Results for the pre-2016 event showing **(A)**Interpolated coral cover on the GBR with the dots representing the sampled points used as input for the interpolation; **(B)**Density plot showing the distribution of coral cover data

Indeed, although no mass bleaching event hit the GBR between 2002 and pre-2016, the following disturbances may explain the apparent decline in coral cover:

- In the summer of 2005-2006, a bleaching event took place in the south of the GBR, around the Keppel Islands (Osborne et al., 2011; AIMS, 2018a). Although the spatial scale of this bleaching was limited to this region, its intensity was worse than for previous mass bleaching events, with up to 98% of corals having bleached on some reefs (AIMS, 2018a).

- During the summers of 2008-2009 and 2010-2011, extreme rainfalls triggered important flooding which led to large amounts of freshwater being discharged into the sea, causing inshore reefs to undergo freshwater bleaching (AIMS, 2018a).

- Between 2009 and 2012, many cyclones have hit the GBR (De'ath et al., 2012), including 2009 cyclone Hamish that damaged more than 50% of coral reefs in the southern region and 2011 cyclone Yasi that hit the centre of the GBR and was one of the most powerful ones on records to have occurred on the GBR (GBRMPA, 2019d).

- COTS predation is a real nuisance for corals and once an outbreak occurs, it

spreads like a wave which can last approximatively 10 years. The most recent outbreak started in 2010 and is still spreading across the GBR (GBRMPA, 2019a).

- All the above disturbances are likely to have affected coral cover on the GBR by inducing mortality, but coral cover is not characterized solely by mortality, it is also dependant on the rates of growth of corals (Osborne et al., 2011; Hughes et al., 2011; De'ath et al., 2012). In their study, De'ath et al. (2012) establish that these rates have declined by 15-20% since the 1990's due to increasing thermal stress.

From the above disturbances, the years 2009-2010 seem to have been a turning point in the levels of coral cover on the GBR and this seems to be confirmed by Mellin et al. (2019) as their results show a steep decline during that period (Fig. 2.16). Between 2003 and 2009, on the other hand, they showed a slight increase in mean coral cover by $+0{,}73\%y^{-1}$.

The mean coral cover of 22,37% estimated in this work for the pre-2016 event is likely to be an overestimation, as it is higher than the mean of ± 19% established by Mellin et al. (2019) and the mean of 19,8% established by AIMS (AIMS, 2019a). Also, as regard to the monitoring method, manta tows are limiting in terms of precisely distinguishing causes of reported mortality, depending on the timing of observations (Hughes et al., 2011). Therefore, even though this work puts an emphasis on mass bleaching events to explain spatial and temporal trends in coral cover, the levels of coral cover reported following each of these events can not be solely ascribed to these bleaching events.

Although this work has integrated the pre-2016 coral cover to its timeline in an attempt to take into consideration the recovery process between 2002 and 2016, it does not truly illustrate the recovery in itself because this process is of course very complex and a simple "snapshot" of the coral cover state at one point in time followed by another one several years later is more of a static representation of two different states of coral reef health than a true integration of dynamic recovery. The analysis of recovery would have required deeper understanding of subregional and evolutionary trends of individual reefs, which is beyond the scope of this work.

Fig. 2.19 and Fig. 2.20 show the coral cover results following the 2016 and 2017 events respectively. The average coral cover estimated for the whole GBR for these two events is 18,24% and 17,46%, respectively. As explained in section 2.2, it is the northern part of the GBR which suffered the most from severe bleaching in 2016. The temperatures were so excruciatingly high that some of the corals literally burned to death. This explains why, on Fig. 2.19, we can see that most dots are practically white indicating a coral cover inferior to 15%. Fortunately, the southern region escaped this ravaging heat stress and levelled the coral cover up to an average of 18,24%. This result seems to be in consistency, once again, with the results from Mellin et al. (2019) (see blue dotted lines on Fig.2.16) but also with the estimated 18% averaged coral cover established by AIMS for this same period (AIMS, 2017).

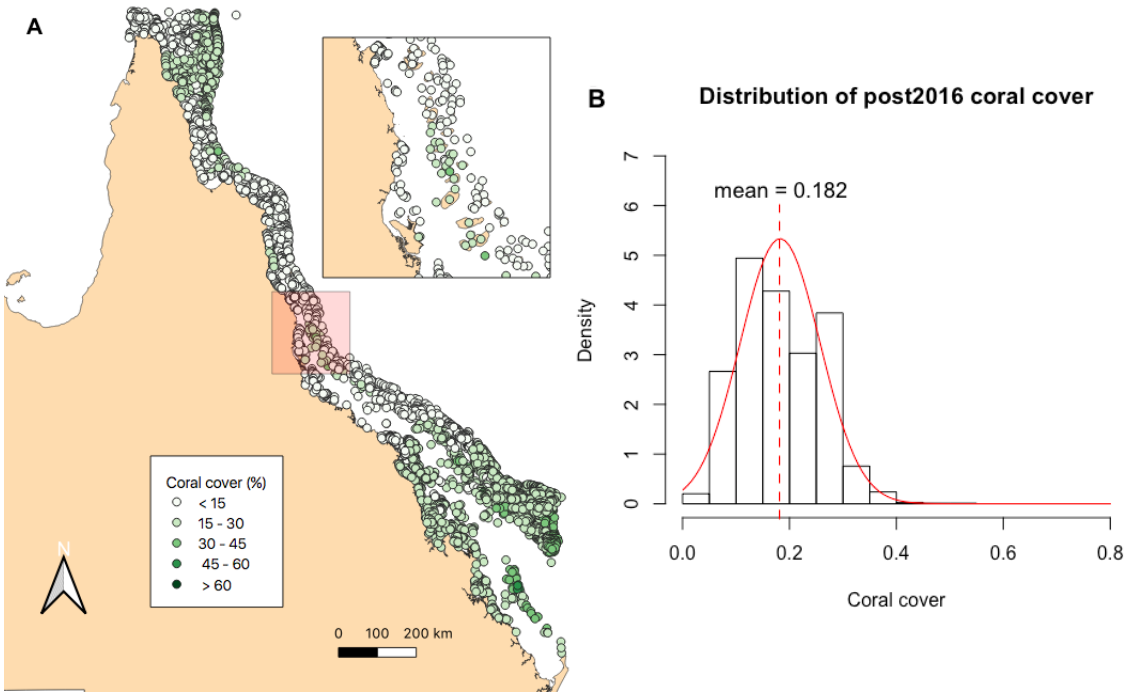


**Figure 2.19:** Results for the post-2016 event showing **(A)**Coral cover on the GBR; **(B)**Density plot showing the distribution of coral cover data

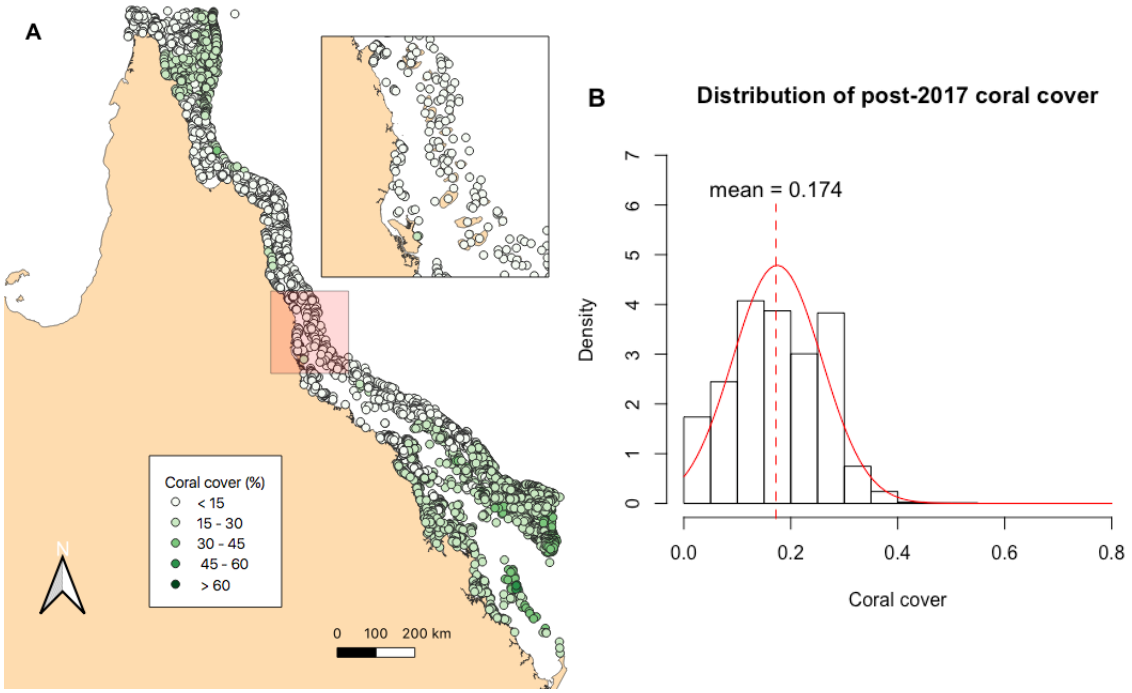


**Figure 2.20:** Results for the post-2017 event showing **(A)**Coral cover on the GBR; **(B)**Density plot showing the distribution of coral cover data

The strike back of a bleaching event in 2017 right behind the 2016 event left little to no chance of recovery for weakened corals. It can take corals up to decades to recover from high mortality rates (GBRMPA, 2017a; Hughes et al., 2018a). In addition to the bleaching event which took place mainly in the central region, 2017 also witnessed the category 4 Tropical Cyclone Debbie (GBRMPA, 2019d; Kerry and

Hughes, 2017). This cyclone could have helped relieve the corals from the heat stress if it had occurred in the same area as the bleaching, but instead it caused additional damage further south to where the bleaching took place (Kerry and Hughes, 2017). The damage from the cyclone was not included in the post-2017 results and these are thus likely an overestimation of the true coral cover for that period. The close-up on Fig.2.20 shows the extent to which coral cover has diminished in the central region due to accumulated bleaching with an area of almost exclusively white dots. A bit less than a decade ago, De'ath et al. (2012) warned that without intervention, coral cover on the GBR would likely fall to 5-10% by 2022. Our results indicate that the estimated coral cover on the GBR averaged 17,46% in 2017, with in mind the fact that this is likely an overestimation. Hence, it is not unlikely that by 2022, the coral cover will further decline to a level close to what was predicted by De'ath et al. (2012).

Overall, according to the approximations made in this work, mean coral cover on the GBR decreased from a hypothetical baseline level of 43,42% to 17,46%. Such results beg the following question: « *What next ?* ». The northern region will take at least 10-15 years to recover its lost corals, that is considering a best case scenario where no major disturbance would occur, which points towards the urgency of tackling climate change related issues by meeting the targets set in the Paris agreement (Kerry and Hughes, 2017). Furthermore, SST prediction models have led to believe that if most corals on the GBR maintain the same existing level of thermal tolerance, they will not be able to withstand future heat stress, leading to annual large-scale bleaching events by the year 2030 (Baird and Marshall, 2002). The possibility of a full recovery hangs by a thread and will need the following conditions to apply:

1. Surviving colonies are able to recover their algal symbionts, regaining their colour, and slowly start contributing to reef growth again. This can be done within a period of weeks to months after SSTs have come back to normal (GBRMPA, 2017a; Hughes et al., 2018a).

2. Longer-lived corals, slow growing species not having bleached can replace dead corals, colonising damaged reefs (Hughes et al., 2018a).

3. Through connectivity, larval recruits reach damaged areas and replace dead corals, allowing for further colonisation (GBRMPA, 2017a; Hughes et al., 2018a).

Unfortunately, these conditions will be unlikely met, given the following conditions. In case scenario (1), due to extensive loss of tissue and important injuries, surviving corals are so weakened, they become easy targets for all sorts of diseases and predation and thus tend to further die slowly (Hughes et al., 2018a). In case scenario (2), slow growing species will likely take decades to replace dead corals, exceeding the return time of future bleaching events (Hughes et al., 2018a). As for case scenario (3), although the replacement of dead corals by larval recruits will take at least a decade according to Hughes et al. (2018a), it seems to be the only condition we can act on favourably. Indeed, if there is a sufficient supply of larvae from fecund, intact reefs to replenish damaged areas through a connected network and if substrate allows for settlement, it will already lay a proper groundwork for

recovery. This means that most resilient and useful reefs (i.e reefs sending over many larvae to other reefs) should be targeted for conservation actions. Identifying these reefs and their influence on the connectivity network across our considered timeline will be the topic of the following chapter.

# Chapter 3

# Weighting connectivity over time

"As connectivity information becomes increasingly available through both empirical and modelling studies, it is foreseeable that conservation planners may soon have access to marine connectivity estimates for designing conservation area networks" (Beger et al., 2010). This accurate prediction highlights the progress that has been made in connectivity modelling over the past decade. Allusions to promoting connectivity in reserve design already emerged in the 1980s with recommendations for the implementation of ecological corridors (Margules et al., 1982; Soulé and Simberloff, 1986) but it is only since the 2000s that ecological connectivity and the impact of its integration in the design of MPAs has attracted escalating attention in the scientific community with scientists today agreeing on the fact it is a crucial component to integrate into conservation planning (Schill et al., 2015; Krueck et al., 2017; Hock et al., 2017; Magris et al., 2016, 2017, 2018; Álvarez-Romero et al., 2018).

This chapter will develop the notion of ecological connectivity and why it so crucial in protecting vulnerable ecosystems such as coral reefs. A summary of how the connectivity used in this thesis was modelled will be provided before analysing the results once this connectivity has been weighted with the coral cover information assessed in the previous chapter. This will allow us to see how the connectivity patterns change across the considered timeline and thereafter use that information in the following chapter using Marxan to integrate connectivity into spatial planning.

## 3.1 Acknowledging the importance of connectivity

When referring to connectivity, many different interpretations can be made (McCook et al., 2009; Kininmonth et al., 2011; Kool et al., 2013), however, in the context of this work, the term connectivity will refer to the demographic linkages that exist between populations (here, coral reefs) through the dispersal of larvae among them, as it is the case in most studies (Almany et al., 2009; McCook et al., 2009; Beger et al., 2010; Schill et al., 2015; Magris et al., 2014, 2016, 2017, 2018). This demographic connectivity, often referred to as ecological connectivity, is different than genetic connectivity which occurs on much longer time scales (e.g. evolutionary). However, maintaining ecological connectivity will, in turn, generally lead to ensuring genetic connectivity as well (McCook et al., 2009).

The benefits of considering this ecological connectivity (hereafter, "connectivity") are manifold, especially in the marine realm (Magris et al., 2018). It is crucial to ensuring ecological integrity of coral reefs and associated ecosystems, biodiversity, and population persistence (Almany et al., 2009; Botsford et al., 2009; McCook et al., 2009; Beger et al., 2010; Schill et al., 2015; Magris et al., 2014, 2016). The supply of larvae provided from robust sources is a key factor in sustaining population replenishment after disturbances such as bleaching events. Even under the highest level of protection, a reef can remain vulnerable if its supply of recruits from other reefs is not ensured (McCook et al., 2009). Besides its implications for ecological aspects, integrating connectivity is also beneficial for fisheries and other economic activities dependent on coral reef ecosystems (McCook et al., 2009; White et al., 2014), as better connected sites are predicted to have higher biomass (Beger et al., 2010; White et al., 2014).

All of the above have implications for conservation planning. Up until recently, most reserve designs did not include connectivity and focused essentially on habitat types or species as proxies for biodiversity, which are more easily represented than dynamic ecological processes (Magris et al., 2016). Questions such as "Where do young recruits arriving to a particular reef come from?", "Where do progeny from adults on that reef go to ?" or "How far do they travel?" remained unanswered (Jones et al., 2009). However, due to the ample evidence that including connectivity is an important factor in boosting the resilience of marine ecosystems, reef managers are starting to transition from a static to a more dynamic-based management to ensure the success of their MPA network (Pressey et al., 2007; Beger et al., 2010; Magris et al., 2014; Abelson et al., 2016; Magris et al., 2018).

Depending on the objectives set out, the connectivity can be derived from different population groups such as fish (White et al., 2014; Beger et al., 2015), migratory species (i.e turtles (Beger et al., 2015; Mazor et al., 2016)) or even sea cucumbers (Beger et al., 2015). In this work, it is the dispersal of coral larvae that was used for connectivity. Seeing corals are the principal engineers of reef habitats, ensuring healthy reefs inherently ensures higher densities of different taxa and thus, community diversity (Hock et al., 2017).

## 3.2 Modelling larval dispersal

A variety of different techniques can been used to measure connectivity between populations (Jones et al., 2009; Kool et al., 2013). The connectivity used in this thesis was established beforehand by Saint-Amand et al. (2018) in the context of PhD research. They used the indirect method of biophysical models which simulate both ocean currents and larval biology to measure connectivity patterns. Unlike empirical methods which are unpractical in terms of data collection and spatial extent, biophysical models are capable of providing potential connectivity information at a larger scale as well as recognizing asymmetrical linkages (Magris et al., 2016). The method developed by Saint-Amand et al. (2018) has the additional advantage of effectively characterizing small-scale exchanges in addition to large-scale ones by using the unstructured-mesh SLIM (Second-generation Louvain-la-Neuve Ice-ocean

Model)[1] which can reach high spatial resolution in specific areas. A brief summary of how the connectivity used in this thesis was calculated is provided in the following subsections but for further information, one will have to refer to Saint-Amand et al. (2018) as details on connectivity modelling go beyond the scope of this thesis.

### 3.2.1 Modelling ocean currents

Ranging from shallow estuarine areas with an average depth of 35 m, to 2000 m deep oceanic waters (GBRMPA, 2018g), the GBR is characterized by a complex water circulation. The ocean currents play an important role in many aspects of marine life, including larval dispersal. Accurately characterizing these currents is thus essential to understanding the exchanges between reefs.

The large-scale circulation is dominated by the South Equatorial Current (SEC) which flows westwards through the Coral Sea, bringing it warm equatorial water (Burrage et al., 1996; CSIRO, 2015). Multiple jets occur immediately north and south of the Queensland Plateau and bifurcate as they approach the Australian continental shelf to form the southward East Australian Current (EAC) and the northward Hiri Current (Fig. 3.1). The EAC stengthens as it exits the Coral Sea further south forming meanders and eddies. The Hiri Current flows along the northern shelf edge of the GBR into the Gulf of Papua, forming a partially closed cyclonic eddy. These two currents form the boundary currents of the GBR. In the central part of the GBR, upwellings of cool Coral Sea waters occur along the bottom of the outer shelf. This cold water ends up surrounding and covering individual reefs in near-surface waters through different mechanisms (eddies, tides, ...) (Burrage et al., 1996; CSIRO, 2015).

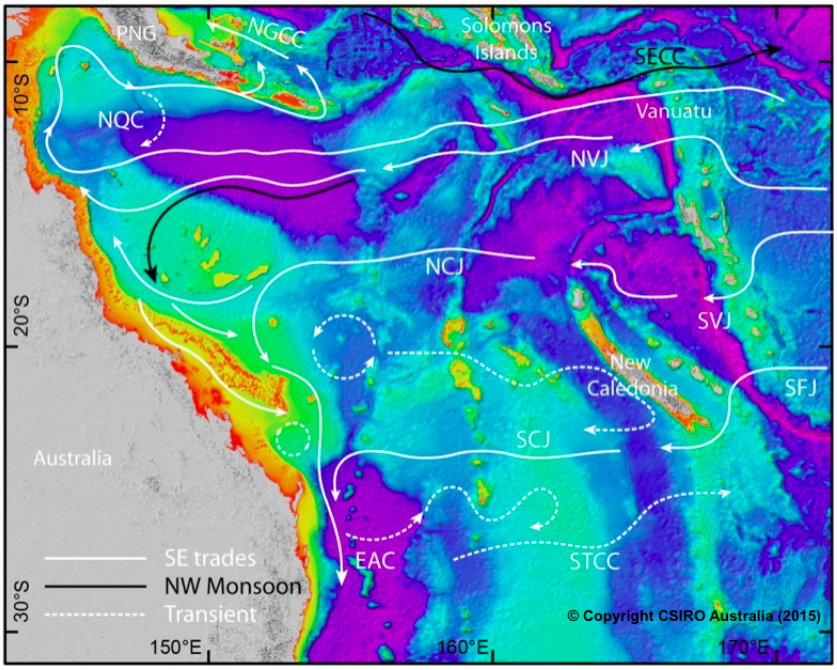


**Figure 3.1:** Bathymetry and key currents in the GBR region (CSIRO, 2015)

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[1] https://www.slim-ocean.be

The EAC and SEC have been reported to strengthen during El Niño events (Steinberg, 2007). However, according to the GBRMPA, there is an overall trend in strengthening and accelerated warming of the EAC, now transporting greater and warmer volumes of water southward (GBRMPA, 2018f). Altered ocean circulation patterns have implications for the transport of larvae among coral reefs.

To capture the patterns of all the different ocean currents surrounding the reefs and to represent both small- and large-scale features, Saint-Amand et al. (2018) used the 2D barotropic version of the unstructured-mesh ocean model SLIM (Fig. 3.2). With this model, they calculated water elevation and depth-averaged current velocity and used the following three factors as external forcings: wind, tides and external currents. The latter are obtained from the 3D model eReefs[2] and account for processes occurring in deep areas which are not captured by the 2D version of SLIM. The results were validated with measurements from 8 IMOS mooring stations and compared with the eReefs results. SLIM performed slightly better at some validation points and eReefs at others (Saint-Amand et al., 2018).

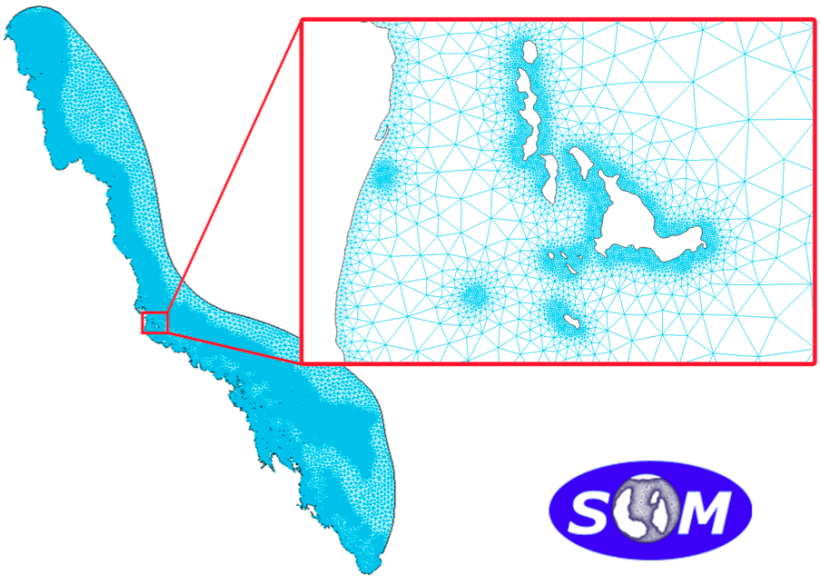


**Figure 3.2:** Unstructured mesh from SLIM consisting of $\sim$ 2.5M elements with a resolution ranging between 200m and 25km (Saint-Amand et al., 2018)

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### 3.2.2 Simulating larval dispersal

The second step of the model consisted in simulating the dispersal of coral larvae and their settlement. The species considered for this simulation was *Acropora Millepora* because of its occurrence across the whole GBR and its biological processes (such as mortality and settling behavior) that have been quantified experimentally (Saint-Amand et al., 2018). The *Acropora* genus is often used in studies as a proxy for coral representation (Baird and Marshall, 2002; Richards, 2013; Wolff et al., 2018). The *Acropora* genus, in addition to accounting for most of the hard coral dynamics

[2]https://research.csiro.au/ereefs/models

due to its sensitivity to disturbances and its fast recovery rates, is predicted to play an important role in ensuring resilience of the GBR under climate change (Wolff et al., 2018), which makes it a relevant proxy in today's context.

The simulation consists in releasing millions of passive virtual larvae over the reefs which then spread following the currents modelled by SLIM in the previous step. The number of larvae is proportional to the area of the reef over which it is released. The physical processes involved in their dispersal were simulated with a depth-averaged Lagrangian particle tracker. It is the mass spawning event of 2012 that was used as target reference. The temporal scale of the simulation extended from December 2012 to February 2013, covering 70 days of dispersal. For biological considerations, the parameters of "mortality" (probability of dying each day), "competence" (ability of larvae to settle on a reef) and "settlement" were considered and adjusted to *A. Millepora*. Also, once a larvae was competent, it was assumed to settle on the first reef it encountered (Burgess et al., 2014; Saint-Amand et al., 2018).

The final result is a connectivity matrix containing information on the exchange of larvae between pairs of reefs (Fig. 3.3(A)). The higher the amount of larvae leaving reef $i$ ('source' reef) and settling onto reef $j$ ('sink' reef), the stronger the connection between those two reefs. This connection is denoted $C_{ij}$, where $i$ is the row and $j$ is the column (Saint-Amand et al., 2018). On Fig. 3.3(A), the strongest connections are in red and the weakest ones in blue. The strongest connections appear on the diagonal and are associated with local retention which is the proportion of larvae released over a reef that settle on that same reef. Local retention identifies the self-persistent reefs (Saint-Amand et al., 2018). The hotspots (i.e. 10% reefs with the highest values in local retention) among these self-persistent reefs are spread out across the GBR (Fig. 3.3(B)). The numbers on the axes of Fig. 3.3(A) represent the ID numbers assigned to every reef of the GBR. These numbers match the IDs from the official GBRMPA reef map, available on the GBRMPA Geoportal[3].

[3]http://www.gbrmpa.gov.au/geoportal/catalog/download/download.page

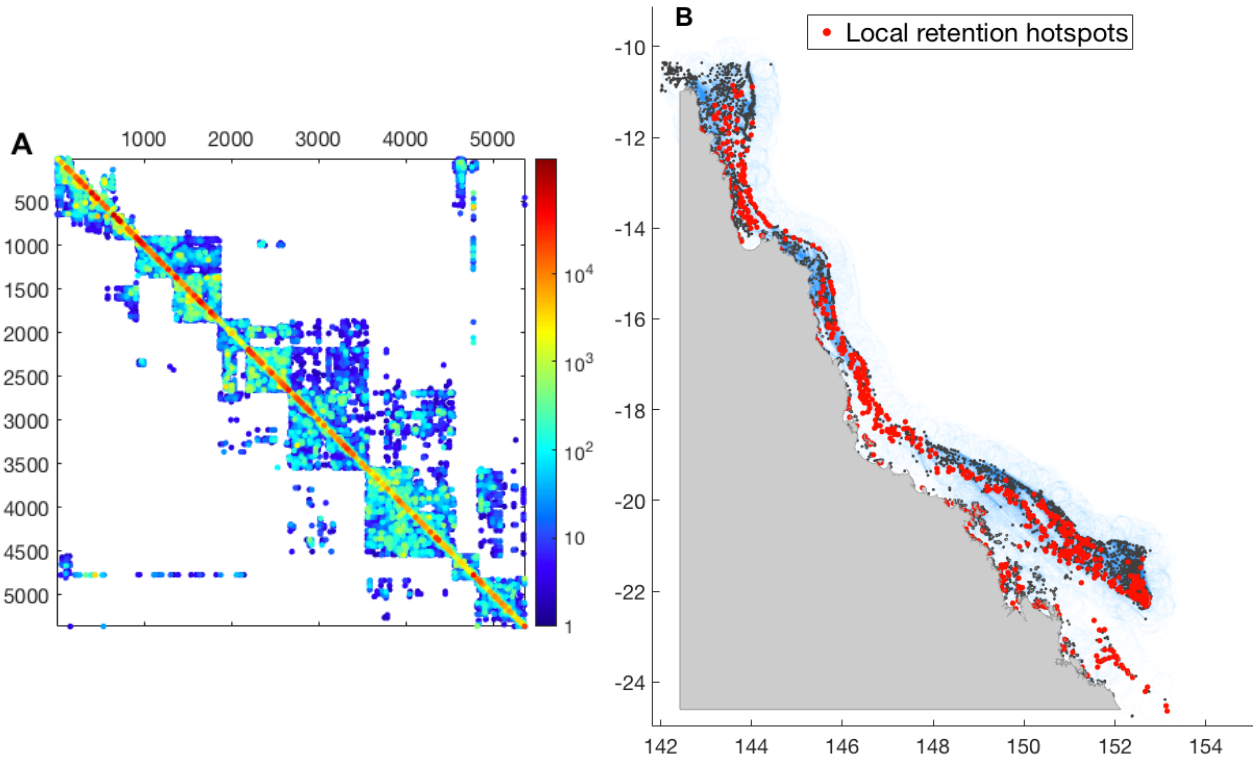


**Figure 3.3:** The connectivity results from (Saint-Amand et al., 2018) visualised through: (A) A dispersal matrix where the rows represent the 'source' reefs and the columns the 'sink' reefs. Red represents the strongest connections. (B) Hotspot reefs in local retention. These hotspots were identified by selecting the 10% reefs with the highest local retention values

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## 3.3 Weighting of the connectivity by coral cover

The biophysical model of Saint-Amand et al. (2018) produces a "potential connectivity", which means the reproductive output of a reef depends solely on its size (the larger the reef, the greater its reproductive output and the more likely it is to attract larvae). However, this is a 'best-case scenario' and, ideally, other factors that might also influence the reproductive output should also be considered to better capture reality (Botsford et al., 2009; Magris et al., 2016). These factors can for example include thermal stress, sedimentation, coastal development or fishing intensity, and can be used as proxies to inform on the quality of the habitat. The factors are then combined into a single "habitat quality index" (HQI) with which the potential connectivity is weighted to become a "realized connectivity" (Magris et al., 2016).

Magris et al. (2016) strongly recommend fine-tuning the potential connectivity with habitat quality data to produce more plausible results. However, in the scope of this work, instead of developing a composite habitat quality index of various threats, fine-tuning was limited to a direct measure of reef quality: coral cover. This remains valid given that most of the factors used in the HQI are known to have significant impacts on coral cover, which makes this approach merely a shortcut. To corroborate this, Magris et al. (2016) found that reef cells with poor quality were characterized by substantial losses of coral cover. The potential connectivity from

Saint-Amand et al. (2018) was thus weighted for each year of our timeline with the coral cover calculated in the previous chapter.

### 3.3.1 Results

Each row of the connectivity matrix was multiplied with the corresponding coral cover of each year which led to 6 different connectivity matrices (respectively baseline, post-1998, post-2002, pre-2016, post-2016 and post-2017).

The results from connectivity analyses can be interpreted in many ways using different graph theory tools. The connectivity metrics often used for network analysis include the outflux, betweenness centrality, local retention, PageRank index, outdegree, and more depending on the objective of the analysis (Magris et al., 2016, 2018; Hock et al., 2017). However, in the context of this work, because the potential connectivity was merely weighted with coral cover over the years and not reassessed from the model with different inputs, the connections themselves remained the same even though their strength was altered. This is why only the outflux and influx metrics, which consider the strength of connections, gave distinguishable results when applied throughout the timeline (Fig. 3.4). Indeed, the outflux metric approximates the source-strength of a reef by measuring its emigration potential and is particularly useful in identifying the reefs most suited to sustaining the surrounding populations whereas the influx measures the immigration potential to a reef predicting its persistence (Minor and Urban, 2007). The graph theory analyses used in this study were performed using the Matlab software (The MathWorks Inc., 2019).

In order to maintain consistency, a threshold value was fixed across the whole timeline for both metrics. This value was determined from the baseline level for which a threshold was calculated beyond which 10% of the reefs with the highest values were selected (i.e. hotspots). This threshold was then used as a reference for the following years. The results for the outflux metric show a visible decrease in the amount of reefs able to reach that threshold with the remaining key sources forming two separate clusters, in the north and in the south of the GBR where the connections seem to be the strongest (Fig. 3.4). The results for the influx metric showed a similar trend with the same two clusters remaining at the end of the timeline. Since connectivity between patches of habitat decreases as the distance between them increases (Almany et al., 2009), it makes sense that these two clusters are isolated from each other. And of all the intra-connected communities, the ones from the north and the south of the GBR seem to be the strongest.

However, outflux and influx are not designed to measure connections strength beyond that of immediate neighbours which means they do not necessarily capture the bigger picture (Minor and Urban, 2007). For this, the Google PageRank index can be very useful (MarxanConnect, 2019). However due to the method of calculation behind this metric (relative weighting applied), it could not be used in the context of this study due to the impossibility of setting a coherent threshold value to apply for all years.

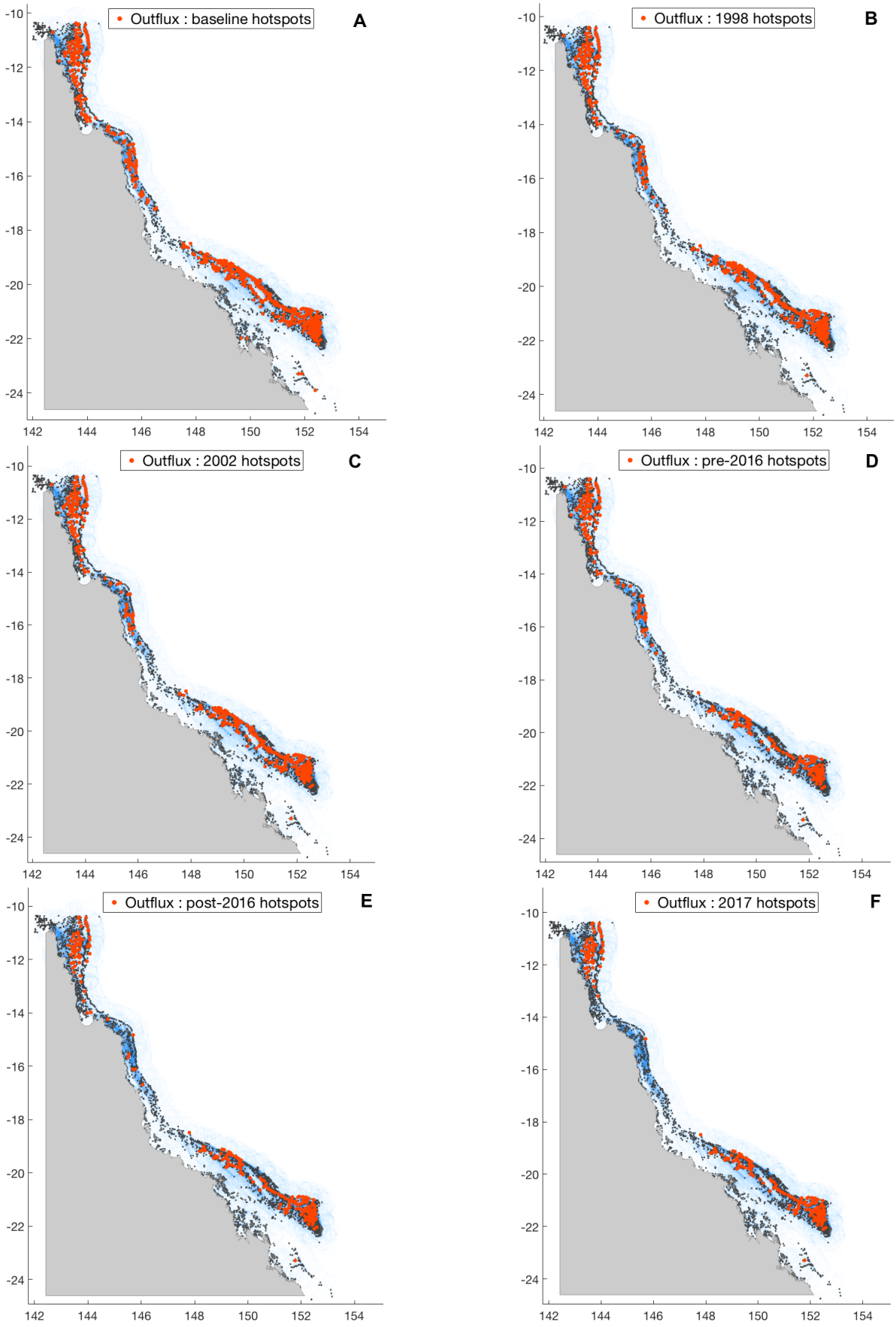


**Figure 3.4:** Results for the outflux metric with hotspots corresponding to the **(A)**Baseline level; **(B)**Post-1998 level; **(C)**Post-2002 level; **(D)**Pre-2016 level; **(E)**Post-2016 level; **(F)**Post-2017 level

### 3.3.2 Improvements

Despite having produced plausible results, the method involved in producing them does entail a few drawbacks and some adjustments could be made to improve the accuracy of the results.

One first drawback is the lack of species representation. Ideally, one should consider multiple species when setting out conservation targets seeing that different species are likely to have different dispersal strategies with possible species interdependencies emerging (Kininmonth et al., 2011; Kool et al., 2013; Magris et al., 2016, 2018). Although capturing these community effects is important, especially for conservation planning, the connectivity used in this study is focused exclusively on a single coral species, *A. Millepora*. Whereas, on the other hand, the coral cover that was used to weigh this connectivity is representative of hard corals and not specifically *A. Millepora*. Although *A. Millepora* was not chosen randomly and the fact that it accounts for most of the hard coral dynamics makes it a good proxy if one single proxy is to be used, the results, however, are relevant for only those reefs where *A.Millepora* is present and its presence is particularly lacking on nearshore reefs roughly between Townsville and Cooktown (Wolff et al., 2018).

Another way of improving the accuracy of the results would be through the use of a composite habitat quality index. Indeed, even though coral cover levels are likely to translate the disturbances (and thus habitat quality) occurring on a reef, the assumption that high coral cover translates high reproductive output is not necessarily true (Hartmann et al., 2018). Indeed, when under acute environmental stressors (e.g. low water clarity due to terrestrial runoff), corals tend to reduce the number of offspring they produce (Hartmann et al., 2018). Even though the processes behind reproductive success are complex and a decrease in the number of larvae does not necessarily guarantee a loss of ecological function (Hartmann et al., 2018), considering habitat quality in addition to coral cover would have been likely to produce more accurate results (Magris et al., 2016).

Also, the potential connectivity used in this study throughout the timeline is specific to the spawning event of 2012. This implies that the connectivity patterns in 1998 are identical to the ones of 2017 (the linkages that is, not their strength), even though they are likely to change from one year to another (Kininmonth et al., 2011), especially considering the prevailing context of global warming to which dispersal patterns are susceptible (Kool et al., 2013; Álvarez-Romero et al., 2018).

Finally, as mentioned in the previous section, not many connectivity metrics could be used without jeopardizing the need for consistency throughout the timeline. An alternative to this would imply changing the calculation method behind these metrics such as the PageRank index in such a way that a threshold can be set allowing for comparison over the years. However, the sole purpose of having calculated these different connectivity matrices is to be able to integrate connectivity information into Marxan and even though most planners focus on the importance of protecting "source" reefs, what truly matters is to ensure whole system connectivity. Indeed, as Kininmonth et al. (2011) pointed it out, "protecting a set of highly productive, but disconnected, sources could be worse than protecting a well-

connected chain of lesser sources, as the overall strength of the network is important for metapopulation persistence". This is why the connectivity used in this study was integrated into Marxan using the method of Beger et al. (2010), which allowed us to identify priority sites that meet conservation targets while still ensuring important connections between reefs were not ignored. Identifying these sites will be the topic of the next and final chapter.

# Chapter 4

# Integrating connectivity into conservation planning with Marxan

"The true effectiveness of any reserve system should be judged not by what is present now but what will persist there in the future" (Game et al., 2008). This final chapter is the culmination of this work in that it uses the information from the two previous chapters on coral cover and connectivity to help answer the question that was raised in the introduction of this thesis: "Which reefs -if there are any- are consistently selected to be in MPAs throughout the timeline and thus appear to be sufficiently resilient due to their performance in the past for us to secure them in the future ?".

This chapter will try and provide an answer to this question using the software Marxan. Although the previous chapter already gave an indication as to which reefs seem to consistently remain important connectivity sources throughout the timeline, this information alone is not sufficient to guide the design of a reserve network (Kininmonth et al., 2011; Almany et al., 2009). Indeed, conservation planning requires taking different aspects into consideration, often environmental, ecological and socioeconomic, to assess which areas are most suitable for protection (Abelson et al., 2016). Also, depending on the objectives set out, the resulting reserve design can vary greatly. These aspects and the sensitivity of the reserve design with regard to the objectives set will be illustrated here through different scenarios.

This chapter will start by providing an insight into how Marxan operates exactly. It will be followed by a section on the groundwork for implementing Marxan with an overview of the different data used as input for the scenarios considered, the reasoning behind the setting of conservation targets and also details on the calibration of certain important parameters. Finally, the results from Marxan will be presented and a discussion on how to understand them as well as use them to try and best address the research question above will conclude this chapter.

## 4.1 Understanding how Marxan operates

Marxan is a decision support software used worldwide for the design of reserve systems (Ball et al., 2009). Marxan stands for "marine reserve design using spatially explicit annealing" (Ball et al., 2009). The guiding principle behind this software is to identify potential sites, referred to as 'planning units' (PUs), that achieve ecological targets whilst minimizing the associated cost. This is known as the "minimum set problem", in contrast with the "maximum coverage problem" which tries to capture as much biodiversity as possible with a pre-determined budget (Game and Grantham, 2008).

There are two main types of algorithms used to solve reserve design problems: the exact algorithms and the heuristic (non-exact) ones. The former singles out an optimal solution whereas the latter identifies several good, near-optimal ones. Marxan finds itself in the second category using a robust heuristic method known as "simulated annealing", which provides solutions that get much closer to the optimal solution than other heuristics (Game and Grantham, 2008). The range of multiple good solutions calculated makes Marxan a flexible tool in decision making, providing stakeholders with a range of alternatives to choose from (Ball et al., 2009; Game and Grantham, 2008). This selection of 'good' over 'less good' solutions is done through the objective function. Marxan continually compares a range of different sets of planning units which are assigned a value through this function and the lower the value the better. The mathematical formulation behind this objective function is illustrated by Equation 4.1 that Marxan minimizes to find near-optimal solutions (Game and Grantham, 2008) :

$$minimize \sum_{PUs} Cost + BLM \sum_{PUs} Boundary + \sum_{ConValue} SPF \times Penalty + CTP \quad (4.1)$$

The first term of Eq. 4.1 is the sum of the costs assigned to each planning unit included in the reserve. This cost can be whatever the user finds most appropriate to determine the value of the planning units (e.g. opportunity cost, socio-economic, geographic, etc.).

The second term is often used to incorporate connectivity into the design of the reserve system. It is the sum of the boundary cost (or 'boundary length'), weighted by the boundary length modifier ($BLM$), for all the planning units included in the reserve. Originally, this term was designed to minimize the fragmentation of the reserve system and help guide Marxan into identifying more clumped solutions, with a lower boundary length implying a less fragmented reserve (Game and Grantham, 2008; Ball et al., 2009). However, the significance behind this boundary cost can now take different interpretations depending on how the user defines the connectivity of its reserve. This term will be further discussed in the next section, however, whatever interpretation it is given, Marxan will always aim to minimize it.

The third term of Eq. 4.1 pertains to the the biodiversity aspect of the reserve. When setting up the reserve design problem, the user must indicate the conservation features he/she wishes Marxan to include in the final reserve. These features can be specific such as endangered species or more generic such as habitat types. Each of these conservation features is assigned with a target (e.g. a % area that must be included in the reserve) which Marxan will aim to meet. If Marxan fails to do so, a penalty increases this term and thus the value of the objective function. Along with the target, each feature is also assigned a Species Penalty Factor (SPF). This factor is multiplied with the internally-derived penalty value and indicates to Marxan how important it is to meet the specified targets (Ball et al., 2009). The higher the SPF, the harder Marxan will work on achieving those targets. The calibration of this parameter will be discussed in the section dedicated to this purpose.

The final term of Eq. 4.1 is the Cost Threshold Penalty. This term is used when the overall cost of the reserve must remain below a pre-defined threshold (Game and Grantham, 2008).

Terms 1 and 3 of the objective function must necessarily be represented in order for Marxan to be able to run whereas terms 2 and 4 are not mandatory, although the second term is recommended. With the simulated annealing implemented through this objective function, Marxan randomly selects new combinations of planning units and judges the new configuration in terms of the resulting objective function value. If the value is improved by the random inclusion of a planning unit, then that planning unit is kept and if the value is worsened, then it is removed (Ball et al., 2009).

## 4.2 Implementing Marxan

For Marxan to be able to run, it requires a set of input data related to the objective function discussed above. The different input data used in this thesis will be developed in this section. To test the sensitivity of the results to changes in these inputs, several scenarios were implemented. Hence, in addition to evaluating the impact of variations in connectivity and coral cover across the timeline on the reserve design, additional variations were considered such as that of cost and conservation features. The reasoning behind the configuration of these different scenarios is illustrated in Fig. 4.1. A detailed framework on the configuration of the input data within these scenarios is provided in Appendix A. Overall, this led to a total of 96 scenarios (and not 120 as Fig. 4.1 would suggest because for the case where the existing MPAs were considered, only the years following the rezoning were considered). These different scenarios were implemented into Marxan using the R software.

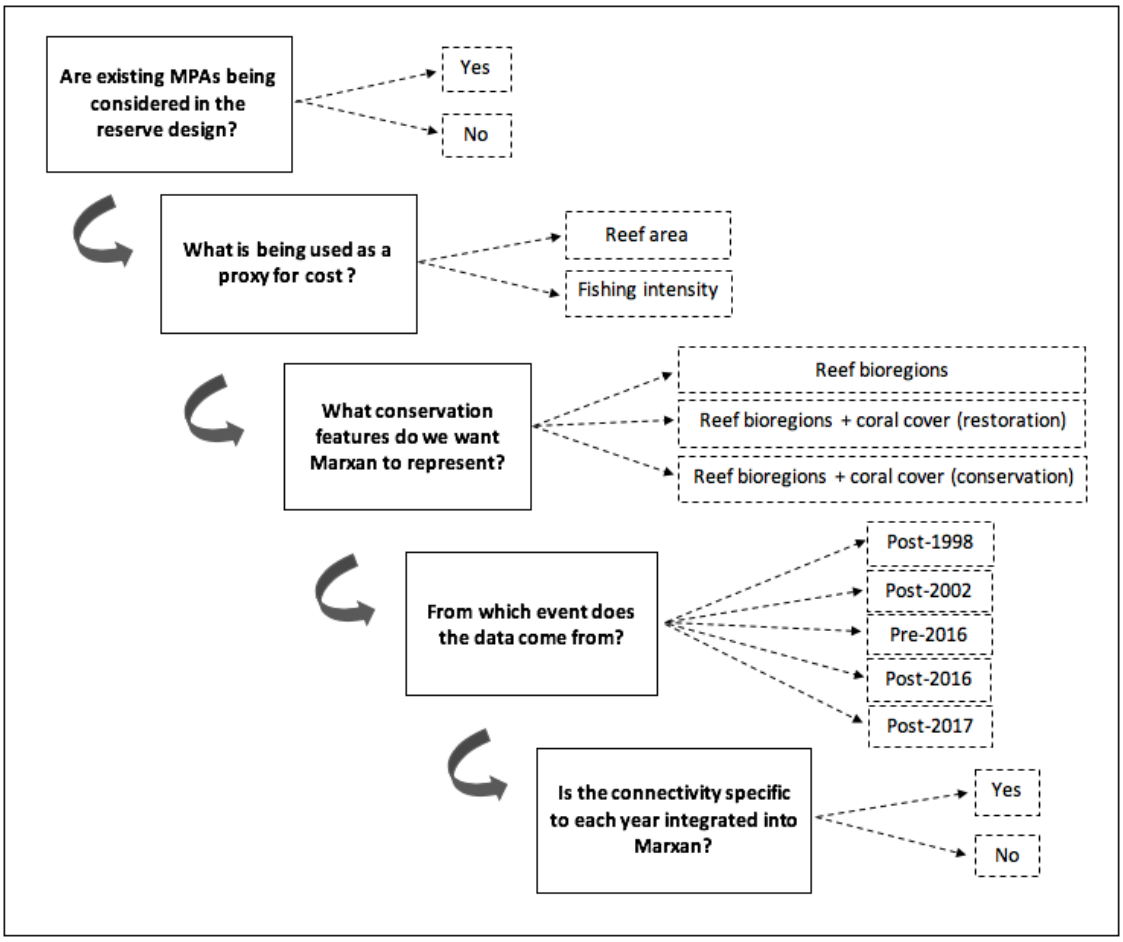


**Figure 4.1:** The key steps in configuring the different scenarios to be implemented into Marxan

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## 4.2.1 Collecting input data

### 4.2.1.1 Planning units

Different types of planning units can be used to divide the study region into potential sites Marxan can choose from to design the reserve (regular grid, adjacent hexagons, environmental units, etc.). Since our work integrated connectivity into the reserve design, the planning units had to be based on the source and sink sites used for the connectivity matrix. In our case, these sites were based on the natural ecological division of the GBR using the 5363 features provided by the GBRMPA official reefmap as planning units. By dividing the region in this way, a reef selected for conservation will always have its entire patch included, which is recommended to avoid fragmentation and enhance protection (Fernandes et al., 2005; Strain et al., 2018).

Once the planning units have been defined, each of them must be given a particular status. This status will indicate to Marxan how to consider the planning unit. A status of '0' indicates the planning unit is available for selection. A status of '2' means the planning unit is already pre-set to be in the final reserve system and considered "locked-in". A status of '3' means the planning unit cannot find itself in

the final reserve and is excluded for selection, it is considered "locked-out" (Game and Grantham, 2008). Since the official reefmap of the GBRMPA contains other features in addition to reef features (e.g. islands, rocks, cays), these features were given a status of '3'. Reefs which found themselves outside the legal boundaries of the GBRMPA were also considered locked-out.

For the attribution of 'locked-in' status, a first division into two major scenarios was made. The first scenario did not consider the existing MPAs created by the 2004 rezoning plan (Fernandes et al., 2005) and assigned a status of '0' to all planning units except the ones already considered excluded. The second scenario did consider the existing MPAs and assigned a status of '2' to the relevant planning units. These MPAs refer to the Marine National Park Zones (or 'Green Zones') from the current zoning (cfr. Chapter 1). In doing so, a comparison with the current reserve design could be made to see how closely it meets our conservation targets. However, this comparison can only be made within a certain extent and no conclusions on the efficiency of the current reserve design can be drawn from it. Indeed, there is no doubt that the zoning plan set out by Fernandes et al. (2005) was one of the most high profile and successful 'real-life' applications of Marxan (Ball et al., 2009). The entire planning process took them about six years, relying on independent experts advice to best address biodiversity issues, gathering information from community input and round-table discussions, reporting the degree of achievement for various design options, planning for negative impacts, etc. (Fernandes et al., 2005). The results produced in this work were based on other objectives, with different parameters values, which limits the comparison but it is still interesting to see how the areas produced by Marxan in this work conform with the current MPA design. It is important to keep in mind that because the planning units used here encompass reef features, the comparison was made by exclusively selecting the reefs the current zoning covers, even though the zoning covers more area than just that of reefs, such as that of estuarine habitats, interreef chanels, deepwater seagrass habitat, and more (Fernandes et al., 2005).

#### 4.2.1.2 Cost

In addition to status, every planning unit must also be given a specific cost. This cost information can be of different natures, from a simple default value of '1' for all planning units to a complex combination of social and economic measures (Game and Grantham, 2008). The cost information used has a significant bearing on the results and to perceive this level of influence, two different proxies were considered for our cost data: reef area and fishing intensity. This was the second crossroad in the formulation of our scenarios.

The rationale behind using the area as a proxy for cost is that the larger the reserve system, the more costly it will be to implement and to manage (Game and Grantham, 2008). The rationale behind using fishing data as a proxy for cost is that commercial fishing is the largest extractive activity in the GBR (GBRMPA, 2018b) and is therefore important to take into consideration when designing the reserve system since it could potentially impact the total annual catch and have significant economic implications (Stewart and Possingham, 2005). In doing so, the

aim is to establish a network of MPAs that can achieve conservation goals whilst at the same time minimizing negative repercussions on the fishing industry even though this might be challenging. If a planning unit is selected to be in the reserve, its cost will be added to the value of the objective function, which means Marxan will avoid targeting highly valued reefs. Since a reef is valued by fisheries for its high productivity in fish biomass, the cost of putting such a reef inside the reserve system will be higher than that of a reef which has a low productivity.

To translate fishing intensity across our study region, fishing data was collected from the Department of Agriculture and Fisheries of the Queensland Government on their 'QFish' platform[1]. The five main types of commercial fisheries active on the GBR are the line, net, trawl, trap and dive-based harvest fisheries (GBRMPA, 2018b). However, only data relative to line, net and trawl fishing were used here as the data relative to trap and dive-based fisheries was not sufficiently comprehensive. The downloaded data contained information on the catch of species (in tonnes) per year and per location (grid code). To project this information on the GBR, it was combined with the logbook grid[2] used by fishers to report the location of their daily catch. In order to use this cost data in both scenarios where the existing MPAs are either considered or not, two different fishing costs were established: one corresponding to the fishing intensity before the GBR underwent its rezoning in 2004 (Fig. 4.2(A)) and another fishing intensity following this rezoning (Fig. 4.2(B)). For the case where the existing MPAs were not considered, the 'pre-2004' cost data was used for all years and for the case where the existing MPAs were indeed considered, only the years following the 2004 rezoning were used and assigned the 'post-2004' cost data. All reefs had a cost value superior to zero as a value of zero would make Marxan automatically include it in the final reserve system even though that reef may not meet any particular target. Seeing no cost threshold was set in our analysis, the last term of the objective function was not considered by Marxan.

[1]http://qfish.fisheries.qld.gov.au

[2]http://qldspatial.information.qld.gov.au/catalogue/custom/detail.page?fid=214AA7AA-BB0B-4137-B84B-6548A95FB95F

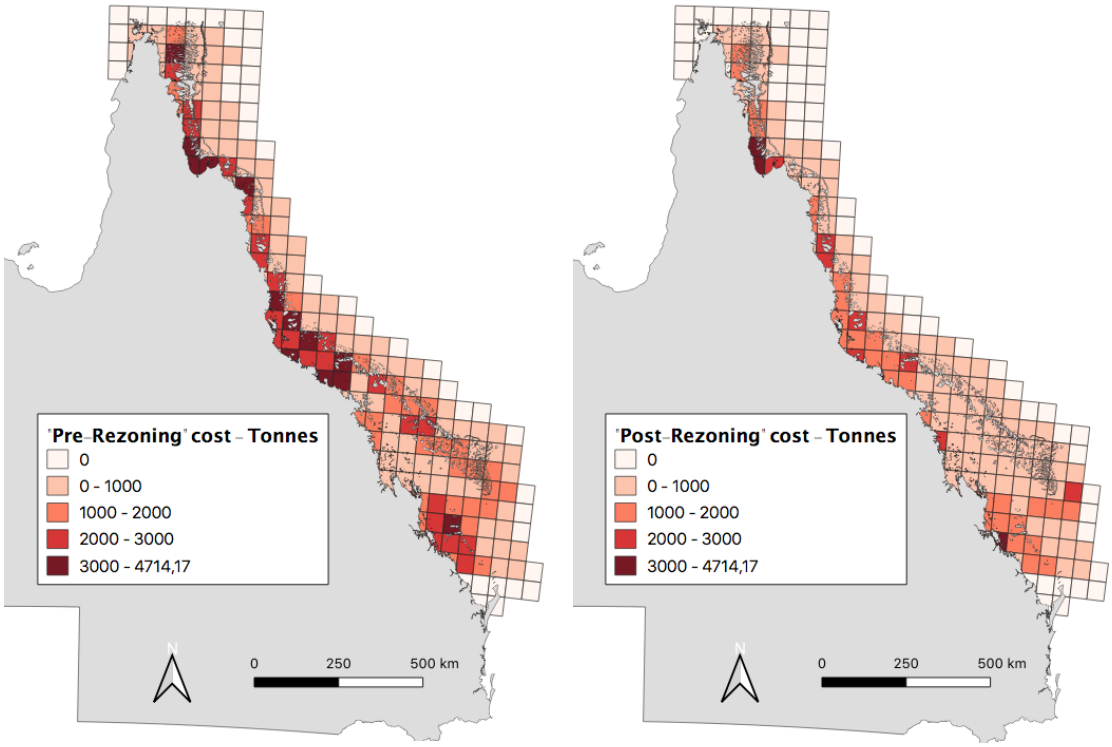


**Figure 4.2:** Two separate fishing costs used into Marxan: **(A)**Fishing intensity prior to the 2004 rezoning of the GBR; **(B)**Fishing intensity after the 2004 zoning was established on the GBR.

#### 4.2.1.3 Conservation features

To be able to run, Marxan must be given information on what features it must involve in the reserve design. This was the third crossroad in the construction of our scenarios. Although applications on spatial planning usually include a range of different features such as individual species, habitat types, foraging and nesting spots (Fernandes et al., 2005; Game et al., 2008; White et al., 2014; Mazor et al., 2016; Magris et al., 2018), developing such a framework was beyond the scope of this study. An alternative was to use the 'bioregions' of the GBR as a proxy for biodiversity representation. These bioregions were identified by a panel of experts in the context of the Representative Areas Program in 1999 (Day et al., 2002). They used more than 40 layers of data compiled through years of research to establish bioregions that are each representative of an area where the animal and plant assemblages as well as the physical features differ sufficiently from the rest of the GBR to be considered as a distinct bioregion (Game et al., 2008). There are 70 bioregions in total comprising 30 reef bioregions and 40 non-reef bioregions. Since the focus here is on reefs, we used the 30 reef bioregions (Fig. 4.3).

In addition to the bioregions, coral cover was also used as a conservation feature. Coral cover was considered in order to distinguish two separate scenarios: conservation and restoration.

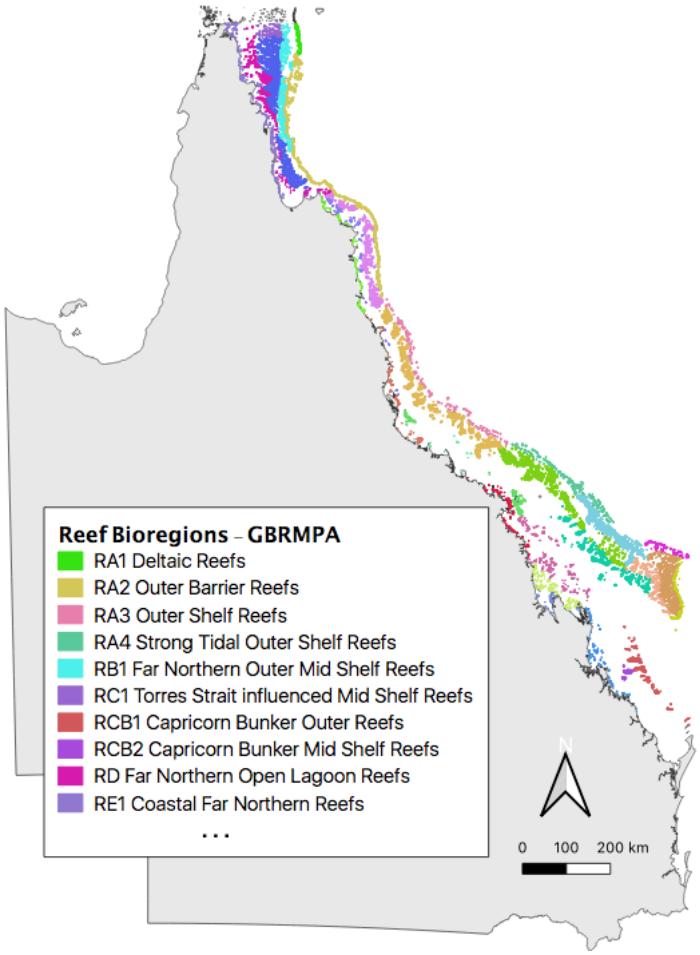


**Figure 4.3:** The 30 reef bioregions defined by the GBRMPA and used in this study as a proxy for biodiversity.

The conservation scenario targeted reefs with high coral cover as this is a good indicator for healthy reefs (cfr. chapter 2). However, although coral cover is one of the preferred metrics for informing reef management, findings have raised awareness on the limits of such indicator in representing species richness (Richards, 2013; Lindenmayer et al., 2015). Indeed, Richards (2013) found that the relationship between coral cover and species richness was not linear and thus high coral cover was not necessarily a good proxy for biodiversity. This may be explained by the fact that high coral cover may occur at any stage of community development. For instance, during recovery, a reef may have low coral cover but the number of different species present may nevertheless be high (Richards, 2013). Still, despite these findings, coral cover does remain a good indicator for identifying general trends (Bruno and Selig, 2007; Osborne et al., 2011; De'ath et al., 2012; Lindenmayer et al., 2015; Mellin et al., 2019) and with regard to species richness, this issue is mitigated by the fact we are considering bioregions for biodiversity representation. Two categories were used for this conservation scenario to explicitly consider coral cover as conservation features:

1. High coral cover (30-40%)
2. Very high coral cover (> 40%)

The lower threshold was set to 30% as this is the level as of which Januchowski-Hartley Fraser A. et al. (2017) found reefs to be in a recovered, positive status and under which Wolff et al. (2018) considered reefs as vulnerable.

The restoration scenario on the other hand targeted reefs with low coral cover. Although the traditional approach to designing MPAs for reef management is to protect healthy reefs, the ongoing decline in coral cover observed worldwide has raised doubts as to whether this approach is still as relevant as it was when the concept of MPAs first emerged decades ago (Abelson et al., 2016). While there is a lack in attempts to consider degraded reefs as explicit protection targets, Abelson et al. (2016) argue that expanding the range of MPAs to include degraded reefs could help reverse this ongoing trend. Although the distinction between a healthy or degraded reef is made in a relatively arbitrary manner, in this work we considered degraded reefs as reefs with low coral cover. Similarly to the conservation scenario, two categories were formulated for the restoration scenario:

1. Very low coral cover (5-15%)
2. Low cover (15-30%)

The lower threshold of 5% was set, once again, according to the findings of Januchowski-Hartley Fraser A. et al. (2017) which reported reef accretion on reefs with more than 5% massive coral cover. This is why reefs with coral cover below 5% were excluded as their cover is so low, it might be more productive focusing restoration efforts on reefs that stand a better chance.

Overall, this led to three different protection objectives for Marxan to implement:

1. Biodiversity representation (30 features) : Bioregions;
2. Biodiversity representation with a conservation perspective (32 features): Bioregions and healthy reefs;
3. Biodiversity representation with a restoration perpective (32 features): Bioregions and degraded reefs.

#### 4.2.1.4 Connectivity

The final input data considered for implementation into Marxan was the connectivity information which was different for every year. The standard approach in integrating connectivity into Marxan is through the boundary length file (see $2^{nd}$ term of the objective function). The designation 'boundary' comes from the fact that in its most typical application, the boundary length refers to the actual geographical length of the boundary between adjacent planning units. It can be considered as the cost to pay for not having both planning units in the final reserve, which is why the 'boundary length' is also often referred to as the 'boundary cost'. In this way, by minimizing the boundary cost, Marxan minimizes fragmentation by privileging a clumped reserve (Game and Grantham, 2008).

However, the boundary length is not restricted to this geographical definition and can take other more complex interpretations such as larval dispersal in our case. In this context, by minimizing the 'boundary', Marxan clumps planning units

not based on their adjacency but rather on the strength of their connections, privileging highly connected reefs to disconnected ones (Schill et al., 2015). Any type of measure can be implemented through the boundary length as long as it respects its core principle which states "it is essentially a relative measure of how important it is to include one planning unit in the reserve system, given the inclusion of another" (Game and Grantham, 2008).

While most studies that integrated ecological connectivity assumed symmetrical strength of the connections, in reality, this is rarely the case and Beger et al. (2010) addressed this issue by formulating an explicit method for incorporating asymmetric connectivity into Marxan. In using an asymmetric connectivity matrix instead of a symmetric one, they changed the interpretation of the boundary length modifier (BLM, Equation 4.1) by introducing the 'connectivity strength modifier' ($CSM$). While the BLM indicates the importance of reducing the fragmentation of the reserve relative to its cost, the CSM expresses the relative importance of ecological connectivity (Beger et al., 2010). For instance, if the CSM is given a value of zero, Marxan will not take connectivity into account, and conversely, it will aim to minimize the missing connections (Beger et al., 2010). The CSM has the potential to add a functional dimension to the BLM's structural connectivity (Magris et al., 2014). The calibration of this parameter will be further discussed in the next section.

In the construction of scenarios, this was the final crossroad in indicating to Marxan whether connections should be considered or not. This allowed for comparison between the reserve designed with and without connectivity.

### 4.2.2 Setting conservation targets

Conservation targets indicate the amount of each feature Marxan must include in the final reserve system (e.g. occurrences of species, % area of habitat type, etc.). Although these targets are referred to "conservation targets" in the Marxan literature, to avoid any confusion with the conservation scenario, these targets will be referred to here as "representation targets". A representation target was set for each of the 34 conservation features considered. These targets were expressed as % area of each feature Marxan needed to include in the final reserve.

Because these representation targets have a strong influence on the resulting reserve, the process involved in setting them ideally involves a panel of experts to agree on a minimum range of targets for each feature that will satisfy ecological objectives (Ban et al., 2013). However, most of the time these targets are set relatively arbitrarily, with other constraints considered than solely ecological ones, such as political convenience (Margules and Pressey, 2000; Stewart et al., 2007). Although there is no strict rule as to what targets should be applied, commonly used targets for selecting reserve areas vary from 10-12% (Stewart et al., 2007) to 30% (Schill et al., 2015; Krueck et al., 2017).

Regarding the bioregions of the GBR, most studies used a 20% target representation (based on Fernandes et al., 2005). In the context of this work, the main objective for Marxan was to design a network of highly connected reefs whenever

connectivity was considered. Seeing that the bioregions are spread out across the GBR, applying to each of them a 20% or higher target would have a strong bearing on the spatial design. The higher the target for these bioregions, the less distinguishable the spatial patterns of the conservation or restoration scenarios will be. Yet, the aim here is to identify those patterns, especially when connectivity is integrated. Even though the reasoning of applying more protection for better protection is valid, it does not address the issue of having to compromise between sites with distinct assets to satisfy overall constraints (Margules and Pressey, 2000). Here a trade-off had to be made between bioregions representation and connectivity representation. The target of 10% was finally set based on the $11^{th}$ Aichi Biodiversity Target which states that 10% of coastal and marine areas should be conserved by 2020 to make sustainable use of the oceans, seas and marine resources (UNEP-WCMC and IUCN, 2016). This target was also applied in an other context (Beger et al., 2015).

For the fixing of the coral cover targets, no reference was found in the available literature as to what amount of high coral cover should be in a marine reserve or what amount of low coral cover should be selected for possible restoration. For this reason, an arbitrary 30% target was chosen for the coral cover categories. Since a 30% target of reef area has already been used in the past for the protection of coral reef habitats (Schill et al., 2015; Krueck et al., 2017), using a similar 30% target seemed appropriate. In addition, seeing that 20% is a common target for bioregions, by considering reefs with high coral cover (or low coral cover) as a feature with added value for conservation (or restoration), it seemed reasonable to fix a >20% target (here, 30%). This is similar to how Ban et al. (2013) distinguished representational (i.e. here, the bioregions) and special features (i.e. here, coral cover feature), assigning higher targets to the latter.

However, as this work covers information from different years, coral cover varied from one year to another, and therefore fixing a 30% specific to every year would have produced different amounts. Yet, what we want is to fix a target which is consistent throughout the timeline in order to be able to compare results on a same basis. In order to do so, a fixed 30% amount had to be deduced from one year and maintained as same target for the other years. This had to be adjusted depending on what scenario was considered, i.e. conservation or restoration.

For the conservation scenario, although it is recommended to rely on baseline information -when available- to set conservation targets (Bellwood et al., 2004; Knowlton and Jackson, 2008; Hughes et al., 2010, 2011; Eddy et al., 2018), here we used the 1998 event as the reference from which to draw the amount of 30% target to apply for the following years. Although this is an example of a shifting baseline (cfr. chapter 2), the rationale behind this is that the gap between the very high coral cover from the baseline level and that of the other years was so great that using the baseline level as a reference would have made it impossible for Marxan to achieve the conservation target for any of the following years. For this reason, the post-1998 level was used. Despite this being a typical case of shifting baseline, the calculated baseline level nevertheless allowed us to assess the importance of this shift. This showed that the 30% target from the post-1998 level resulted in an amount which represents only 6% of the baseline level. However, this is only for the 'Very

high coral cover' category (feature). For the 'High coral cover' feature, the post-1998 level actually had higher amounts of reefs than the baseline level. This makes sense since 30-40% coral cover appeared to be the lower threshold in historical levels.

For the restoration scenario, the same approach was adopted, using instead the end of the timeline as reference, seeing that reefs' need for restoration is higher at the end than at the beginning of the timeline. In view of this method producing a much higher amount in this scenario than that produced in the conservation scenario, the attribution of targets was slightly adjusted. Instead of applying a 30% target for both features, 30% was applied to the 'Very low coral cover' feature and 20% to the 'Low coral cover' feature. In this way, the area covered by these targets (494 911 ha) was still higher than that of the conservation scenario (412 840 ha) but to a more reasonable extent.

### 4.2.3 Calibrating parameters

The final step before producing the results was setting the value of a few important parameters in Marxan. The calibration of these parameters ensures Marxan produces robust results which meet our requirements. Ideally, for every scenario implemented into Marxan, parameters should be calibrated accordingly (Game and Grantham, 2008). However, because this work involves many scenarios, a calibration was not performed for every single one but rather one year was chosen (post-2002 for when exisiting MPAs were not considered and pre-2016 for when they were considered) on which calibration was performed for every scenario (type of cost, features considered, connectivity integrated or not). Then a cross-check with the post-2016 event was performed for a few of those scenarios to see if the parameters calibrated for 2002 performed just as well for post-2016. Indeed, they did, signifying a certain consistency could be maintained across the timeline, which has been one of our main concerns throughout this work. Indeed, if the parameters were drastically different from one year to another, we would not know if the changes in the results would be due to the difference in the input data or in the parameters. The parameters should also avoid being drastically different from one scenario to another for the same reasons. The different parameters calibrated were the CSM, SPF and the number of iterations.

The CSM was calibrated to make sure Marxan put enough emphasis on representing ecological connectivity. This was done using the method of Stewart and Possingham (2005) which involves finding the trade-off between minimising the boundary length (in our case, this corresponds to minimizing the amount of missing connections) and minimizing cost. The CSM did not vary much among scenarios, ranging from $\sim$ 20 to $\sim$40 and thus, an acceptable trade-off was considered for a CSM of 30 (Fig. 4.4). Of course, for the scenarios where connectivity was not considered, the CSM was set to zero.

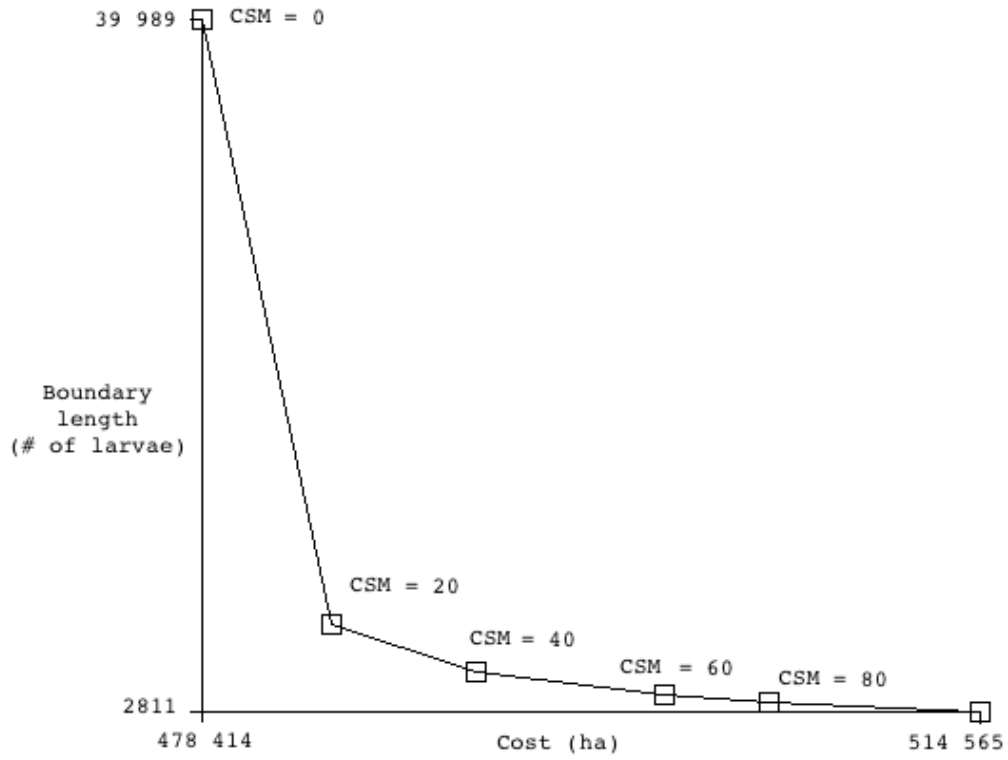


**Figure 4.4:** The appropriate CSM value of 30 was determined by finding the trade-off between minimizing the boundary length and minimizing the cost.

The calibration of the SPF meant finding the SPF value for which each feature is adequately represented in the reserve system. This value is found by selecting the SPF for which the missing values is close to zero. This was satisfied by assigning a SPF of 4 to all features for when the area was used as a surrogate for cost (Fig. 4.5) and a SPF of 15 for when the fishing intensity was used.

Finally, a calibration was also performed to assess how many iterations Marxan should perform in the simulated annealing process to consistently produce near-optimal solutions. The higher the number of iterations, the more time Marxan will spend on designing the reserve system, which has its practical limits (Game and Grantham, 2008). This meant finding the trade-off between the execution speed (i.e. number of iterations) and the score of the objective function (i.e. efficiency) (Watts et al., 2017). This resulted in a value of 10 millions iterations being an acceptable trade-off (Fig. 4.6).

Marxan applied the appropriate parameter values for each scenario using a value of 100 runs, commonly used value by Marxan users (Schill et al., 2015).

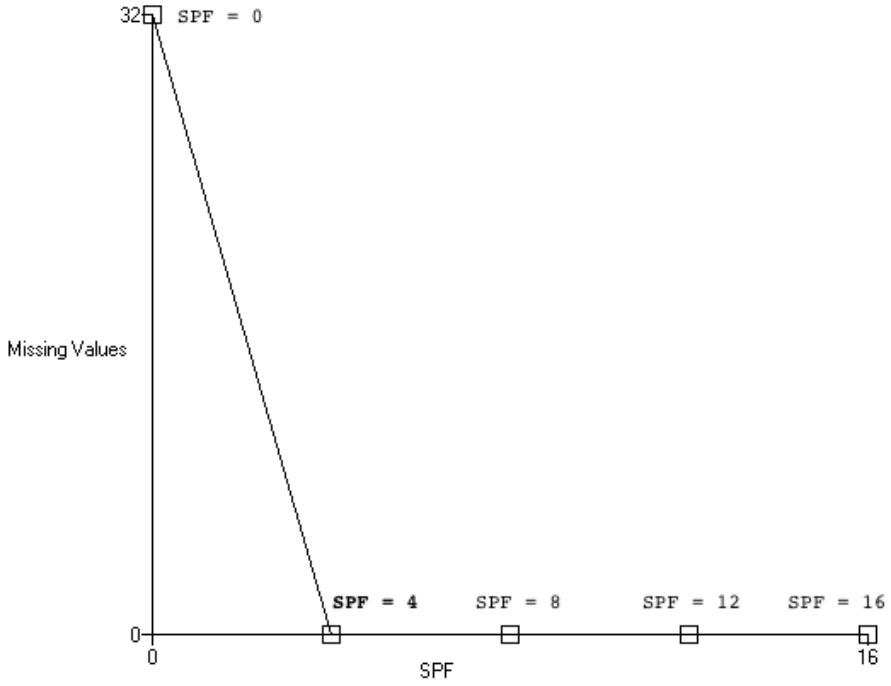


**Figure 4.5:** For the scenarios which used the area as a proxy, the appropriate SPF value was set to 4. This value was determined as the one as of which there were no more missing values. The same method was used for when the fishing intensity was used as a proxy for cost.

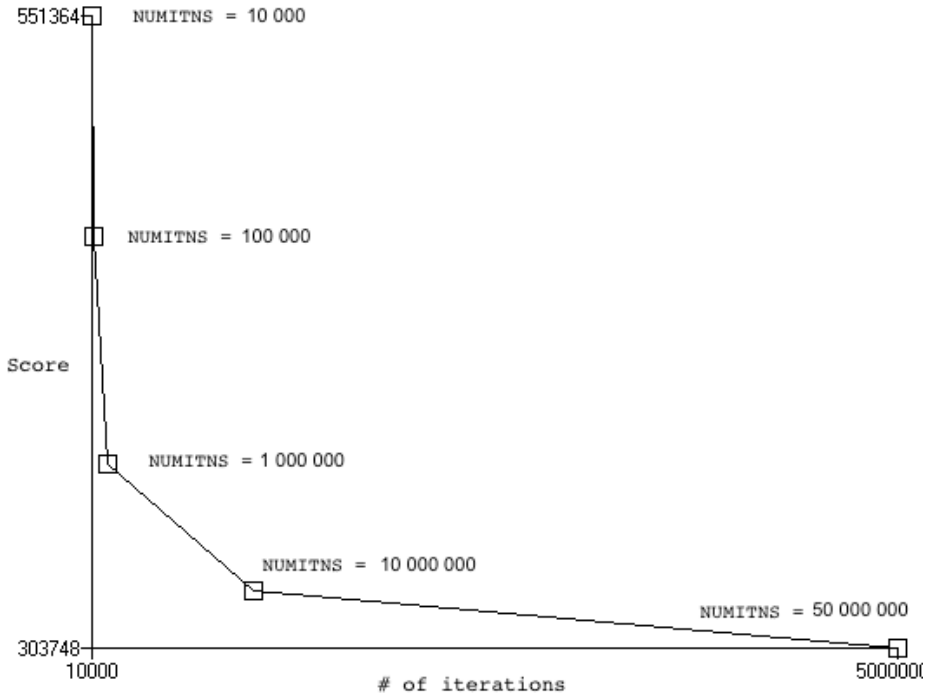


**Figure 4.6:** The appropriate number of iterations ('NUMITNS') was determined by finding the trade-off between execution time and efficiency. The value of 10 million iterations was chosen.

## 4.3 Results

The results exposed in this chapter were produced with the aim of identifying reefs which are consistently selected as priority areas across our considered timeline. These priority areas may vary depending on the way we formulate the problem to Marxan, which is why we considered several scenarios. The main results from the different scenarios will be presented, with always the aim of targeting those consistent reefs and in particular, identifying the effect of integrating connectivity.

Two standard outputs are used to visualize the results: the "best solution" file and the "summed solution" file. The "best solution" corresponds to the run which had the lowest objective function value compared to all the other runs (in our case, 100 runs in total per scenario). The planning units that are part of that final 'best' reserve are given a value of 1 and the others a value of 0. However, Marxan designers warn users that this 'best' solution should never be interpreted as *the* answer to the problem, but rather as one among many possible good ones. When performing simulated annealing, Marxan selects planning units in a pseudo-random way which means each run is likely to be different (Game and Grantham, 2008). The "summed solution" file provides the number of times each planning unit was selected in the final solution across all runs (i.e. selection frequency of each planning unit). The planning units which are the most often selected can be considered as the ones most useful for designing an efficient reserve system (Game and Grantham, 2008). In other words, a planning unit which is selected almost all the time can either mean that it contains features that cannot be found in any other site or that it contains features that are found in other sites but not at such a low cost which makes that particular planning unit more efficient and possibly irreplaceable.

Since the aim here is to identify the reefs which were consistently selected throughout the timeline, the 'best' results for each event of that timeline were added together. Since five events were considered, a planning unit which was selected across each event will have a value of 5 and if it was never selected, it will have a value of zero. This indicated the selection frequency over the years. For the 'summed' solution, an additional step was performed before adding together the results across the timeline. For each event, a value of 1 was attributed to planning units which were selected more than 70% of the time and a value of '0' to the others. The target of 70% was chosen as this is the threshold that was cited as an example by Game and Grantham (2008) for selecting commonly used sites. The results were then added together across the timeline and a planning unit which had a value of '5' meant it had been selected more than 70 times over the 100 runs to be in the final reserve system for all five of the events. This indicated the selection frequency over the number of runs as well as over the years. After manipulations, the results from both the best and the summed solutions were communicated using a color scale with darker areas corresponding to the most selected planning units and the lighter ones to the rarely selected ones.

**Assessing the influence of cost**

Seeing that many scenarios were considered, when analysing the results, several observations could be made. The first crossroad in the designing of scenarios was the consideration of existing MPAs or not. Let us first examine the case where existing MPAs were not included in the design. The first and most conspicuous observation to be made was the effect of the cost data on the spatial design. This was the second crossroad in our scenarios. The results, whatever representation scenario was considered (i.e. 'bioregions', 'restoration' or 'conservation'), all shared the same statement. That is, when the area is used as a proxy for cost, the reserve system designed is highly fragmented and spread out across the whole GBR (Fig. 4.7(A)). Conversely, when the fishing intensity is used as a proxy, the reserve produced is much more clumped, although still spread out across the GBR (Fig. 4.7(B)). Although we expected the change in cost data to have an influence on the results, such a strong contrast was not expected. Nevertheless, with the objective function in mind, these results make sense. Indeed, Marxan aims to minimize the overall cost of the reserve system. When the proxy for this cost is the reefs' area, Marxan will preferably choose small reefs over larger ones whenever that choice is possible. This resulted in small priority areas scattered across the GBR, in comparison to larger ones selected under the fishing cost.

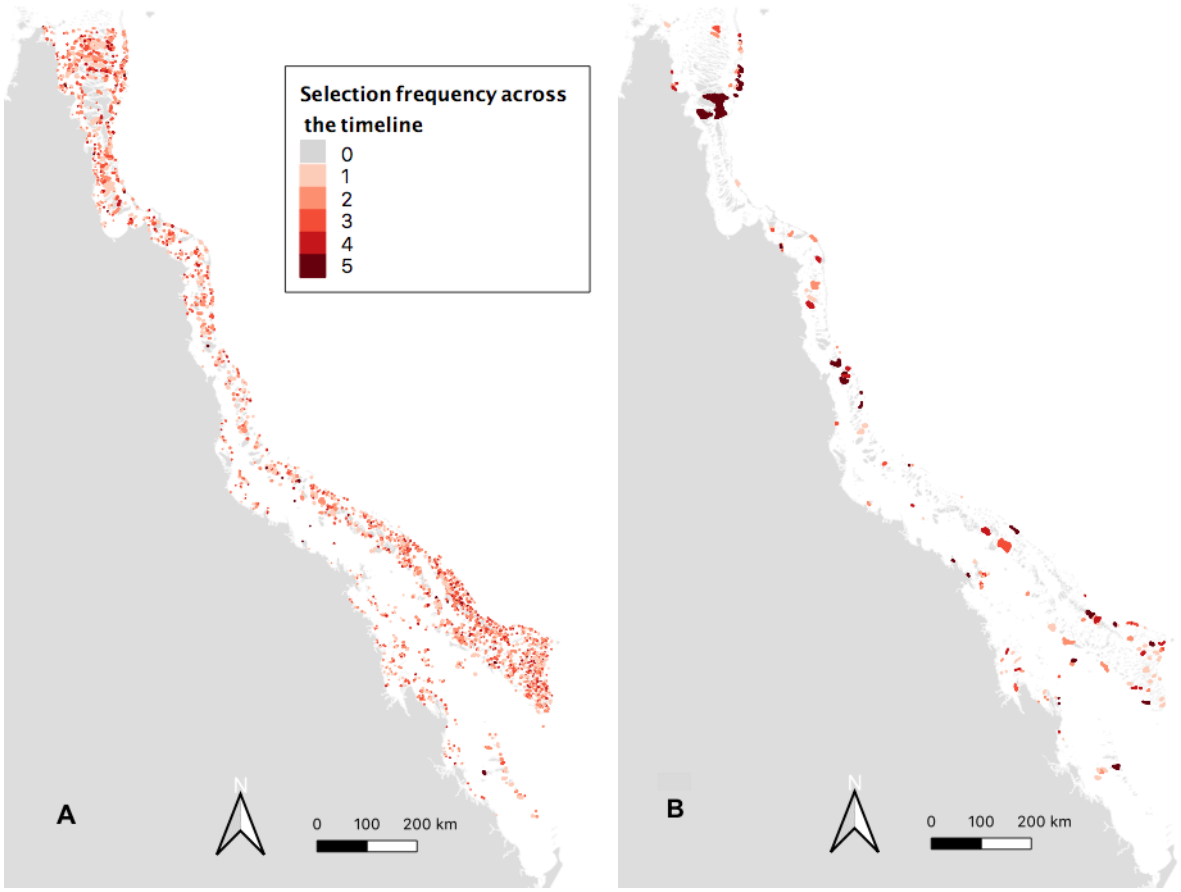


**Figure 4.7:** Results from the 'best' solution for the bioregions scenario without connectivity considered were produced for the different cost data: **(A)** Area as a proxy for cost; **(B)** Fishing intensity as a proxy for cost.

When connectivity was added to the equation, the reserve system associated to the cost using area hence showed an increase in the level of clumping (Fig. 4.8(A)). Although the contrast with the reserve associated to the fishing cost is still visible (Fig. 4.8(B)), it is less pronounced than when connectivity was not considered. Fig. 4.8(A) also shows more consistency in the selection of reefs over the years. Indeed, reefs in Fig. 4.7(A) seemed to be chosen quite randomly as they are mostly orange with no distinct gradient in the selection frequency. Fig. 4.8(A) on the other hand shows more reefs with high selection frequencies (i.e. dark coloured reefs), indicating the areas which seem to act as common ground over the years.

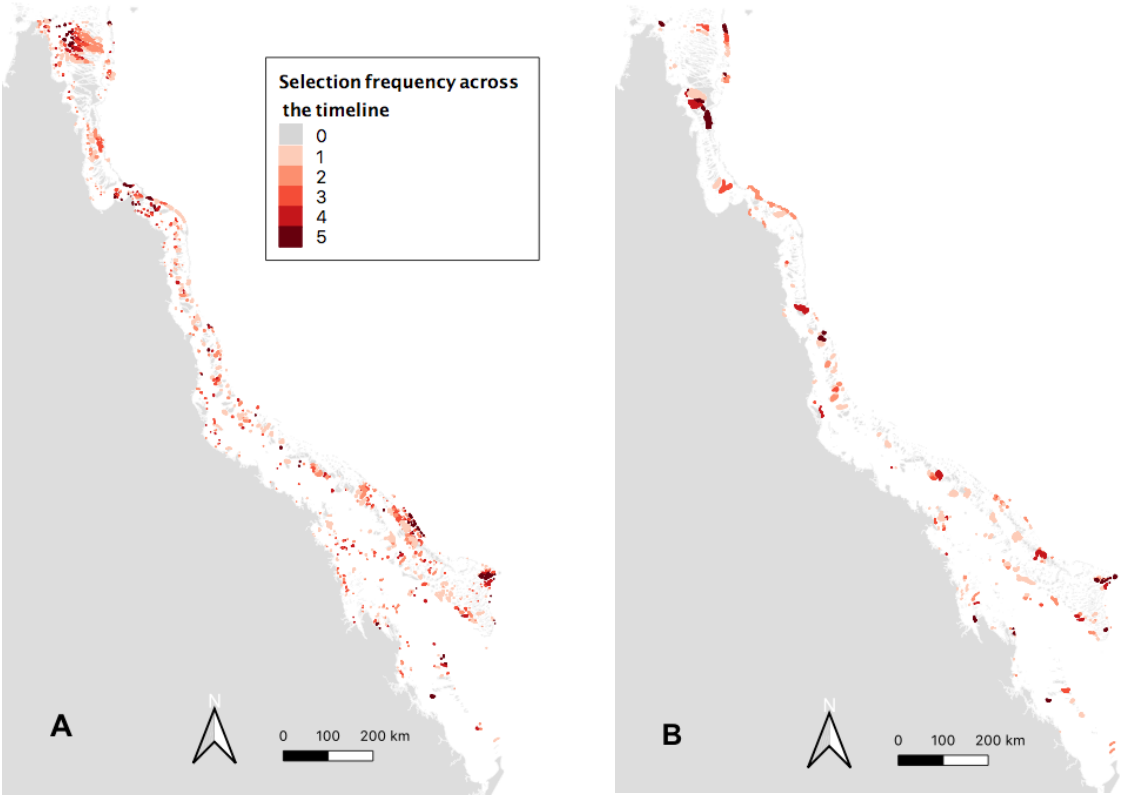


**Figure 4.8:** Results from the 'best' solution for the bioregions scenario with connectivity considered were produced for the different cost data: **(A)** Area as a proxy for cost; **(B)** Fishing intensity as a proxy for cost.

**'Restoration' and 'Conservation' scenarios without connectivity**

Now that the level of influence of cost on the selection of priority sites has been established, let us see how the different representation scenarios influence the results. This was the third crossroad in the implementation of scenarios. The 'bioregions' scenario was already illustrated in Fig. 4.7 and Fig. 4.8. For the 'restoration' and 'conservation' scenarios, the area covered by the selected reefs roughly doubled in comparison to the 'bioregions' scenario. This increase in area was expected seeing that coral cover features were considered in addition to the bioregions. Without the connectivity considered, the restoration scenario required a larger reserve area (1 433 530 ha) than the conservation one (1 237 140 ha) which makes sense since the area covered by the restoration targets was larger than that of the conservation targets. The results for the restoration scenario do not show a distinct spatial pattern in the selection of reefs over the years with most of the GBR under MPAs (Fig. 4.9(A)). The results for the conservation scenario on the other hand seem to indicate a slight

tendency for clustering of the consistent reefs towards the two poles of the GBR (Fig. 4.9(B)).

**'Restoration' and 'Conservation' scenarios with connectivity**

Let us now see how these results change for restoration and conservation when connectivity is considered, which was the final crossroad in our scenarios' structure. When connectivity was integrated in the design, a pattern in the selection of consistent reefs appeared much stronger. The restoration scenario which showed no strong consistency in the selection of priority sites at first when connectivity was ignored now shows several dark patches of reefs which seem to be consistently selected over the years (Fig. 4.10(A)). As for the conservation scenario, the pattern which was already somewhat distinguishable before connectivity was considered is now strongly highlighted, with the largest patches of reefs clustered in the north and the south of the GBR (Fig. 4.10(B)), with most of the highest selection frequencies in the south. When observing Fig. 4.10, it seems the restoration scenario covers a much wider area of the GBR than the conservation scenario. However, both scenarios cover a roughly similar area of the GBR (1 278 970 ha for 'restoration' against 1 172 760 ha for 'conservation'). Although the results visible on Fig. 4.9 and Fig. 4.10 are related to when the area was used as a proxy for cost, comparable patterns were visible for when the fishing cost was used with similar areas selected. For the restoration scenario, these important areas seem to include the reef belt above Lizard Island (roughly from Ribbon Reef No 10 to Tydeman Reef), reefs in the Whitsundays Region, reefs above Mackay Region and reefs at the top of Swains Region. For the conservation scenario, important areas included reefs in Capricorn Bunker Region, reefs in most of the Mackay Region, reefs at the southern tip of Swains Region and a big part of the northern region of the GBR.

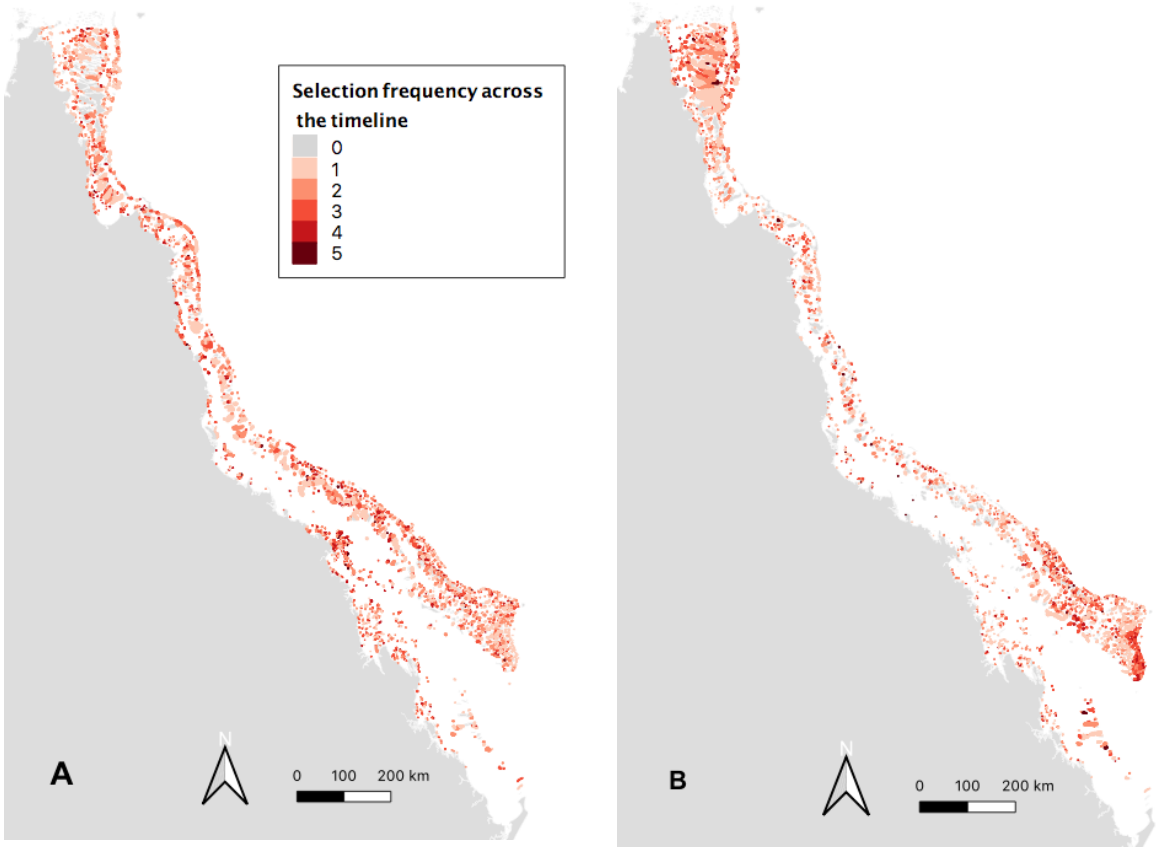


**Figure 4.9:** Results from the 'best' solution using the area as cost without connectivity considered are shown for: **(A)** The 'restoration' scenario; **(B)** The 'conservation' scenario.

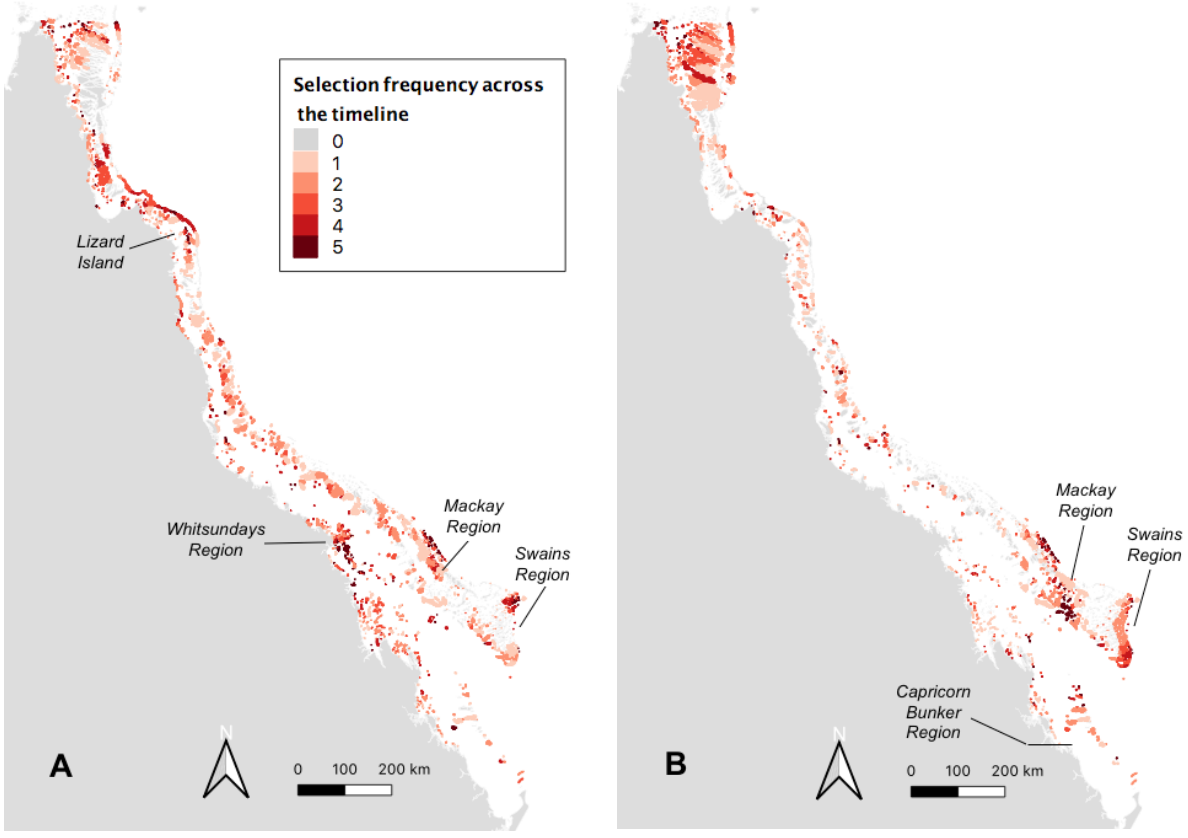


**Figure 4.10:** Results from the 'best' solution using the area as cost with connectivity considered are shown for: **(A)** The 'restoration' scenario; **(B)** The 'conservation' scenario.

The results presented above are related to the 'best' solutions. As mentioned earlier, it can also be interesting to look at the 'summed solution' output which was used to assess the reefs which were selected more than 70 times out of the 100 runs and this across the whole timeline. The results from the summed solution show similar trends to the ones of the 'best solution' but with much fewer reefs selected (Fig. 4.11). We can see that the fewer sites selected with the summed solution correspond for the most part to the reefs that were selected over most years with the 'best' solutions (dark areas on Fig. 4.10), indicating the most useful reef units.

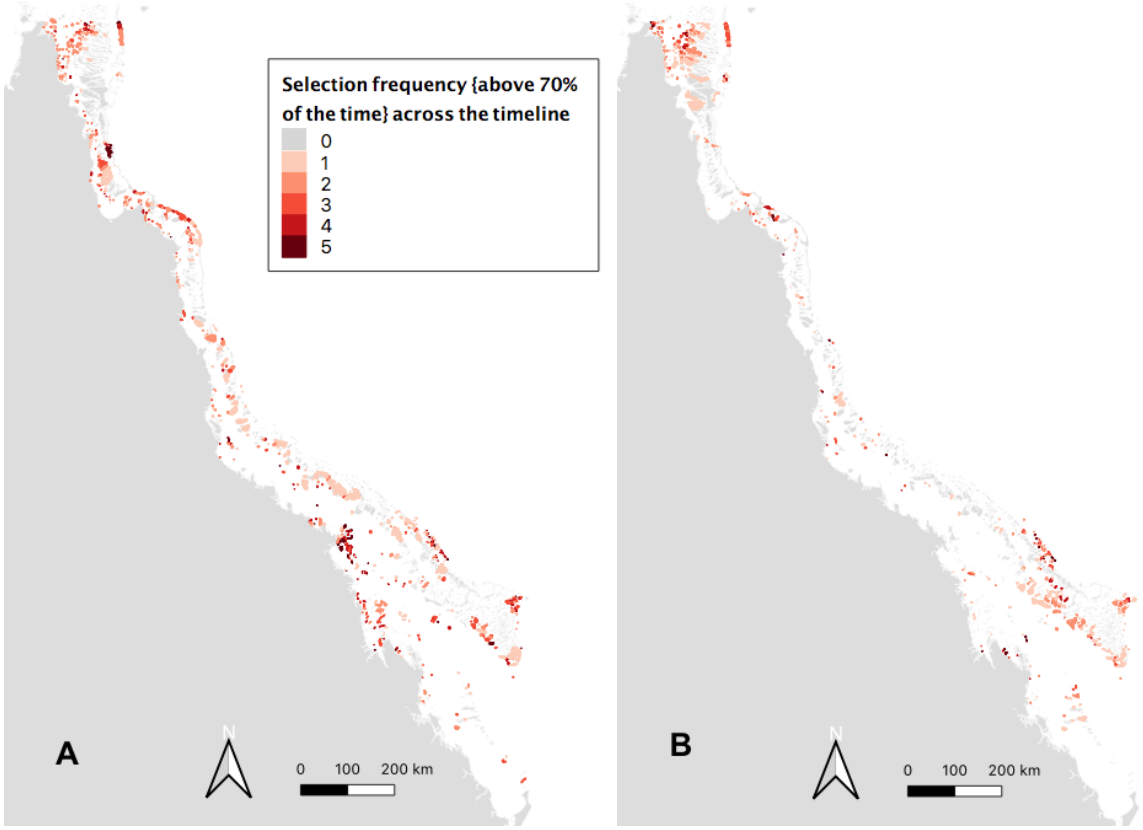


**Figure 4.11:** Results from the 'summed' solution using the area as cost with connectivity considered are shown for: **(A)** The 'restoration' scenario; **(B)** The 'conservation' scenario.

Although these maps, whether they are from the 'best' or the 'summed' solutions, reveal visible trends in the location of consistent reefs, they do not indicate whether Marxan has achieved the given targets or not. For this, Fig. 4.12 indicates for each year the level of achievement reached by Marxan for the coral cover features. While Marxan was able to achieve the targets for the bioregions across every scenario, this was not the case for the restoration and conservation scenario. Indeed, Fig. 4.12(A) indicates that the restoration target for the 'Very low coral cover' feature was not met for the post-1998 event. For that particular event, Marxan was not able to find many reefs which had less than 15% coral cover. Fig. 4.12(B) indicates that the conservation targets for both the 'High coral cover' and 'Very high coral cover' features could only be met for the first three events of the timeline. In both cases, this lack in features representation was translated by a much lower cost for the corresponding events.

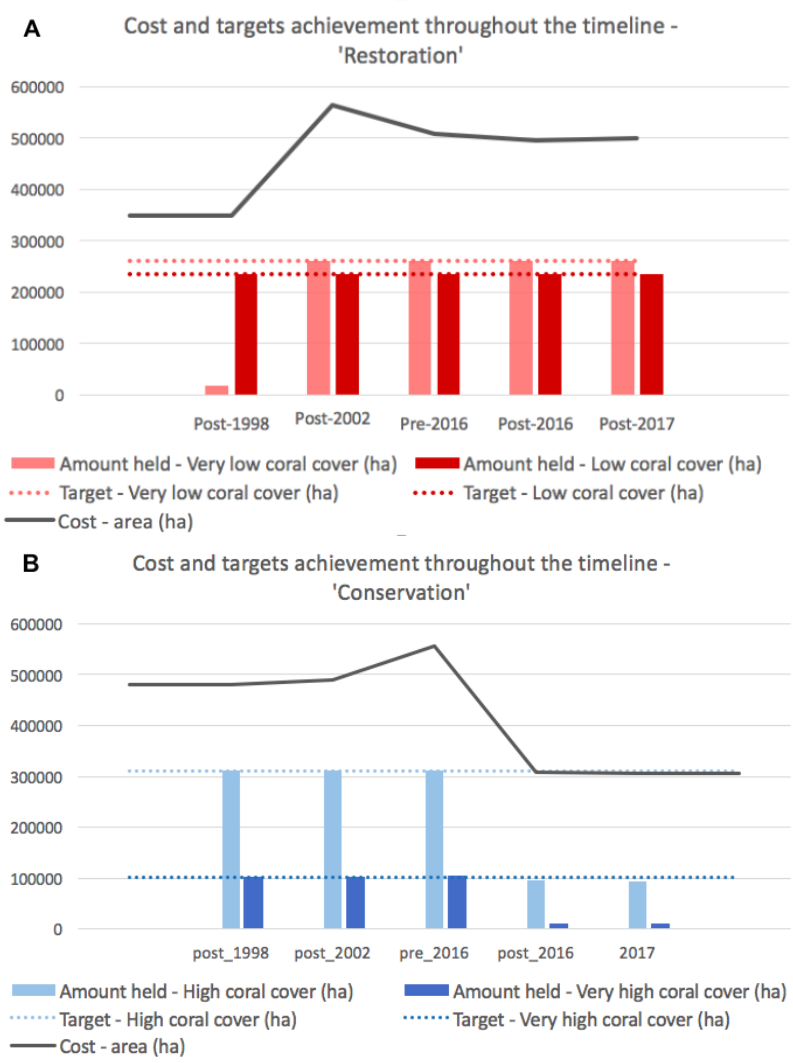


**Figure 4.12:** Cost and level of achievement reached by Marxan when connectivity is considered for : **(A)** The 'restoration' scenario; **(B)** The 'conservation' scenario.

To visually summarize which planning units are most strongly related to either the restoration or the conservation scenario, 'difference' maps were established (Fig. 4.13). These difference maps illustrate the preference of a reef to either be selected for restoration purposes -in which case it will lean towards the red- or conservation purposes -in which case it will lean towards the blue. Reefs with a color in between the red and the blue are those that were selected for both scenarios an equal amount of times. Reefs which have no colouring at all (i.e. same grey color as the mainland) are the ones which were never selected for either of the two scenarios.

The visual assessment of these difference maps enabled us to take a step back and have a look at the bigger picture. These maps reveal three main observations. Firstly, whatever cost data is used, whether it is the area (Fig. 4.13(A) and (B)) or the fishing intensity (Fig. 4.13(C) and (D)), they both agree on the general trend of the reefs' selection. This leads us to our second main observation which is the conspicuous trend itself. This trend indicates that reefs which were most often selected for conservation purposes are located in the far northern and the far southern part of the GBR. With a closer look, additional 'conservation' reefs, less noticeable but nevertheless present, are sparsely located in the central part of

the GBR among many 'restoration' reefs. Indeed, the reefs which were most often selected for restoration purposes are mostly located within the central part of the GBR, although extending quite far out towards the poles of the GBR. The third main observation that can be made from these maps pertains mainly to the case where the area is used as a proxy. When connectivity is integrated into the design of the reserve system, it appears Marxan selects planning units in a less random manner over the years with consistently targeted reefs emerging in both restoration and conservation scenarios (Fig. 4.13(B)).

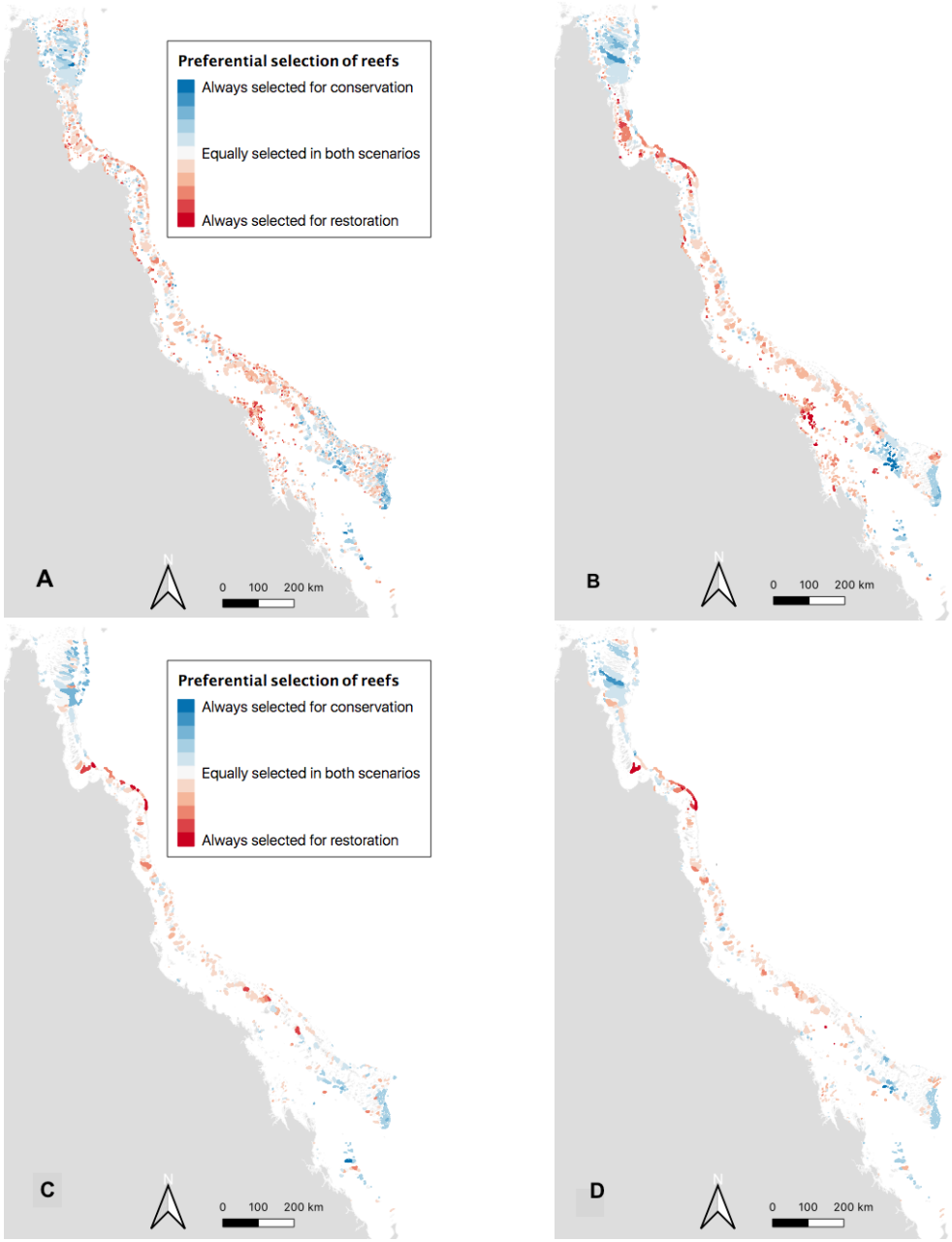


**Figure 4.13:** 'Difference' maps illustrating the preferential selection of reefs for either restoration or conservation purposes when: **(A)** Connectivity is not considered with area as cost; **(B)** Connectivity is considered with area as cost; **(C)** Connectivity is not considered with fishing intensity as cost; **(D)** Connectivity is considered with fishing intensity as cost. These difference maps were based on the 'best' solutions from Marxan.

**Considering the existing network of MPAs**

The results presented so far pertain to the case where existing MPAs were not considered. However, it can be interesting to assess the closeness of our results with the currently established MPAs of the GBR. Our 'bioregions' scenario is the one that fitted the best with the existing MPAs, which makes sense since these bioregions were also used as conservation features by Fernandes et al. (2005). The 'representation' and 'conservation' scenarios on the other hand required more reefs to achieve the respective targets. Three main observations can be made from the comparison with the existing MPAs when connectivity is not yet considered. Firstly, whatever the representation scenario, using the fishing intensity as cost produced results that were the closest to the current reserve design. This is especially noticeable for the 'restoration' scenarios (Fig. 4.14(A) and (C)) whereas the contrast is more subtle for the 'conservation' ones (Fig. 4.14(B) and (D)). Secondly, the spatial patterns of selection frequency for both scenarios have similar tendencies to those produced when the existing MPAs were not considered. Thirdly, Fig. 4.14 indicates that the current MPA network seems to be more in line with the conservation results than the restoration ones and this, regardless of the type of cost data used. The best fit with the existing design is the case when fishing intensity is used as cost data under the conservation scenario (Fig. 4.14(D)).

When connectivity is integrated into the design (Fig. 4.15)), four additional observations can be made. First of all, most of the reefs identified as priority sites have become much more consistent with a maximum selection frequency of 3 (not 5 since here we considered only the years of the timeline that followed the 2004 rezoning). Secondly, the difference in results from both cost data is less visible with much more common ground established between them. This is comparable to the same observation that was made when the existing MPAs were not considered and connectivity was added. Thirdly, the contrast between the 'restoration' and 'conservation' scenarios is less pronounced, especially for the fishing cost (Fig. 4.15(C) and (D)). Both representation scenarios seem to agree on the areas to protect which leads us to our fourth and final observation which is that these specific areas tend to be clustered around the already existing MPAs. Indeed, except for the reef belt over Lizard Island in the 'restoration' scenario and the reefs in Capricorn Bunker Region in the 'conservation' scenario, most of the consistently selected reefs are closely located to the reefs already within MPAs, forming a sort of buffer around them.

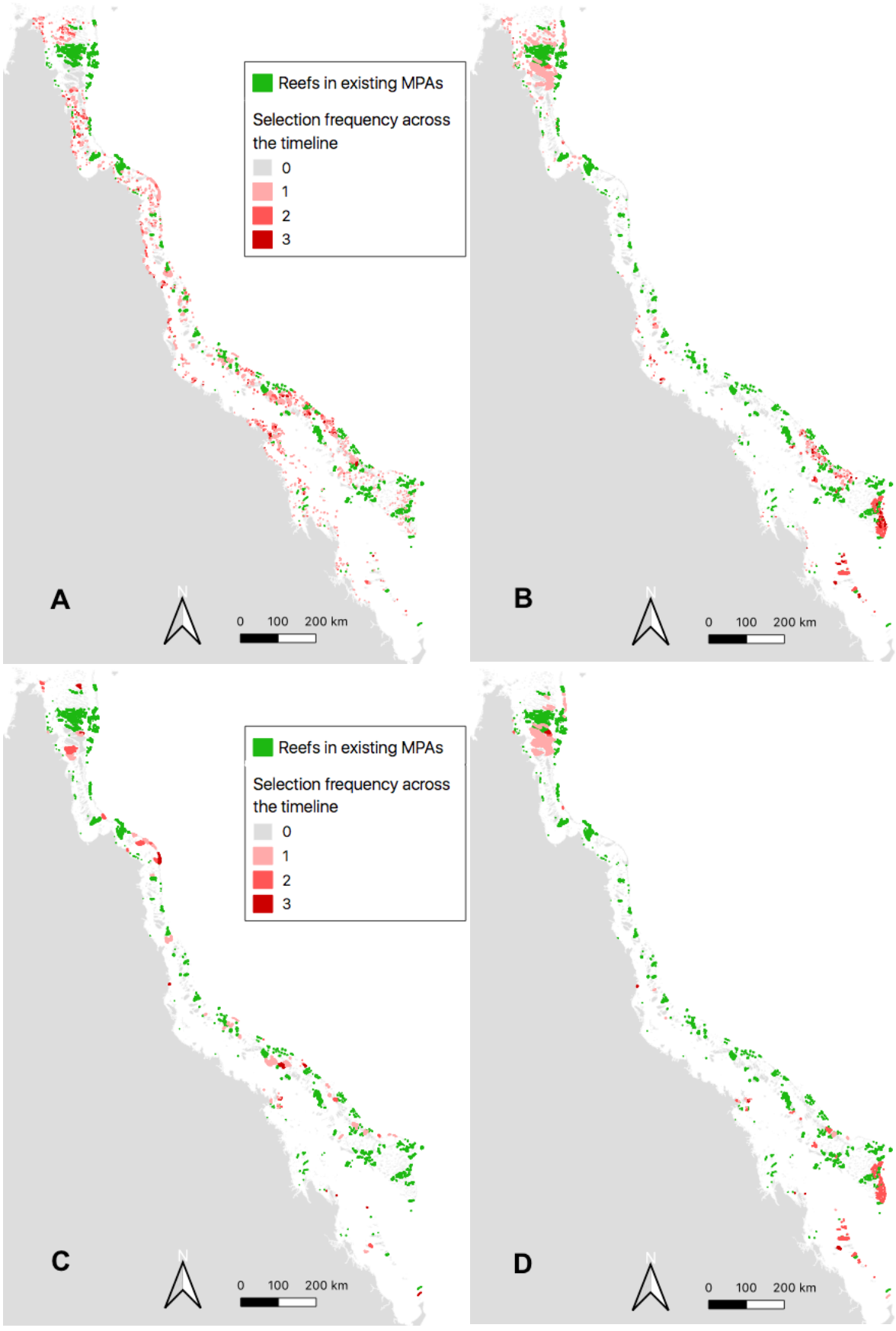


**Figure 4.14:** Results for the case where existing MPAs were considered without connectivity for : **(A)** The 'restoration' scenario with area as cost; **(B)** The 'conservation' scenario with area as cost; **(C)** The 'restoration' scenario with fishing intensity as cost; **(D)** The 'conservation' scenario with fishing intensity as cost.

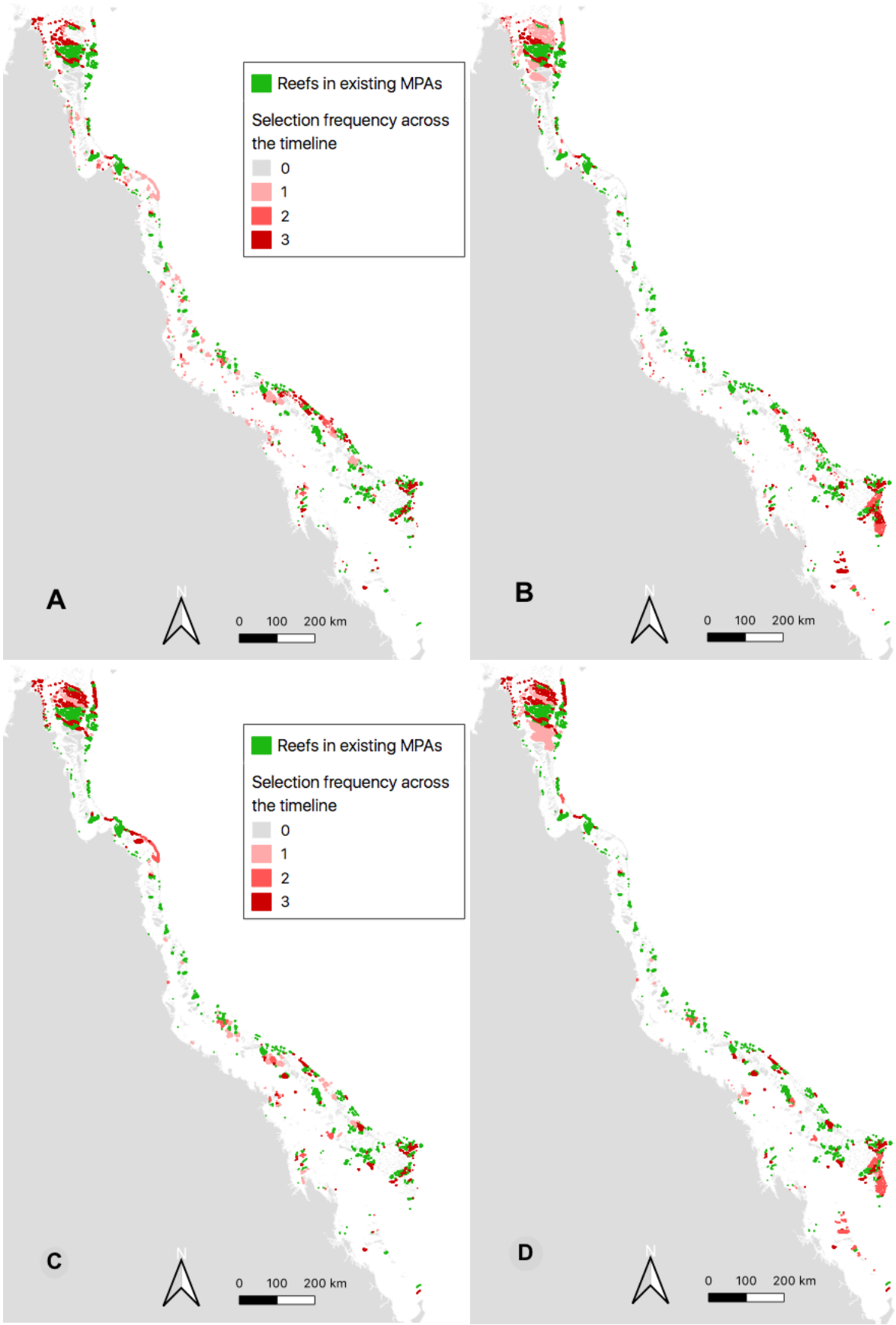


**Figure 4.15:** Results for the case where existing MPAs were considered with connectivity for : **(A)** The 'restoration' scenario with area as cost; **(B)** The 'conservation' scenario with area as cost; **(C)** The 'restoration' scenario with fishing intensity as cost; **(D)** The 'conservation' with fishing intensity as cost.

## 4.4 Discussion and conclusions

This last section concludes our study and is structured in such a way that it will start by elaborating on the above results and see if our findings can be supported by some available literature on the matter. An account as to where the current reserve design stands with regard to our results will be provided followed by the caveats as well as the future perspectives our work entails.

First and foremost, our results have enabled us to identify the reefs that are consistently selected as priority sites across our timeline, and which are therefore likely to ensure persistence due to their performance in the past. These resilient reefs only truly started standing out from the others when connectivity was integrated into the design of the reserve. Although the type of cost data used had a strong bearing on the results in terms of fragmentation, when connectivity was added as a criterion, both cost scenarios began clumping their priority sites around similar areas, identifying similar hotspots. This is interesting as it means that even though the two cost layers used were very different, a few reefs showed sufficiently strong and distinct connectivity patterns to make both cost scenarios agree on their selection for priority sites and this, across the whole timeline.

Making a distinction between the restoration and conservation perspective was enlightening as it enabled us to identify two types of reefs. The conservation scenario helped identify reefs with relatively high coral cover ($> 30\%$) remaining despite the repeated occurrences of bleaching events on the GBR. However, the reefs that managed to surpass that threshold throughout the entire timeline are scarce (only 5%), even though the map provided by Marxan (Fig. 4.9(B)) seems to suggest otherwise. This is because Marxan also had to achieve the bioregions' targets in addition to the coral cover targets and thus the reefs selected by Marxan were not exclusively targeted for their high coral cover but also for biodiversity representation. When connectivity was considered, Marxan's results were much more systematic and identified the far northern and far southern part of the GBR as the areas that could best answer conservation purposes. These results make sense when put in the context of the bleaching events. Indeed, the areas pointed out by Marxan are those that were for the most part spared by these bleaching events. Athough the northern part of the GBR seemed of high conservation value, the damage which was caused by the 2016 event induced high mortalities and a likely permanent shift of the corals' assemblage structure (Hughes et al., 2017). The highest coral cover now seems to be found in the southern part of the GBR. This clustering of priority sites at both ends of the GBR is similar to the one observed for reefs that consistently had the highest outflux degree (cfr. Chapter 3). Indeed, reefs with high reproductive output are more likely to have high coral cover since coral cover was used to weigh the potential connectivity. Hence, one might assume that combining both coral cover and connectivity data is redundant and will only emphasize one another's results. Yet our results for the restoration scenario have proven otherwise. Indeed, the restoration scenario highlighted degraded reefs with low coral cover ($< 30\%$) which were spread out across the whole GBR when connectivity was not considered. Integrating connectivity in this scenario allowed us to sort out among those scattered reefs the ones which, despite their low coral cover, nevertheless retained significantly high

reproductive outputs and this, throughout the entire timeline. The areas selected were different than those of the conservation scenario which shows that high value reef units are not necessarily confined to areas of high coral cover. In this scenario, connectivity enabled us to identify those 'hotspots' reefs which are most worth focusing restoration efforts on. It is however important to keep in mind that there may be other reefs which have just as high connectivity as these hotspots but were not selected by Marxan due to the higher cost of including them in the reserve or their inability to improve biodiversity representation or both. In this way, understanding the objective function that controls the mathematical optimization performed by Marxan helps understand its choices better which makes it less of a 'black box'. However, Marxan does not serially select planning units in the same way a designer would do by hand and some of the selected units might be unexpected. Nevertheless, Marxan, with the complexity it is given, will always operate in such a way that the whole is more than the sum of the parts, which is characteristic to reserve design (Ball et al., 2009).

The clumping performed by Marxan following the incorporation of connectivity was sufficiently high as to identify important clusters of reefs across the GBR. In relation to what was mentioned in Chapter 3, it is important to keep in mind that these reefs which appear as connectivity hotspots correspond to strongly connected intra-communities rather than strongly connected inter-communities. This was expected seeing that it had already been established that the connectivity between patches decreases as the distance between them increases (cfr. Chapter 3). The species of interest for our connectivity model was *A.Millepora* and from our results, it seems this species has a relatively short-distance larval dispersal with a few key highly self-connected patches. More importantly, we established in the previous chapter that the connectivity matrix had higher values across its diagonal elements which translates the success of local-retention in maintaining self-persistence (Burgess et al., 2014; Saint-Amand et al., 2018). In the attempt of optimizing the connectivity patterns of our species of interest, Marxan has tailored the reserve to areas which mostly promote self-persistence. Although overall connectedness is often recommended in designing successful MPAs, in the perspective of considering multiple species, White et al. (2014) found that optimizing local-retention (i.e. self-persistence) actually tends to produce better results.

Our results met on some common ground with those of Hock et al. (2017) which studied connectivity and systemic resilience of the GBR. They identified reefs that exhibit high connectivity and act as key sources on the GBR with high replenishment potential (Fig. 4.16(A)). From these reefs, they identified as robust sources the ones which were at a low bleaching risk and had low supply of COTS larvae (Fig. 4.16(B)). When comparing our results to those of Hock et al. (2017), the combination of our priority areas created under the 'restoration' and the 'conservation' scenario appear to encompass the reefs identified as robust sources by Hock et al. (2017). Although some of the regions highly selected in our maps were not highlighted in Fig. 4.16, they should not for all that be discredited. Indeed, the Capricorn Bunker Region, for instance, was often selected in our conservation scenario and was considered highly resilient by Halford et al. (2004) which may support our findings. Although our maps included additional regions, we strongly agree with the findings of Hock et al. (2017) which considered the southern region likely to be exceptionally resilient as

that is the region where most of our maximum selection frequencies were located.

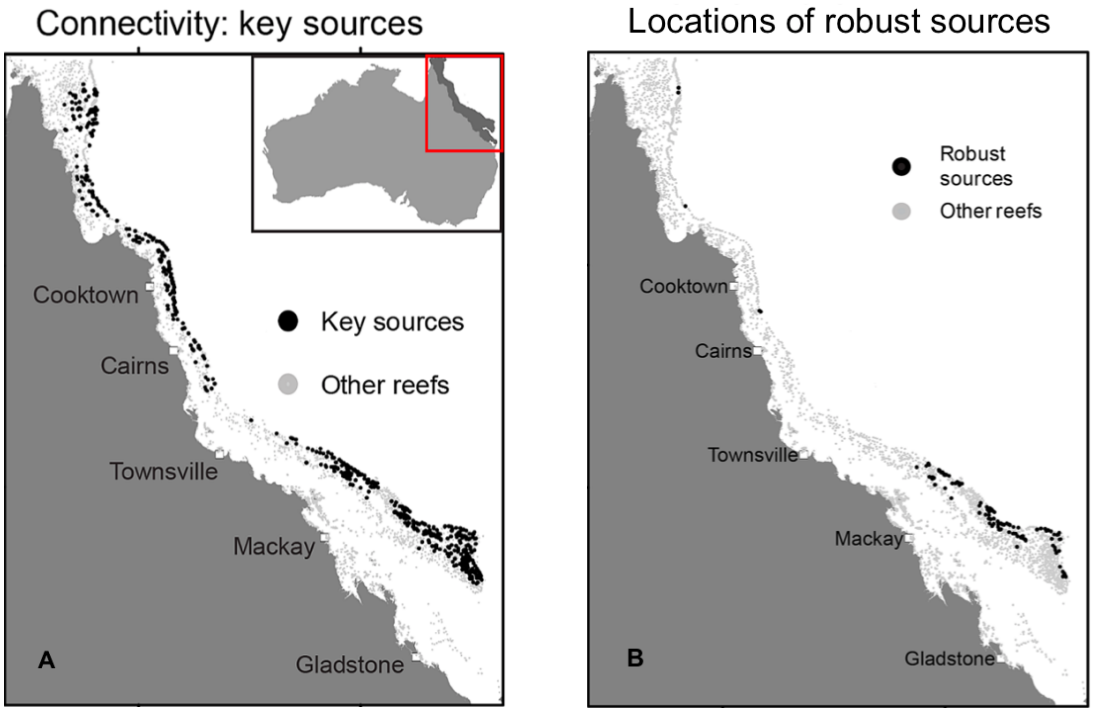


**Figure 4.16:** Results from the study of Hock et al. (2017) identifying: **(A)** The key sources for connectivity; **(B)** The robust sources left after selecting reefs which have low risks of bleaching and COTS outbreaks.

Hock et al. (2017) also found that the current zoning of the GBR already showed an inherent level of systemic resilience as it included almost half of the reefs they identified as robust sources. Our results substantiate this. Indeed, when we included the existing MPAs in our scenarios, the results produced were very close to the current zoning. Furthermore, the reefs that were added by Marxan were practically clumped around the existing MPAs. A slight expansion of these MPAs to include the selected reefs surrounding them might just improve the resilience of the existing network.

The consideration of the existing MPAs also showed that the current zoning was in better conformity with the conservation perspective rather than the restoration one. This is not surprising seeing that the context in which Fernandes et al. (2005) designed the current zoning was not yet marked by such strong manifestations of climate change than it currently is. Hence, designing the reserve system with an entirely conservation approach was appropriate. However, in the current context, the approach developed by Abelson et al. (2016), which calls for integrating degraded reefs in the design of reserve systems, seems to be supported by our results. Indeed, as shown in Fig. 4.12(B) the conservation targets fixed across the timeline (i.e. 30% of high and very high coral cover) could no longer be achieved for the years following the 2016 bleaching event. Even though the years prior to the 2016 bleaching event did manage to achieve those targets, one must not forget the target for very high coral cover represented an amount worth only 6% of our estimated baseline level. Abelson et al. (2016) had predicted the minimum fraction of healthy reefs required for future protection might not be available. Although this minimum fraction has not been scientifically established, the fact that Marxan was far from achieving the

conservation targets we had fixed for the end of our timeline indicates there is indeed a need to adopt a different approach to systematic *conservation* planning than solely focusing on healthy reefs. A future attempt in improving this work could try and find the best way of combining both conservation and restoration perspectives together in a single design without compromising too much the other objectives (e.g. cost).

For our results to be interpreted properly, they must be communicated along with the main caveats they entail. Overall, this work insisted on promoting a dynamic-based management opposed to a static one by considering temporal variations in reefs' coral cover to assess areas of higher resilience. However, truly assessing resilience requires a comprehension of the processes involved behind such changes. Yet, in this work we focused mainly on bleaching events to explain the change in patterns over time even though other factors such as COTS outbreaks, cyclones or river plumes interfered in the shaping of the reefs' resilience. Although impacts from such factors may have been accounted for through the AIMS monitoring data, the years used in this timeline were based solely on the mass bleaching events. Considering all years for which some type of stressor induced mortality on the GBR would have probably required a different approach than that adopted in this thesis due to the excessive amount of years this would have implied. The choice of using mass bleaching events as reference points was also guided by the current context of global warming.

In order to truly translate changes in connectivity patterns across the timeline, re-calculating the potential connectivity matrix for each year with the SLIM model would have been better. This would have produced more genuine results than simply weighing the potential connectivity from 2012 by the coral cover from each year. Doing so would have enabled us to assess, in addition to the changes in the strength of connections, the changes in the connections themselves and hence the overall network. This would have required collecting specific data for each of those years (e.g. tides, wind data), which would have been extremely time costly and possibly not feasible if that data is not available. Integrating a composite HQI in addition to coral cover into the weighting of the potential connectivity would have also provided more accuracy.

The method used in this work to integrate connectivity into Marxan was that of Beger et al. (2010). Other methods include using different connectivity metrics as separate conservation features in the same way Magris et al. (2016, 2017) did. This would have forced Marxan to consider different types of connectivity (e.g. centrality, outflux and local retention) at the same time. This method might have produced more comprehensive reserves in terms of connectivity patterns seeing that in our case, Marxan seemed to have especially advantaged self-persistent sites. Using connectivity metrics can be particular useful when integrating a multi-species connectivity into reserve design (White et al., 2014; Magris et al., 2016). Species representation was another one of this work's drawbacks since we only considered the larval dispersal of *A. Millepora* and it would definitely be worth considering in further research. Another reason for using connectivity metrics as an alternative method to integrate connectivity is related to the objective function. We established that the objective function guides Marxan's choices on which planning units to consider for enhancing connectivity by using a "boundary cost". Assuming two

planning units are strongly related to each other and thus have a high boundary (connection) cost, if Marxan selects one of them to be in the final reserve and not the other, the connection cost must be paid. If they are both within the reserve, that cost is not paid. However, if both sites are outside of the reserve, the connection is not paid either (Ball et al., 2009). This implies that Marxan could exclude a highly connected pair from the reserve system and yet not pay the cost for this exclusion of connectivity. This could be avoided by using connectivity metrics as conservation features.

Overall, this work could use some improvements in its approach to balancing biodiversity and connectivity targets, while at the same time satisfying different objectives under a combination of costs. However, being able to best translate to Marxan the level of complexity that lies behind such a systemic approach remains challenging, which is why simplifying assumptions are often made. Although our work involved such assumptions, it nevertheless provides an innovative approach to addressing the resilience of corals reefs in the face of global warming. Indeed, by overlaying patterns of change in reaction to mass bleaching events, we have been able to identify recurrent hotspots that seem promising in maintaining the resilience of the GBR. In doing so, we have also targeted areas of the current zoning whose expansion could possibly improve the reserve's performance. By considering temporal patterns in addition to spatial ones, we have addressed one of the major challenges in using connectivity information for designing MPAs (Krueck et al., 2017). Our maps can be used to prioritize reefs on the GBR that consistently contribute to meeting conservation or restoration targets while achieving connectivity. In times where major challenges lie ahead, learning from the past can be instrumental in helping us develop strategies. Although many projects and programs have already been put in place to protect the GBR (see GBRMPA, 2019b)), in the long term, the objective of maintaining worldwide health of coral reefs will require a global effort in curbing our emissions and meeting the ambitions of the Paris Climate Agreement ($<$ 1.5°C warming) (Donner et al., 2005; Hughes et al., 2017; Wolff et al., 2018; Hughes et al., 2018a) as this is the biggest threat to coral reefs and one which MPAs cannot prevent from happening.

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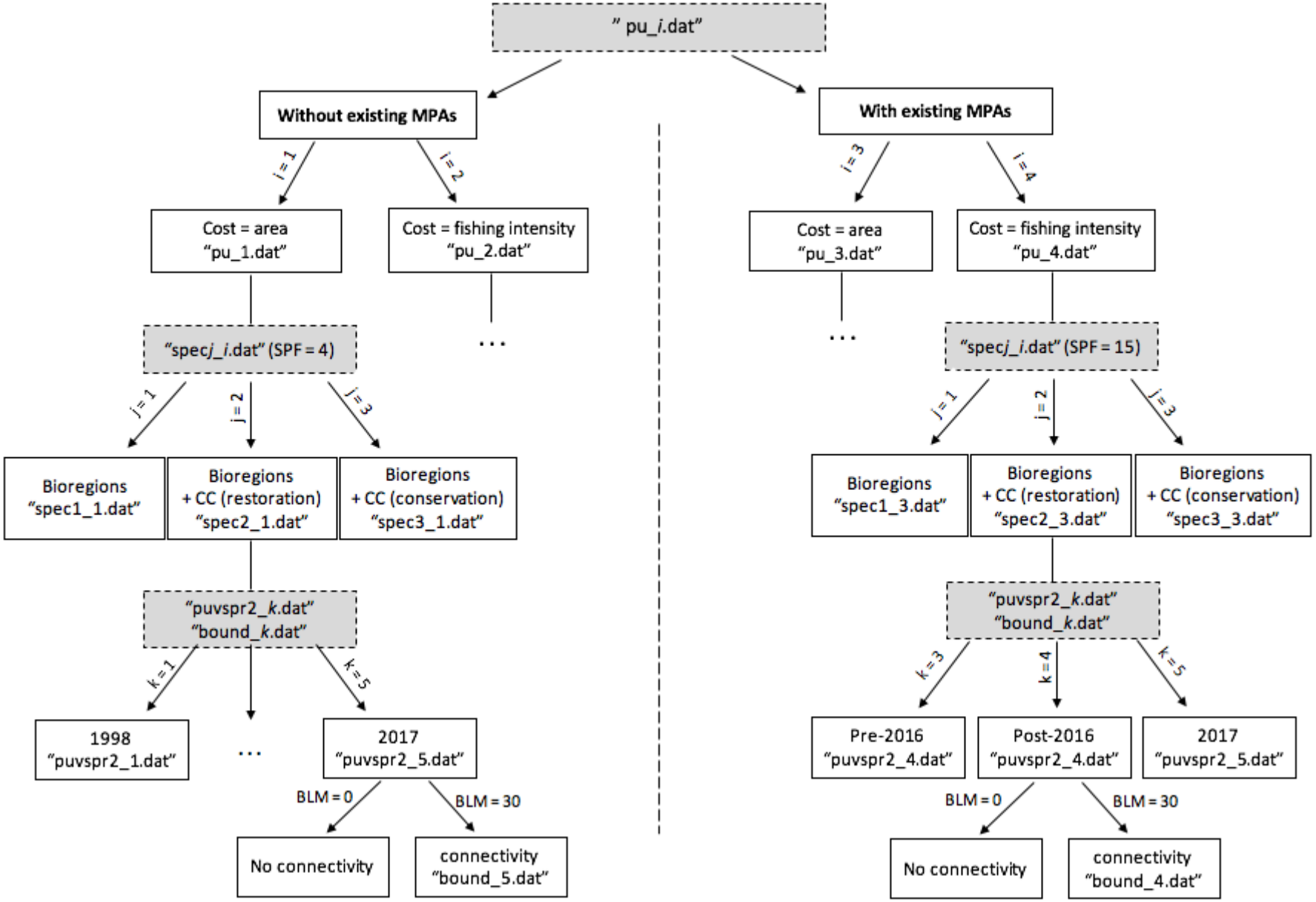
# Appendix

**Appendix A: Diagram illustrating the set up of the different input files according to the different scenarios**

Since this work involved many scenarios, they were all implemented into Marxan through the R software using a series of different loops. Marxan requires 3 specific input files to be able to run: Firstly, the 'Planning Unit file' (*pu.dat*) which gives information on the planning units' IDs, their status and their cost; secondly, the 'Conservation Feature file' (*spec.dat*) which gives information on the name, ID and target to reach for each conservation feature; and thirdly, the 'Planning Unit versus Conservation Feature file' (*puvspr*2.*dat*) which indicates the amount of each conservation feature within each planning unit. In addition to these three mandatory files, an additional file pertaining to the connectivity can be used, this is the 'Boundary Length file' (*bound.dat*). These key files are all represented in our diagram (coloured frames) and are different depending on the scenario considered (white frames). The major split in the middle of the diagram was made in order to clearly distinguish the scenarios where the existing MPAs are not considered from the scenarios where they are considered.

Appendix A: Diagram illustrating the set up of the different input files according to the different scenarios



**Integrating past and present connectivity into the design of marine protected areas to plan for persistence in the Great Barrier Reef**

Jessica Deurinck

The recent decline in coral cover worldwide has led to a global concern about the ability of coral reefs to withstand the impacts of climate change. One of the tools developed to protect coral reefs is the design of marine protected areas (MPAs) through systematic conservation planning.
In this context, our work used the Great Barrier Reef (GBR) as its study area. The aim was to identify which reefs would be most resilient in the face of global warming and therefore strategic to consider as part of MPAs. In order to do so, a retrospective method was adopted. We constructed a timeline based on the most important mass bleaching events that hit the GBR over the past decades. The reefs which maintained a certain level of resilience throughout the timeline were considered most fit to ensure persistence on the GBR.
Three main steps led to identifying these reefs. The first step involved calculating the coral cover for the $\sim$ 3800 reefs making up the GBR and this, for each event of our timeline. Coral cover was used as it is an important indicator of reef health. In addition to coral cover, we used the connectivity of corals to identify important hotspots within the network. The second step involved weighing this connectivity with the coral cover to identify temporal variations in the strength of connections between coral reefs. Finally, we integrated the information from the previous steps into Marxan, a decision support tool, to identify a set of priority areas across the GBR through different scenarios. Priority areas that were consistently selected throughout the timeline were considered best able to answer our call for resilience.
Our results show that resilient areas on the GBR only truly started standing out once connectivity was integrated into the design. The areas that showed the highest selection frequencies over the years were located mostly in the south of the GBR, indicating that this might be a region worth securing for the future. Our maps also indicate that the current level of protection of the GBR seems to already satisfy a certain level of resilience which could nevertheless be improved by expanding some areas. Our findings also indicate that the approach of considering both healthy and degraded reefs rather than focusing solely on healthy reefs when designing MPAs is more appropriate in today's context. Overall, this work highlights the importance of integrating connectivity into spatial design and more specifically, how combining past and present connectivity may help plan for persistence on the GBR.

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