

# Options for the Early Prevention of Crown-of-Thorns Starfish Outbreaks on the Great Barrier Reef

*Accelerate  
Partnership*

FINAL REPORT

Collated by

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**Cover image:** Aggregation of adult Crown-of-Thorns Starfish (*Acanthaster* sp) feeding on corals on Australia's Great Barrier Reef. Photo credit: Katharina Fabricius, Australian Institute of Marine Science.

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# Executive Summary

## The Challenge

The Great Barrier Reef (GBR) is facing pressures from multiple sources, with growing threats from climate change. Stronger storms and warming events that drive coral bleaching episodes are now adding to the suite of GBR pressures. The consequence is cumulative impacts and the challenge is to build ecosystem resilience. While climate change is beyond the direct control of reef management and policy, impacts by coral-eating Crown of Thorns Starfish (CoTS) are amenable to local and potentially regional control. Two current avenues for such control are land-use management, building on the assumption that nutrient enrichment is implicated in outbreak initiation, and the manual culling of CoTS. However despite continuing efforts and investments from science and management to reduce CoTS impacts, CoTS have accounted for around 40% of the decline in coral cover on the GBR over the past 30 years. Going forward in a time of climate change, there is now an urgent need to identify the most effective CoTS management strategy.

A central question is whether CoTS outbreak initiation can be prevented or the risk of outbreak initiation reduced. Once an outbreak is fully developed, it becomes the equivalent of a wildfire. Management objectives then shift from protection of all GBR coral reefs to local and regional damage-control and a focus on the protection of individual high-value reefs. To inform policies that focus on whole-of-GBR protection, this project analyses the efficacy and cost-benefits of early outbreak prevention and/or containment using mixed strategies of land-use management and CoTS control.

## Objectives and Approach

Current management of CoTS is aimed at limiting the severity of damage stemming from an established outbreak wave through targeted surveillance and cull operations. In contrast, this project explores options for reducing the *risk* of outbreak initiation and containing primary outbreaks. We sought to develop clear policy advice on whether losses from CoTS can be reduced by lowering the risk of outbreak primary outbreaks, and if so, then what are the benefits and costs of interventions?

To tackle this challenge the Partnership brought together a team of biologists, mathematical and statistical modellers, oceanographers, economists, decision scientists, marine park managers, marine tourism industry partners and a regional natural resource management group.

The team explored the effectiveness of a range of options currently available to managers under two overarching strategies: (a) reducing the incidence of nutrient-rich riverine discharge into the area identified as the source of primary outbreaks via land use change, and (b) arresting the spread of outbreaks in the initiation phase via scaled-up surveillance and cull operations. We analysed these options as parts of two lines of early defence analogous to a pest or fire management strategy. Our approach involved the development of models linking CoTS and corals, integrated with analyses of the links between land use and CoTS outbreak risk, a novel connectivity network model, and a bio-economic study of the benefits versus costs of different management options.

## Key Results and Policy Recommendations

Estimates of the costs of implementation varied across strategy options from \$52 million to \$3.5 billion. For land-use changes, the least costly option involved widespread adoption of B-standard practice and the most expensive alternative involved retirement of all cane land and conversion to biodiversity conservation. For surveillance and cull options to prevent or contain a primary outbreak, we explored scenarios involving as few as 1 surveillance and 1 cull boat and as many as 20 surveillance and 40 cull boats.

Results were unambiguous. Two options for land-use change in Wet Tropics basins represent the greatest value-for-money with the objective of preventing outbreaks.

- *Option 1. At least all B level practice in all areas of sugar cane production throughout Wet Tropics basins.* B level refers to use of best available farming practice for soils and nutrients to reduce risks to water quality. At a cost of \$51.9 million this option was estimated to increase the mean return time of CoTS outbreaks from 15.9 years to 17.3 years. Although a modest extension in return time, the benefits in terms of improved coral cover over a 30-year time horizon were estimated to exceed costs under a range of plausible estimates for credibility of nutrient-enrichment as a cause of primary outbreaks, adoption rates among farmers, the response of coral cover to an increase in bleaching risk, and the total economic value of the Great Barrier Reef.
- *Option 2. Retire all Johnstone and Tully-Murray sugar cane production to low-input grazing and at least B*

practice elsewhere throughout Wet Tropics. This option was estimated to extend the mean return time of CoTS outbreaks to 17.8 years at a cost of \$244.6 million. Again, benefits consistently exceeded costs across a range of plausible scenarios.

In the context of an objective of reducing the risk of a primary outbreak, investment in the best of the CoTS surveillance and cull options in the initiation and early outbreak stage was a more marginal proposition. Reasons for the relative underperformance of surveillance and cull options as a means of preventing or containing a primary outbreak were threefold: (1) challenges associated with detecting and killing CoTS, especially juveniles and young adults, (2) the extreme fecundity of adult CoTS and (3) dispersal capabilities of larvae. Under an unrealistic assumption of perfect detectability of adults, intensive operations using a single pre-outbreak surveillance boat and 6 post outbreak initiation cull boats would increase the mean return time of CoTS outbreaks from 15.9 years to 26.4 years, at an estimated cost of \$199.5 million. But when we account for the reality of imperfect detectability, the same management scenario extended return times to just 16.0 years - a negligible benefit relative to the status quo scenario of 15.9 years. Even with sufficient resources to fund 20 surveillance boats and 40 cull boats (at an estimated cost of \$3.5 billion), the mean return time under imperfect (and currently realistic) detectability was less than 17 years.

We emphasise that this assessment of surveillance and cull options relates only to the objective of arresting outbreaks in the initiation phase. Our analyses did not extend to exploration of the effectiveness of options for protecting high-value reefs or reducing the rate of spread or intensity of downstream secondary impacts. Other related projects



under the National Environmental Science Program (NESP) are demonstrating an important role for surveillance and cull operations beyond the initiation phase. The complementarity of these projects with this Accelerate Partnership form a strong case for an integrated CoTS management framework and decision-support system for the entire outbreak cycle and the entire GBR system and catchments.

In conclusion, we recommend intensified efforts to improve practices of sugar cane producers in Wet Tropics Basins as a matter of high priority. While the benefits of this investment are expected to far exceed costs, we emphasise that the episodic alignment of environmental conditions (e.g. ENSO, oceanography) and ecosystem state (coral cover) can still align to promote CoTS outbreaks despite optimal land use. Therefore, recognising the acute need to maintain ecological resilience and promote post-disturbance recovery in a warmer climate, we recommend parallel investments in surveillance and cull operations at all outbreak stages. To aid their effectiveness, we recommend adaptive investments in research and development to substantially enhance starfish detectability during cull campaigns.

## ***Where to from here?***

The policy advice provided by this project has only been possible through a collaborative Partnership of researchers and managers and other reef professionals. We have drawn on the expertise and experience of a range of partner organisations in synthesising biological understanding, operational realities, social feasibility, and economic analyses. In Section 8 of this report we identify a number of avenues for building further on this multidisciplinary partnership approach to advance technologies and strategies capable of improved outcomes in the future.



*Five-month old juvenile CoTS (5 mm disc) grown in AIMS' Sea Simulator. Photo credit: Charlotte Johansson, AIMS*

# I. Introduction

The coral-eating Crown-of-Thorns Starfish (CoTS, *Acanthaster* sp) is a native species complex on Indo-Pacific coral reefs and often occurs at background densities that represent low environmental risk, i.e. coral growth and recovery rate exceed predation rate by CoTS (Birkeland and Lucas 1990, Pratchett et al. 2014). Episodically, mass population outbreaks of CoTS cause large-scale coral mortality (Moran et al. 1992, Uthicke et al. 2009, Baird et al. 2013) with consequent loss of habitat and resources for many species (Kayal et al. 2012, Emslie et al. 2014).

On Australia's Great Barrier Reef (GBR), CoTS caused around 40% of recorded loss in coral cover from 1985 to 2012 (De'ath et al. 2012). Despite 40 years of research and management, the CoTS challenge persists on reefs across the Indo-Pacific as starfish population dynamics and the effectiveness of management actions remain uncertain.

Contrary to climate-driven pressures, such as coral bleaching and storms, which are not under the direct control of regional or local management agencies, CoTS are considered to be amenable to interventions. The Association of Marine Park Tourism Operators (AMPTO) under contract by the Great Barrier Reef Marine Park Authority (GBRMPA) have culled over 500,000 adult CoTS from the northern GBR during the current outbreak (GBRMPA 2016). Their effort aims to limit the severity of the current outbreak by reducing adult coral mortality on highly valued and connected reefs to provide Reef wide recovery benefits.

Longer-term strategies to build reef resilience include efforts to reduce nutrient run-off from the land and establishing marine reserve networks (Fernandes et al. 2005, Commonwealth\_of\_Australia 2015). A linked management objective under these strategies is to lower the likelihood of CoTS outbreak

initiation via two presumed processes: reducing nutrients that promote CoTS larval survival (Fabricius et al. 2010b) and increase the abundances of natural CoTS predators (Sweatman 2008, Cowan et al. 2017).

## I.1 Purpose of this Accelerate Partnership

This project was borne out of a shared interest in tackling what is one of greatest challenges for GBR management and policy. What has been lacking from decades of research and management efforts has been clarity around the efficacy and pay-offs of different management and investment strategies. Part of the unknowns in this equation has been lack of sufficiently resolved CoTS population and connectivity models, a deeper understanding of the constraints and uncertainties around CoTS monitoring and cull operations, and transparency around the benefits and costs of various strategy options. This project brought together a multidisciplinary team of coral reef biologists, ecosystem modellers, oceanographers, economists, decision scientists, marine park and natural resource managers in a Partnership to tackle this challenge.

The overarching challenge of CoTS management comprises two constituent challenges. Firstly, how can we reduce the risk of CoTS outbreak initiation? Secondly, how can we reduce the severity of outbreaks? We focused in this Partnership on options for reducing the risk of outbreak initiation, or containment in the initiation phase. A related NESP project conducted by four of the Accelerate Partners (RRRC, GBRMPA, AMPTO and UQ in collaboration with CSIRO) focuses on improving tactical responses to reduce the severity of outbreaks once they are underway.



## 1.2 Reducing the risk of CoTS outbreak initiation - What's at stake?

The urgency of reducing the risk of CoTS outbreak initiation is heightened when considered alongside the threat from climate change, including the scale and intensity of recent bleaching events (Hughes et al. 2017). In Section 6.2 we describe a simple model linking outbreak initiation to changes in coral cover under climate change to characterise the benefits of management intervention. **Figure 1** shows the model prediction of coral cover for the period 2017 to 2046 (red line) under current management, an intermediate carbon emissions scenario consistent with Representative Concentration Pathway P4.5 (IPCC 2014) and no adaptive response of corals to warming.

Together with bleaching and CoTS outbreaks, the model prediction incorporates disturbance by storms and other factors and post-disturbance recovery. The shape of the trajectory oscillates in response to predicted periodic waves of CoTS outbreaks and subsequent (partial) recovery. The underlying trend is one of decline, as rates of disturbances exceed the capacity of the coral to recover.

We note that this trajectory is not the worst-case scenario. Predictions for an unmitigated carbon emissions scenario (RCP 8.5) would predict stronger net decline over time and clusters of cumulative disturbances could drive coral cover considerably lower. In this project we avoid worst-case or best-case climate scenarios to provide a risk-neutral basis for decision-making and public policy (Arrow and Lind 1970).

The blue line in **Figure 1** represents expected coral cover under the hypothetical scenario of no CoTS outbreaks. Under

**Box 1.** Current understanding and hypotheses for processes that drive or influence the five stages of the CoTS outbreak cycle Fig. 2.

### 1: Latency

- Allee effects: Low adult starfish density is a barrier to the formation of spawning aggregations (Dulvy et al. 2004, Babcock et al. 2014).
- Predation on CoTS keeps the population below threshold densities for outbreak initiation. (Ormond et al. 1990, Reichelt et al. 1990)

### 2: Initiation

- Increased CoTS larval nutrition in the initiation zone following severe weather events, upwelling and/or major flood events in northern river catchments (Fabricius et al. 2010).
- Local mass recruitment event under conditions of long water residence times (neutral ENSO conditions) and conditions favourable for larval survival Wooldridge and Brodie (2015).
- Reduced starfish predation in areas open to fishing as natural predators are reduced (Dulvy et al. 2004, Sweatman 2008).
- Life-history characteristics (e.g. high fecundity and short generation times) (Uthicke et al. 2009, Roche et al. 2015).
- High larval resilience (Wolfe et al. 2017)

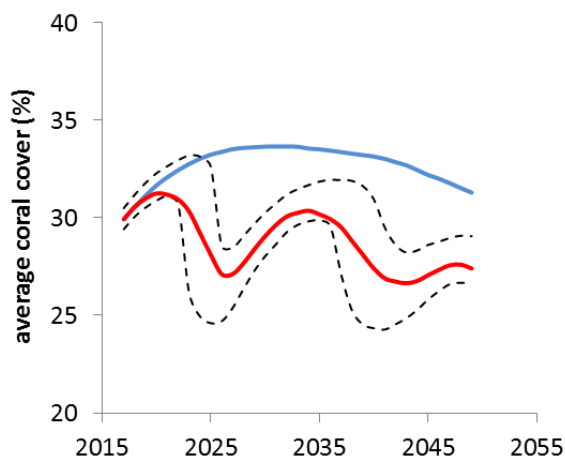
### 3&4: Primary and secondary outbreak

- The production of larvae from the exponentially growing and spreading stock of adults overwhelm recruitment limitations in secondary outbreaks (Babcock et al. 2014, Pratchett et al. 2014)
- Size and connectivity of the GBR enhances spread Hock et al. (2014)
- Increased CoTS survival in areas open to fishing (Dulvy et al. 2004, Sweatman 2008)

### 5: Collapse

- Limitation/exhaustion of coral resources Pratchett et al. (2014).
- Density-dependent onset of disease (Pratchett 1999).

currently available technology, this scenario is infeasible. Nevertheless, the gap between the blue line and the red line in the figure illustrates the upper bound on the expected benefit of active management seeking to prevent CoTS outbreaks. A more realistic view is that management interventions may act to reduce the frequency of primary outbreaks, or equivalently, extend their return time.



**Figure 1.** Expected coral cover trajectories funder status-quo management (red line) and a hypothetical scenario where CoTS outbreaks are non-existent (blue line) under RCP 4.5 and no adaptive response to an increased incidence of bleaching. Dashed lines show the 50% uncertainty interval around the status quo expectation.

### 1.3 Stages of GBR CoTS Outbreaks

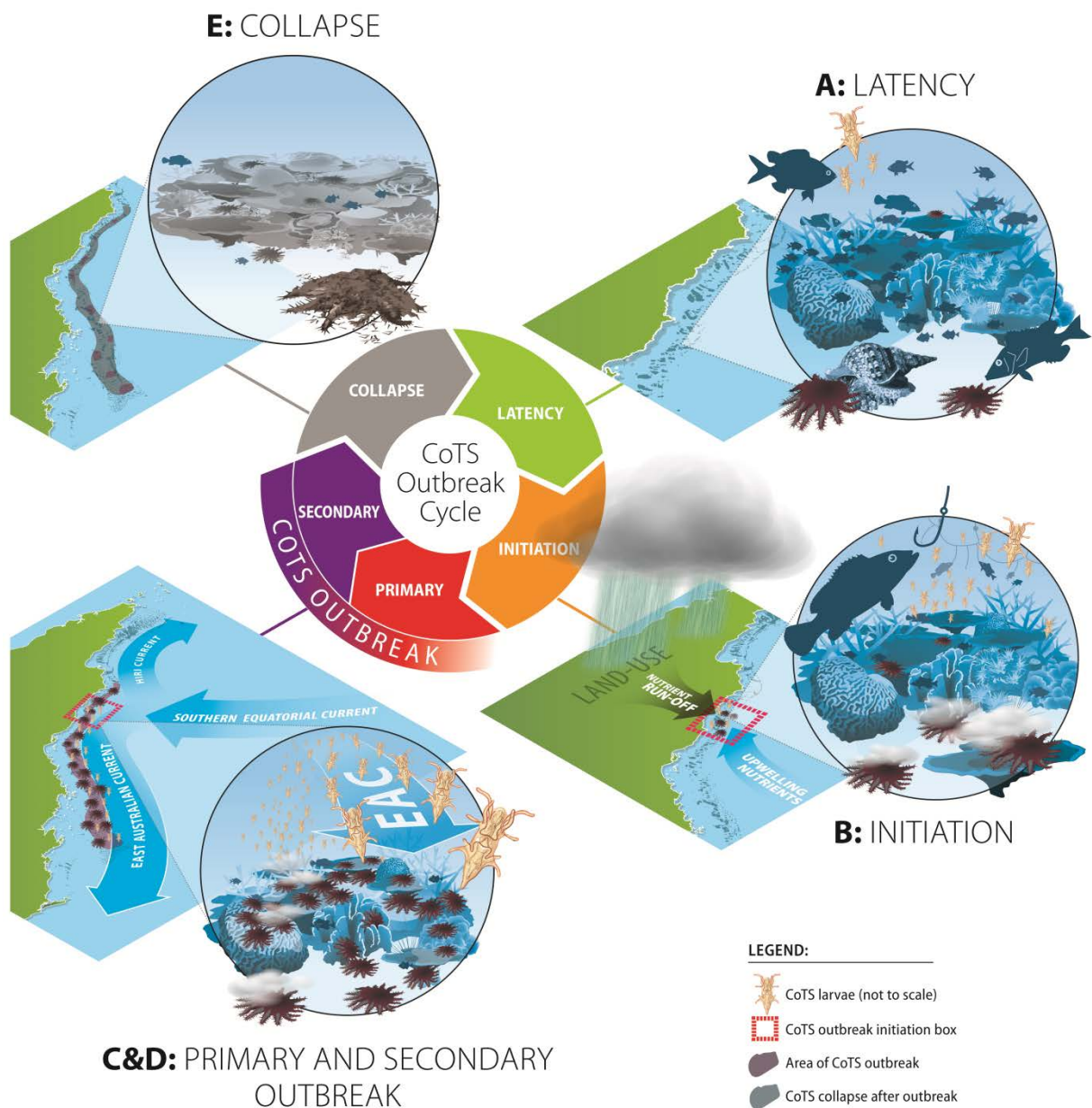
CoTS outbreaks on the GBR have been observed four times since the 1960's, with an average return time of 16 years (Pratchett et al. 2014). Outbreaks progress in waves from north to south over a 4 - 8 year period (Sweatman 2008, Vanhatalo et al. 2016) and can be divided into five stages (**Figure 2**).

In the **latency** stage, CoTS occur in background densities typically below 10 starfish  $\text{ha}^{-1}$  (Babcock et al. 2014), possibly set by a suite of mechanisms including predation on larvae, juveniles and adults (Cowan et al. 2017)(Fig. 2A). At these densities, CoTS do not represent a significant mortality factor for corals (Dulvy et al. 2004). Episodically, outbreak **initiation** takes place in an area referred to as the “initiation zone”, a cluster of reefs in the northern sector close to the Queensland coast, specifically between Cairns and Lizard Island (**Figure 2B**).

Here, a primary outbreak starts as a local build-up of starfish numbers, with major storms and consequent nutrient run-off (Fabricius et al. 2010b), upwelling and enhanced water-column mixing or reduced fish predation (reviewed by Pratchett et al. 2014) implicated as likely causal factors.

Increased densities of adult reproductive starfish on reefs in the initiation zone increase the likelihood of fertilisation during spawning, promoting larval recruitment and further population growth. Once this **primary outbreak** is underway, the population grows rapidly and the higher density of reproductively mature starfish increases fertilisation success and the export of larvae to reefs downstream.

This drives the onset of **secondary outbreaks** (**Figure 2C&D**) that form a wave driven by downstream dispersal. As CoTS consume corals and grow in size, the dense starfish population starts to face reduced prey availability. As local CoTS populations become resource-limited, the physiological condition of starfish deteriorates, aggregation behaviour intensifies, and the risk of disease increases (Pratchett 1999). In both primary and secondary outbreaks the local CoTS population eventually **collapses** (**Figure 2E**), most likely driven by starvation, disease, or both (Pratchett et al. 2014).



**Figure 2.** The five stages of CoTS outbreaks on the Great Barrier Reef. Infographics by Andreas Wagner (Cool Planet Design).

### Outbreak Initiation

Documented primary outbreaks in the CoTS initiation zone in the central-northern GBR have generally followed severe summer weather events and associated flood events and river plumes extending to mid-shelf reefs (Fabricius et al. 2010b, Brodie et al. 2017).

This provides the basis for the hypothesis that nutrient enrichment promotes primary CoTS outbreaks, and likely exacerbated by ocean warming (Uthicke et al. 2015b). A recent study suggests that outbreak initiation is driven by temporal and spatial alignment of environmental conditions that favour high phytoplankton concentrations and high larval

retention (Wooldridge and Brodie 2015). Specifically, the four major CoTS outbreaks on the GBR have occurred following a sequence of El Niño Southern Oscillation (ENSO) conditions that began with La Niña floods and followed by a neutral ENSO phase, where shelf currents are low and promote larval retention within the initiation zone.

Predator removal in areas open to fishing is a key alternative hypothesis, potentially contributing to both primary and secondary CoTS outbreaks (Dulvy et al. 2004; Sweatman 2008).

### *Outbreak Wave*

The onset of an outbreak wave from primary and secondary outbreaks can be driven by multiple factors. Firstly, the biology of *Acanthaster* sp. is characterised by fast growth (Yamaguchi 1974), early reproductive maturity (~2 years, Stump 1996), and high fecundity (Babcock and Mundy 1992). Large females may produce over 60 million eggs in one spawning (Kettle and Lucas 1987). Secondly, where conditions favour the build-up of adult starfish numbers, for example under reduced predation pressure and high food (coral) availability, CoTS spawning aggregations can produce large volumes of larvae that can drive CoTS recruitment at local and/or downstream reefs. If conditions for larval survival and spread are also favourable then there is scope for the positive feed-back mechanism that can drive a population explosion.

On the GBR, expansive reef habitats and coral resources in a highly connected seascape generates potential for population spread for broadcast spawning species, not dissimilar to the mechanism driving the spread of insect pests in agricultural landscapes (Mazzi and Dorn 2012). The GBR is comprised of over 3,000 reefs in a 2,200 km long seascape (GBRMPA 2014) and has regions of high

intra- and inter-connectivity (Bode et al. 2006). A subset of reefs within the GBR network may represent connectivity gateways between the northern initiation zone and reefs to the south, with a number of highly connected reefs (“super spreaders”) representing high-risk nodes for secondary outbreaks (Hock et al. 2014). Lastly, spread from the initiation zone to southern reefs has been proposed to be strongest under El Niño conditions, typically following neutral ENSO conditions (Wooldridge and Brodie 2015).

The progression of CoTS outbreaks bears resemblance to the initiation and spread of wildfires and the spread of invasive pests. Therefore, the Partnership team developed its thinking using lessons from fire prevention and response strategies (Thompson and Calkin 2011, Kalabokidis et al. 2012) and pest management (Pysek and Richardson 2010, Blackburn et al. 2011).

In short, our approach is centered on the CoTS outbreak cycle, the biological, ecological and environmental processes that are in play at each stage and their uncertainties. We consider each outbreak stage as a potential decision point for management and policy and with both opportunities and constraints around different management options.

The management objectives for the different outbreak stages are formalised in the CoTS Management Strategy and Contingency Plan developed by the Great Barrier Reef Marine Park Authority (GBRMPA 2016). In sequence, they are: (1) **Prevent** outbreak conditions, (2), **Contain** outbreak initiation, (3) **Suppress** outbreak intensity and spread, (4) **Protect** priority reefs, and (5) **Promote** outbreak collapse. These operational objectives are analogous to those invoked during the life of a wildfire (Thompson and Calkin 2011) and during the invasion of a pest (Blackburn et al. 2011). Importantly, the likely outcome of management actions to address one stage (e.g., success of containment) influences the



strategic priorities and tactics deployed to address the subsequent stages.

Importantly, as an outbreak progresses, the logistical difficulty, coral damage, and costs of control efforts increase. This intuitively places a high priority on prevention or early intervention as the preferred strategy. In other words, early prevention represents the decision point with the greatest potential payoff in terms of ecosystem protection, analogous to the hierarchy of priorities in fire-fighting strategies (Alonso-Betanzos et al. 2003). But for this potential to be realised, interventions need to be feasible and cost-effective.

Managers of invasive biological pests often place the bulk of their resources in prevention, most notably through very substantial investment in quarantine. Once a pest is established and actively spreading, the economic argument for vigorous intervention often becomes less convincing (Kompas et al. 2017). To date, CoTS management resources have focussed largely on the later stages of the outbreak cycle, partly because the feasibility and effectiveness of earlier intervention has not been apparent.

This project sought to establish whether an argument for earlier intervention may have merit. If the argument is found to be compelling, the policy recommendation may be to preferentially shift management resources and effort towards earlier intervention. If the argument is without merit, continuation of the current emphasis on later stages seems justified. And if the argument is sound, but not overwhelmingly so, then a mixed strategy involving resource deployment at several stages throughout the outbreak cycle may be preferred. After presenting insights and results of our analyses in Sections 4 – 6, we provide our policy recommendations in Section 7.

We focused in this Partnership on a key objective of **preventing** or **containing** outbreak initiation as a means of reducing the risk of outbreak initiation and progression beyond the initiation box. A related NESP project focuses on tactical approaches to **suppress** outbreak intensity and spread, and to **protect** key value reef sites at later outbreak stages.

Two candidate strategies for **preventing** CoTS outbreaks on the GBR are: land-use change and no-take zones. In this project we focused primarily on land-use change as a prevention option because evidence to support the efficacy of no-take zones as a preventative measure for CoTS is unclear (Sweatman 2008). Although not universally supported, land-use change is currently considered a major management lever for CoTS under the Reef 2050 plan (Commonwealth of Australia 2015). Bringing clarity around the efficacy and payoffs of land-use changes as a CoTS prevention strategy in combination with proactive measures to arrest or **contain** outbreaks through intensive surveillance and cull operations at the outset will help support GBR management and policy decision-making.

## 1.5 Report Structure

Consistent with a focus on reducing the risk of outbreak initiation under currently available technologies, the main body of this report deals with options for intervention in the latency and initiation stages of an outbreak. In the next section we describe options to address the challenge of reducing the risk of outbreak initiation under the two overarching strategies of land-use change and manual starfish control.

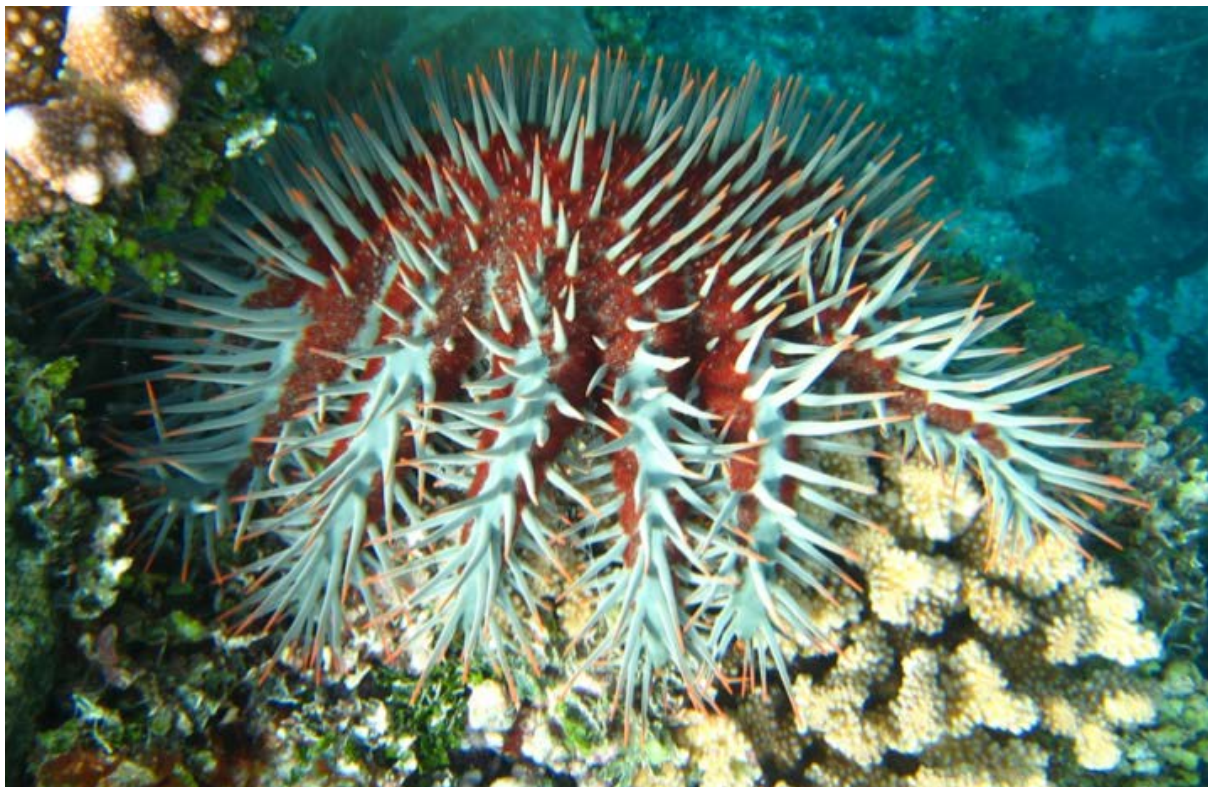
In Section 3 we outline how the suite of models and integrated analyses developed and

conducted under this Partnership are used to explore the merit of options. In Sections 4 and 5 we detail insights from analyses involving land use options and manual control options, respectively. In Section 6, we account for the costs of implementing options and translate results described in terms of outbreak frequency to the more relevant descriptor of coral cover for use in a simple probabilistic benefit-cost analysis.

After making policy recommendations in Section 7, we return to the full cycle of CoTS outbreaks in Section 8, and building on the lessons from this partnership, produce recommendations for future CoTS research and management on the Great Barrier Reef. While this Partnership has contributed new insight into the challenges and opportunities

for management and policy to prevent or arrest CoTS outbreaks at the early stage, new technologies and methods will eventually be required to control CoTS more effectively at all outbreak stages. We conclude Section 8 with the description of an integrated and adaptive strategy for CoTS science and management. Its key purpose will be to open up information bottlenecks that currently prevent early effective action, and to inform decisions around the development and deployment of next-generation interventions.

Finally, in Section 9 we describe the achievements of the Accelerate Partnership made possible by this project.



*Adult CoTS feeding on coral colony of one of its preferred species groups – Pocillopora. Photo credit: AIMS*



## 2. Management

### Actions to Prevent or Stop Outbreak Waves

*This review was conducted by AIMS*

The payoff of any candidate intervention aimed at reducing the risk of a CoTS outbreaks starting needs to be assessed against the default option of 'do nothing' beyond current management alignment of a combination of environmental and ecological processes – e.g. floods, larval connectivity, coral cover in the initiation region etc. For the purpose of reducing outbreak risk, interventions then seek to prolong the return time. We model CoTS return time using a Weibull distribution and we assume that 90% of all interarrival times under the status quo scenario fall in the interval [11, 20 yrs]. This implies a distribution with a shape parameter of 6.8 and a scale parameter of ~ 17.0, a mean expectation of 15.9 years and a mode 16.6 years. The merit of all candidate interventions can then be compared against this assumed background rate.– i.e. the status quo. Four CoTS outbreaks have been recorded on the GBR with a mean return time of 16 years.

On the basis of currently available technology and understanding, two main strategies for reducing the risk of outbreak initiation have been advocated (Fabricius et al. 2010b, Doherty et al. 2015, Wooldridge and Brodie 2015, Anthony et al. 2017):

- Land use and management practice change to limit nutrient inputs

implicated in enhanced survivorship of larval CoTS

- Surveillance and cull in the early stages of an outbreak.

Land-use practices that effectively lower nutrient run-off into the CoTS initiation zone are a proposed means of lowering the likelihood of CoTS outbreak initiation (Fabricius et al. 2010b, Wooldridge and Brodie 2015, Kroon et al. 2016). If we could halve the incidence of conditions conducive to initiation, we could in concept double the return time of outbreak waves from 16 years to 32 years.

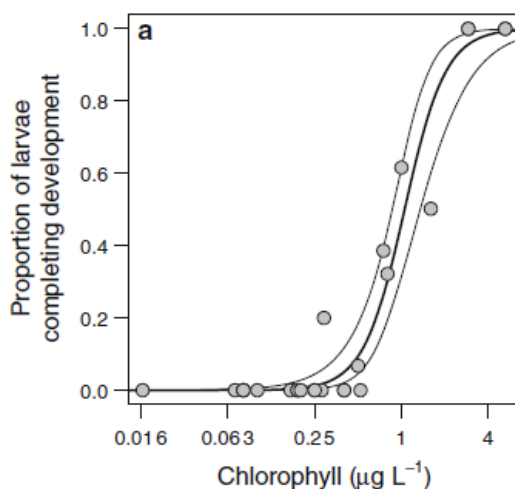
The aim of CoTS surveillance and cull operations in the latency and initiation phases of an outbreak is to provide early detection and to stop the primary outbreak becoming a secondary outbreak (**Figure 2**). Options for surveillance and cull that are successful at containing the outbreaking CoTS population within the initiation zone will extend the overall return time of CoTS outbreak waves. For example, if surveillance and cull operations succeeded in arresting one in two outbreaks at an early stage, we could likewise extend the mean return time of outbreak waves from 16 years to 32 years.

#### 2.1 How does change in land use and practice reduce the risk of outbreak initiation?

Improvements in land use and management practices seek in part to reduce nutrient inputs into the GBR lagoon (Kroon et al. 2014). One of the potential environmental benefits of reduced nutrient run-off is an extended return time of conditions conducive to enhanced larval survival and recruitment of CoTS (Fabricius et al. 2010b).

Experimental evidence suggests enhanced CoTS larval survivorship at  $\sim 0.8 \text{ ug chlorophyll-}a \text{ L}^{-1}$  (**Figure 3**). Field-based evidence provides some support for this approximation. Wooldridge and Brodie (2015) used a threshold of  $0.85 \text{ ug/L}$  in their study of conditions at Green Island. Wolfe et al. (2015) used chlorophyll-*a* from MODIS in areas where the underlying algorithm is relatively reliable (water deeper than 20m) to derive chlorophyll-*a* concentrations for their growth experiments, and used  $1 \text{ ug L}^{-1}$  as their ‘trigger’ concentration.

Based on data collated for Green Island, Wooldridge and Brodie (2015) show six exceedences of a  $0.85 \text{ ug L}^{-1}$  threshold over a 30-year time horizon, from 1969 to 1998. This implies a per annum probability of enhanced larval recruitment of  $6/30 = 0.20$ , with a 90% confidence interval  $[0.08, 0.32]$ . Even at the lower bound of 0.08 (implying a return time of primary outbreaks of 12.5 years), nutrient enrichment alone appears an insufficient explanation for CoTS outbreaks. Some other factor(s) must be involved. Here we assume that factor is sufficient time between outbreaks to allow coral prey to recover to a point where outbreak densities of the CoTS predator can be sustained.



**Figure 3.** Relationship between water-column chlorophyll *a* concentration and the proportion of CoTS larvae completing their development. Source: Fabricius et al. (2010).

We estimate that the per annum probability of satisfying the nutrient enrichment condition is constant at 0.25 (or an average of 1 in 4 years). The probability of satisfying the condition of sufficient coral prey to support an outbreak is a (logistic) function of time since the last disturbance. The parameters of the logistic function were fitted so that the product of the two conditions lead to the Weibull distribution specified above for the background status quo return time of primary outbreaks.

The aim of improved land-use is to extend the return time of nutrient enrichment conditions beyond a mean expectation of 4 years, and in so doing, extend the return time of CoTS outbreaks. We note that the relationship between nutrient enrichment and CoTS outbreaks is uncertain (**Box 1**). For now, however, we proceed on the basis that nutrient enrichment triggers CoTS outbreaks. In Section 7 we test the sensitivity of outcomes to different beliefs in the credibility of the nutrient-enrichment hypothesis.

## 2.2 Options for change in Land Use and Management Practices

*Analyses of the different options for land use and management practice was led by Natural Decisions, TropWater and Terrain NRM.*

Nutrient enrichment implicated in CoTS primary outbreaks is principally associated with dissolved inorganic nitrogen (DIN) of which the vast majority is sourced from areas under sugar cane production. Our current

hydrodynamic understanding is that riverine waters discharging from the Wet Tropics and Burdekin regions contribute to nutrient concentrations in the CoTS initiation zone. Unfortunately, we were unable to include land use change scenarios for the Burdekin River because of the unavailability of data from the Haughton Basin, which contributes most of the DIN load from the Burdekin region.

We explored a range of options for change in management practice and land use for areas currently under cane production in basins in the Wet Tropics (**Table I**). Options varied in scale, geographic focus and cost.

**Table I. Options explored for land use and management practices\*.**

<b>Label</b>	<b>Management option</b>
<b>S1</b>	No change in land use. At least all B practice (A remains as A as default).
<b>S2</b>	Retirement of all cane land to low input grazing land.
<b>S3</b>	Retirement of all cane land to biodiversity conservation.
<b>S4</b>	Retire all C and D practice cane areas to low input grazing, remainder remains in A and B.
<b>S5</b>	Retire all Johnstone and Tully-Murray to low input grazing and status quo practice everywhere else.
<b>S6</b>	Retire all Johnstone and Tully-Murray to low input grazing and at least B practice everywhere else.
<b>S7</b>	Retire Russell-Mulgrave, Johnstone, Tully-Murray and Herbert to low input grazing plus status quo practice in Barron & Daintree-Mossman
<b>S8</b>	Retire Russell-Mulgrave, Johnstone, Tully-Murray and Herbert to low input grazing and at least B practice in Barron & Daintree-Mossman.

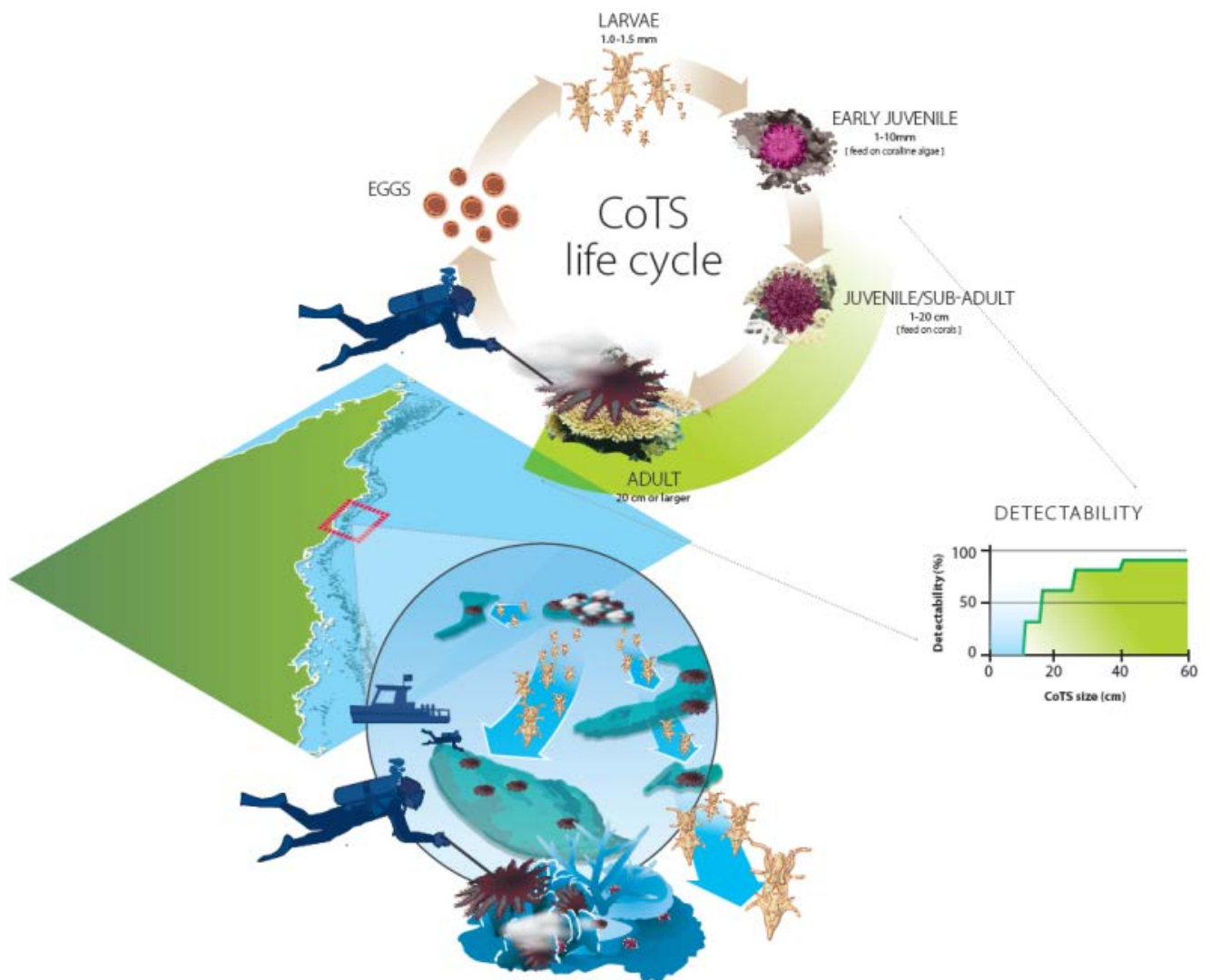
\* Management practices are graded lowest risk (level A practice - innovative practices that have the lowest water quality risk, but their commercial feasibility is unproven) to high risk (level D practice - superseded practices that have the highest water quality risk). Level B is currently proven commercially feasible best practice for soil and nutrient management.

### 2.3 How can surveillance and cull operations reduce the risk of outbreak initiation?

Similar to the case for fire and pest management, prevention is the preferred option to minimise downstream damage and cost. However, for CoTS, early warning and outbreak prevention or containment is logistically more complicated than fire prevention and containment for at least three reasons. Firstly, adult CoTS can be highly fecund, producing up to 60 million larvae (Kettle and Lucas 1987) with a relatively long range of dispersal. Secondly, CoTS are generally difficult to detect and a proportion of the population, including large adults, are hidden from view inside crevices and under corals (De'ath and Moran 1998). Thirdly, small juveniles that feed on coralline algae can live for extended periods of time on coral-denuded reefs (M. Hall pers com). In short, indicators and operational elements that support effective fire-fighting, such as locating and extinguishing smoking embers, generating fire breaks and reducing fuel loads strategically and tactically (Thompson and Calkin 2011, Kalabokidis et al. 2012) are difficult to deploy for CoTS control teams. Nevertheless, we might reasonably speculate that the challenges of imperfect detection and a difficult working environment may be overcome with greater resources dedicated to surveillance and cull in the latency and initiation phases of an outbreak.



**Figure 4.** The CoTS initiation zone in the northern Great Barrier Reef. A key challenge of the Accelerate Partnership modelling team was to assess the likelihood that primary outbreaks can be contained within the initiation zone by understanding connectivity and using different surveillance and cull strategies. Colors of reefs in the northern half of the area indicate relative CoTS abundance from surveys by QPWS and AMPTO in 2012-14 (Doherty et al. 2015).



**Figure 5.** Proactive CoTS surveillance and cull as a second line of defence for outbreak prevention and containment. One of the greatest challenges for CoTS control divers operating from vessels is that starfish, and especially juveniles, are hard to detect (graph inset). If teams leave enough adult starfish behind that they can form a spawning aggregation, then the primary outbreak is not contained. Infographics by Andreas Wagner (Cool Planet Design).

Surveillance and control teams targetting CoTS at the initiation stage are faced with three challenges. First is the challenge of finding outbreking sites or reefs in a large seascape of hundreds of reefs. Second is the problem of detecting CoTS including cryptic juveniles. Third is the challenge of culling enough starfish that densities are suppressed below the threshold that prevents further spread beyond the initiation zone (**Figure 4**).

The probability of outbreak leakage outside of the initiation zone is high unless key source and gateway reefs are identified early and the CoTS population is suppressed below threshold before spawning aggregations can form. The challenge of the task and the processed involved are captured in **Figure 5**.

## 2.4 Options for CoTS Surveillance and Cull

*This study was led by AIMS in consultation with GBRMPA, the Queensland Parks and Wildlife Service, RRRC and AMPTO.*

Development of options for surveillance and cull included three complimentary operations:

- Continuous pre-outbreak detection surveillance in the initiation zone
- Post-outbreak detection cull operations in the initiation zone
- Post-outbreak cull operations in the Innisfail sector, immediately south of the initiation zone designed to prevent further spread south ('mop-up' operations).

Within any resource constraint, the optimal configuration of effort across operations is not apparent. There are 284 reefs in the initiation zone and 122 reefs in the Innisfail sector. The capacity to visit, search and treat reefs clearly varies with the level of effort dedicated to each operation. In this context, effort is largely about the number of boats deployed for each operation. (In section 5 we demonstrate the role of the configuration of boats and impacts of structured versus unstructured search and cull patterns.) We here included 15 combinations for the number of boats deployed in pre-outbreak surveillance and post-outbreak cull operations (**Table 2**). For each of these 15 combinations, we included four levels of resourcing for mop-up culling in the Innisfail sector.

For surveillance, effort also involves the intensity with which a reef is covered. We assumed all surveillance operations would involve fixed area surveys (Reef Health and

Impact Surveys, RHIS plots)<sup>1</sup>. Greater sampling intensity implies a higher likelihood of outbreak detection if in fact CoTS is present at outbreak densities. But it also implies that a fewer number of reefs can be visited. Again, the optimum sampling intensity within this trade-off is not apparent. For each of the options shown in **Table 2**, we explored three levels of sampling effort per reef:

- 10 ha per reef
- 20 ha per reef
- 50 ha per reef.

**Table 2. Options explored for surveillance and cull. The numbers in each cell refer to the number of boats deployed.**

Pre-outbreak surveillance	Post-outbreak culls	'Mop-up' culls
1	1	1, 5, 10, 25
	2	1, 5, 10, 25
	4	1, 5, 10, 25
	5	1, 5, 10, 25
	6	1, 5, 10, 25
4	1	1, 5, 10, 25
	2	1, 5, 10, 25
	4	1, 5, 10, 25
	6	1, 5, 10, 25
5	5	1, 5, 10, 25
	10	1, 5, 10, 25
10	10	1, 5, 10, 25
	20	1, 5, 10, 25
20	20	1, 5, 10, 25
	40	1, 5, 10, 25

<sup>1</sup> We assume fixed area surveys because we have information on detectability of starfish using this method. The detectability of timed swims or Manta tows is unknown.



In addition, for surveillance operations within any one year we explored three alternative search patterns among the 284 reefs in the initiation zone:

- A naïve random search pattern.
- Preferential allocation of effort towards the north of the initiation box, where early detection may limit downstream dispersal.
- ENZI, or prioritisation according to on expected non-zero impact, based on our current understanding of the hydrodynamic connectivity between reefs and CoTS dispersal dynamics (see section 5.1).

In total we explored 15 initiation box resource levels × 5 Innisfail sector mop-up resource levels × 3 surveillance sampling intensities × 3 surveillance search patterns = 675 options for surveillance and cull.

### 3. Assessing options

*The design of the methodological approach was developed by AIMS, RRRC and the ARC Centre of Excellence for Coral Reef Studies.*

The analytical focus of this project was to estimate the extent to which various management options can extend outbreak return times relative to the status quo ('do nothing') scenario. Extending return times is the equivalent of reducing the risk of outbreak initiation and/or increasing the chance of early outbreak containment. Return times are a distal descriptor of the pay-off of management intervention. Of more immediate interest to reef managers is the implications for coral cover. For those concerned with the high-level allocation of public resources, coral cover is a difficult metric. Benefit-cost ratios

**Box 2:** Contributions by individual Accelerate Partners and collaborators to the bio-economic analysis of CoTS management options.

See also diagram in Figure 6.

#### Benefit-cost analysis

- Natural Decisions

#### Model predictions of coral cover

- AIMS

#### Costs of implementation

- Land use and practice - Terrain NRM
- Surveillance and cull – RRRC, GBRMPA, AMPTO, Queensland Parks & Wildlife Service

#### Land use and practice

- End of basin DIN loads - TropWater, Natural Decisions, Terrain NRM
- Transport of DIN to the initiation zone - TropWater
- Return time of nutrient enrichment - TropWater, AIMS

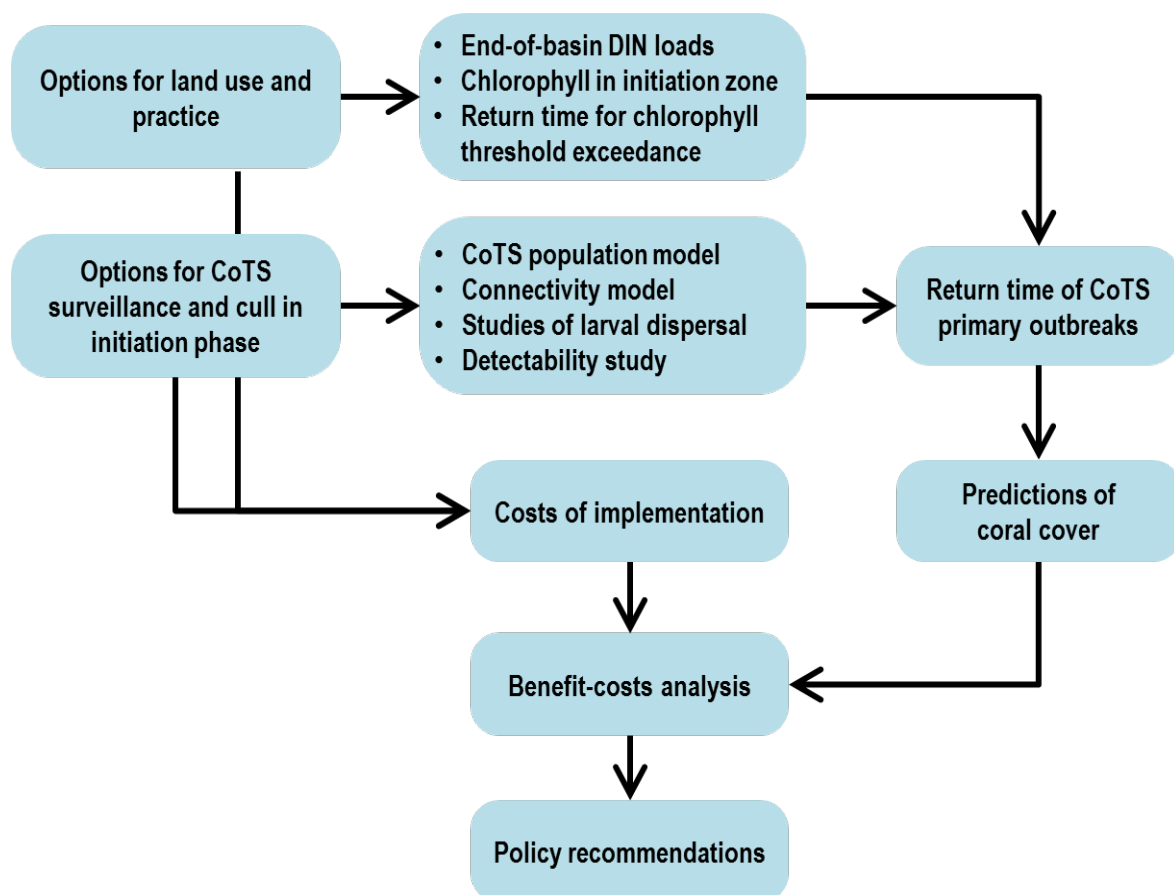
#### Surveillance and cull

- CoTS population model - AIMS
- Connectivity model – University of Queensland
- Field studies – ARC Centre of Excellence for Coral Reef Studies
- Surveillance and cull data – QPWS, AMPTO, GBRMPA

are the preferred metric for considering the merit of proposed investments among economists concerned with the public good. The Partnership designed a methodological approach that accommodated and coherently linked the three descriptors of benefit: (1) extensions in return times of outbreaks, (2) implications for coral cover, and (3) benefit-cost ratios (**Figure 6**).

Characterising the benefits and costs of management options is a complex puzzle with many pieces. Informative outcomes relied on the contributions of a range of organisations and individuals involved in the Accelerate Partnership.

We used a multidisciplinary bio-economic analysis which assessed the merits of options over a 30 year time horizon (2017 to 2046). The analysis was led by Dr Anna Roberts (*Natural Decisions*) contracted under this partnership. To generate and compile data and integrate information streams for the bio-economic analysis, the entire Accelerate Partnership contributed with the elements outlined in **Box 2**.



**Figure 6.** Summary of project approach to characterise benefits and costs of two key management options.

## 4. Return Time of CoTS Outbreaks Under Land-Use Options

*These analyses were conducted by TropWater in collaboration with Natural Decisions and AIMS*

Nutrient enrichment of GBR lagoon waters and its implications for phytoplankton blooms and the potential enhancement of CoTS larval survivorship is described in terms of the water-column concentration of chlorophyll *a* (**Figure 3**). When we began this research, the link between end-of-basin nutrient loads and the offshore concentration of chlorophyll-*a* in the initiation zone had not been made operational in eReefs. However, while the modelling to underpin estimates of end-of-basin DIN loads under various land-use and practice scenarios was available, the task remained to translate these loads into annual variability of chlorophyll-*a* concentrations in the waters within the initiation zone. To do so we developed a set of methods comprising three key elements:

- Estimation of end-of-basin DIN loads under each land use and management practice option,
- Transport of DIN into the initiation zone,
- Relating the index of DIN concentrations to chlorophyll concentrations in the

initiation zone, and estimating the return times of chlorophyll *a* levels that exceeding the critical threshold under each candidate option.

### 4.1 End-of-Basin DIN Loads

*End of basin loads under each land use and practice option was developed by TropWater, Natural Decisions and Terrain NRM.*

The most current estimates of area under cane production and current practice was collated for all Wet Tropics basins using Paddock to Reef practice adoption estimates (**Table 3**), along with end-of-basin loads (**Table 4**). Then for each of the eight candidate options, the basin-specific proportional reductions were calculated using the Queensland government's Source Catchment model (Carroll et al. 2012). Results are shown in **Table 5**.

Note that only northern rivers catchments are included in the analyses, thought to directly influence the CoTS initiation box. Recent analyses indicate that Burdekin waters may reach Lizard Island (Brinkman et al. 2014), but we here make the conservative assumption that DIN from the Burdekin is processed and converted in the water column before reaching the initiation box.

**Table 3.** Current area (ha) of cane production and practice for each of the Wet Tropics basins.

<b>Basin</b>	<b>A</b>	<b>B</b>	<b>C</b>	<b>D</b>	<b>Total</b>
Daintree	130	423	2,637	65	3,255
Mossman	148	481	2,996	74	3,699
Barron	-	551	6,292	1,022	7,865
Russell Mulgrave	657	1,533	19,053	657	21,900
Johnstone	116	463	10,881	116	11,576
Tully-Murray	1,231	2,154	24,920	2,461	30,765
Herbert	1,888	1,888	54,762	4,406	62,945
<b>Wet Tropics total</b>	<b>4,170 (3%)</b>	<b>7,493 (5%)</b>	<b>121,541 (86%)</b>	<b>8,802 (6%)</b>	<b>142,005</b>

**Table 4.** Average annual end-of-basin DIN loads (tonnes) under current land use and practice.

<b>Basin</b>	<b>Sugarcane</b>	<b>Total</b>
Daintree	94	478
Mossman	98	160
Barron	49	152
Russell-Mulgrave	393	934
Johnstone	415	1059
Tully-Murray	558	1191
Herbert	966	1522
<b>Wet Tropics total</b>	<b>2572</b>	<b>5496</b>

**Table 5.** Reductions under each land use and practice option as a proportion of total DIN

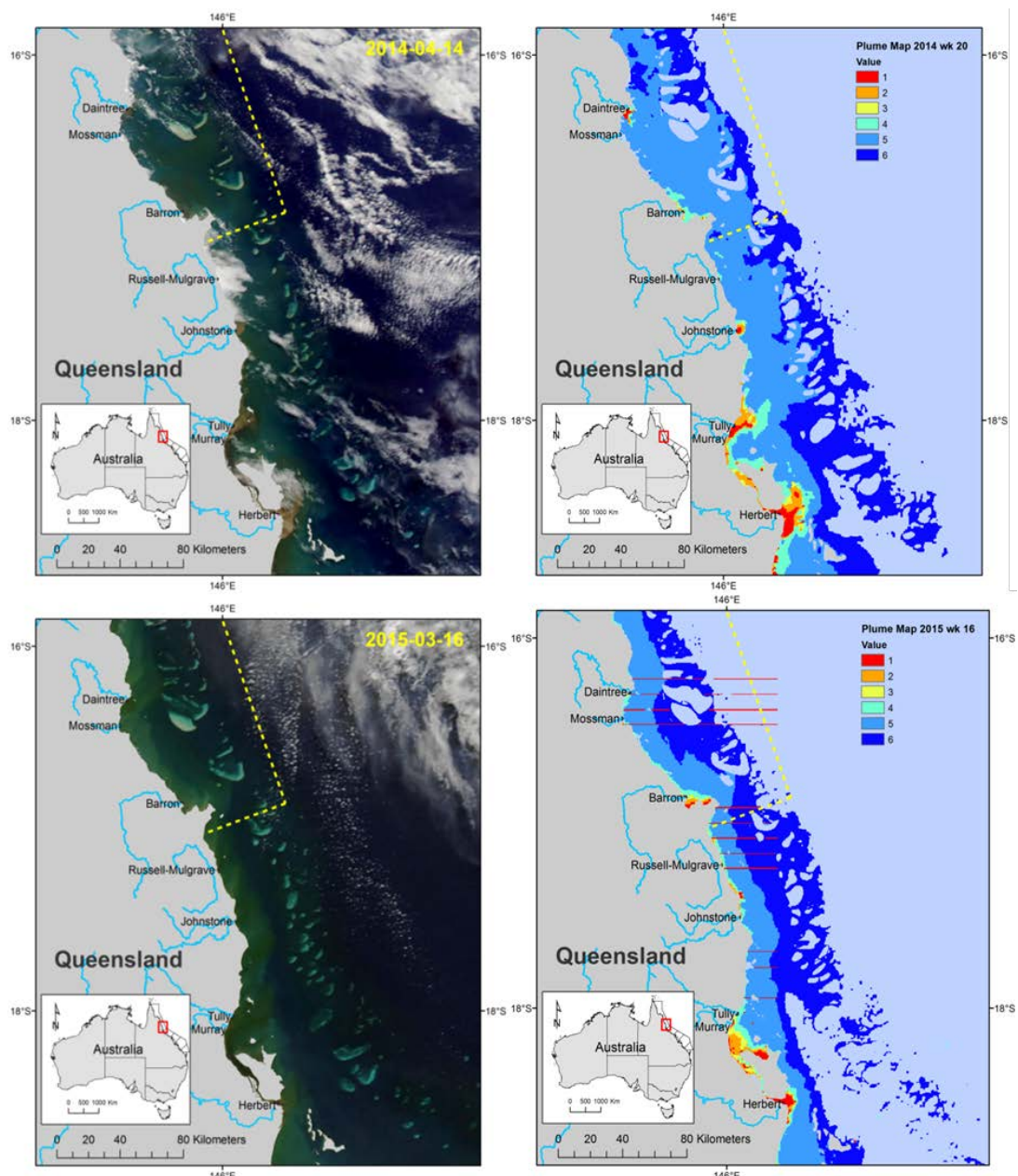
<b>Basin</b>	<b>S1</b>	<b>S2</b>	<b>S3</b>	<b>S4</b>	<b>S5</b>	<b>S6</b>	<b>S7</b>	<b>S8</b>
Daintree	0.08	0.20	0.20	0.16	0.00	0.08	0.00	0.08
Mossman	0.28	0.62	0.62	0.51	0.00	0.28	0.00	0.28
Barron	0.16	0.32	0.32	0.30	0.00	0.16	0.00	0.16
Russell-Mulgrave	0.14	0.42	0.42	0.38	0.00	0.14	0.42	0.42
Johnstone	0.12	0.39	0.39	0.37	0.39	0.39	0.39	0.39
Tully-Murray	0.19	0.47	0.47	0.42	0.47	0.47	0.47	0.47
Herbert	0.31	0.63	0.63	0.60	0.00	0.31	0.63	0.63

## 4.2 Transport of DIN in the Initiation Zone

The transport model used to predict the concentration of DIN in the Initiation zone was developed by TropWater.

An ocean colour based model was developed to estimate the dispersion of dissolved inorganic nitrogen delivered by river plumes to the Great Barrier Reef waters. This model, built on Alvarez-Romero et al. (2013), uses

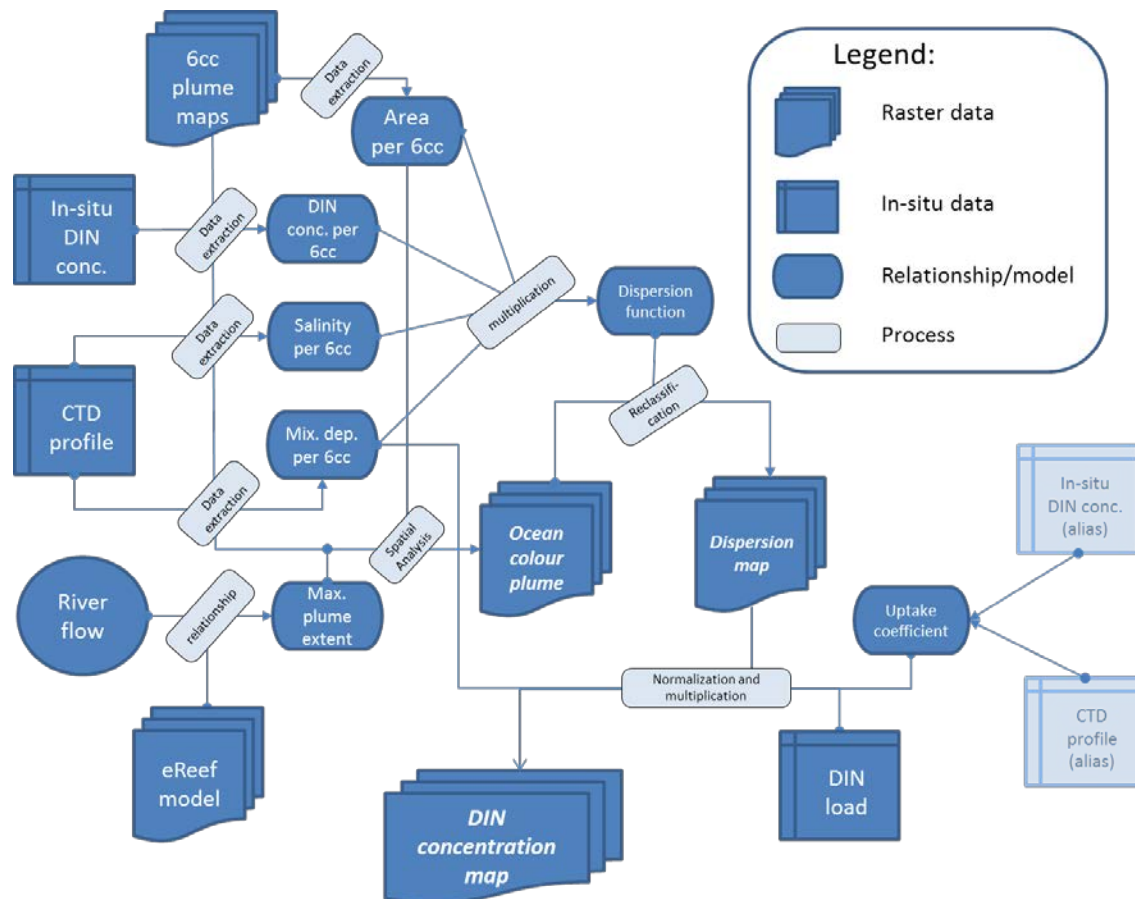
end of basin loads as inputs and satellite imagery to characterise the direction and intensity of DIN mass dispersed over the GBR lagoon. The model produces maps of average DIN concentration in the GBR waters (**Figure 7**). The main modifications applied to the method presented in Álvarez-Romero et al. (2013) are that the qualitative assessment of pollutant dispersion in river plumes is replaced by a relationship between in-situ DIN mass and the six colour classes in the river plume maps; the cost-distance function used in Álvarez-Romero et al. (2013)



**Figure 7:** Selected satellite images showing plume water intrusion into the COTS initiation zone over two years (left column) and the corresponding weekly plume colour class map. Plume colour classes vary from 1 to 6, being 1 for more riverine water and 6 for marine water with some riverine influence.

to reproduce the shape of each individual river plume is replaced by the path-distance function, which is also available in ArcMap Spatial Analyst (ESRI 2010); and a DIN decay function is applied to DIN mass exported from the rivers to account for potential biological uptake.

relationship based on the mass proportion of DIN in each plume colour class determined at GBR scale is used. To account for potential DIN uptake, the ratio between an in-situ DIN x salinity relationship and the theoretical DIN decay due to dilution (i.e., freshwater – marine water mixing) is used. This ratio

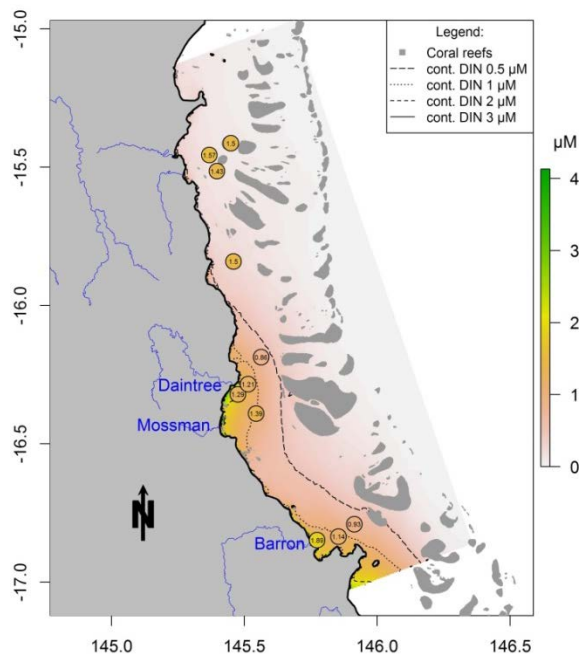


**Figure 8.** Schema of methods used to derive offshore DIN concentrations from end of catchment loads.

Our model has four main components: (a) modelling of individual river plumes; (b) DIN dispersion function; (c) DIN decay function; and (d) mapping of DIN concentration over the GBR lagoon. The schematic diagram in **Figure 8** shows how each model component is set up and how they are combined to produce the DIN dispersion maps. The basic idea of the DIN dispersion maps is to produce river plume maps, for each individual river in the model. To control this dispersion a

defines a DIN decay coefficient, which is multiplied by the dispersed DIN load. After the load has been dispersed over each individual river plume, and corrected for DIN uptake, the resultant dispersed DIN from each river is summed together to represent the total annual DIN dispersion over the GBR lagoon discharged by the rivers for a specified time period (**Figure 9**).





**Figure 9.** Model-predicted dissolved inorganic nitrogen (DIN,  $\mu\text{M}$ ) over the COTS initiation zone lagoon for 2012-2013 water year (c.a., October 1st to September 30th).

### 4.3 Return Time of Nutrient Enrichment Conducive to Triggering CoTS Outbreaks

The method used to estimate the return time of nutrient enrichment from estimates of DIN was developed by TropWater and AIMS.

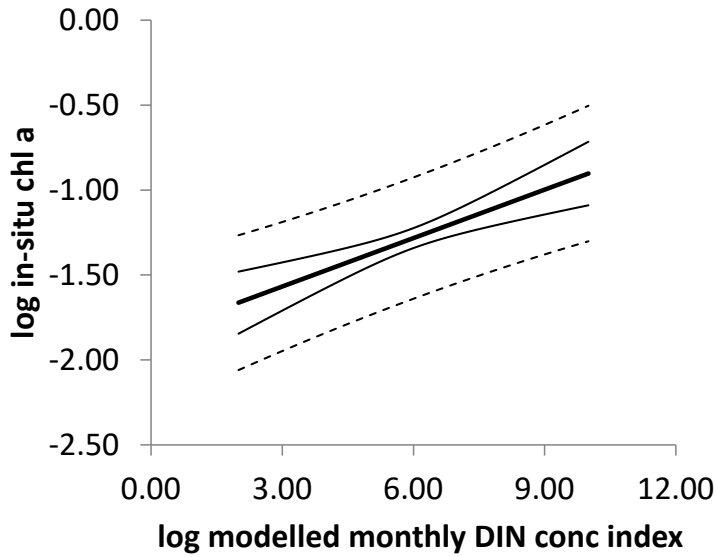
We now needed to characterise the statistical relationship between model predicted DIN concentrations and chlorophyll-*a* concentrations within the initiation zone. The underlying biochemical relationship is that as DIN is dispersed in the plume from a river it is eventually taken up by phytoplankton when light conditions are suitable (low turbidity) (Devlin and Brodie 2005). At this stage phytoplankton biomass increases, observed as an increasing chlorophyll concentration. The nitrogen, originally present as DIN, is now present as particulate nitrogen (organic

matter). The time lag between conversion of DIN to particulate nitrogen under suitable conditions is a matter of only days as DIN in the river plume is transported off-shore to the mid-shelf.

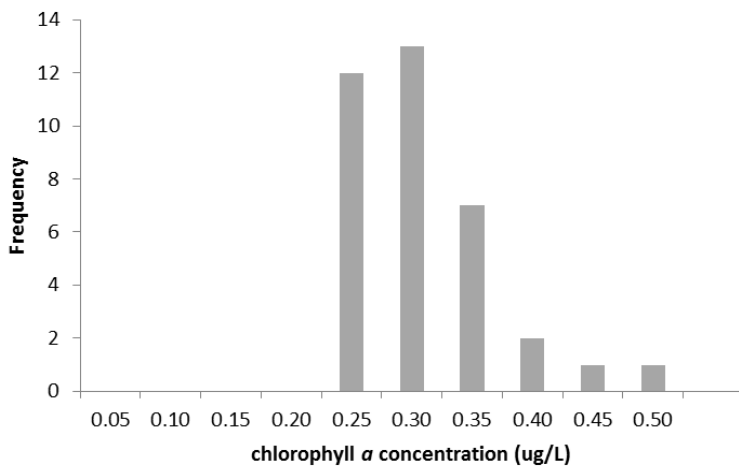
Monthly DIN maps for wet season months December, January and February were produced using plumes and loads stemming from river basins of the Wet Tropics region for the period 2003 – 2015. Note that plume maps are unavailable for the wet season month of November, but we assume the relationship developed here is valid for the month of November too.

Corresponding data were collated for monthly chlorophyll-*a* concentrations. Measurement and prediction of the in-situ concentration of chlorophyll-*a* is notoriously difficult. Grab samples are relatively accurate at the time and location of collection, but offer poor coverage over extensive spatial and temporal scales. The opposite is true for satellite data. Here we use monthly concentration of chlorophyll-*a* as reported by satellite data (available from the Bureau of Meteorology at <http://ereef.tds.bom.gov.au/ereefs/tds/catalog/reef/mwq/P6M/catalog.html>) over the entire initiation box.

Monthly DIN maps were converted to the same spatial resolution as raster-based chlorophyll maps. Means of both variables were calculated after masking out areas where depth was less than 25m and areas where modelled DIN concentrations were zero. These constraints were used to reduce noise in the chlorophyll data set, and to exclude areas not affected by DIN discharged by rivers in the Wet Tropics, respectively. An index of mean monthly DIN concentration over the initiation zone was used as the predictor variable and mean chlorophyll-*a* as the response variable in a simple linear regression analysis (**Figure 10**). The regression analysis



**Figure 10.** The statistical relationship between modelled monthly DIN concentration and satellite-derived chlorophyll-*a* concentration in the CoTS initiation zone, for December, January and February, 2003 – 2015. The line of best fit (bold) is bounded by the 95% confidence interval (continuous lines) for the regression estimate and 95% prediction interval (dashed lines). In natural units, the response variable, mean satellite-derived chlorophyll-*a* concentration, ranges from approximately 0.14 ( $e^{-2.0}$ ) to 0.61 ( $e^{-0.5}$ )  $\mu\text{g/L}$ .



**Figure 11.** Histogram of mean (wet season) monthly chlorophyll-*a* concentrations over the whole initiation box derived from satellite data, 2003 – 2015 (December, January, February). These data were used as the response variable in the regression analysis shown in Figure 10. The critical threshold for sufficient nutrient enrichment to trigger a primary outbreak of COTS is estimated to be 0.40  $\mu\text{g L}^{-1}$ .

exceeding any specified threshold for mean monthly chlorophyll-*a* concentration via the estimate of the standard error for the prediction interval (Sokal and Rohlf 1995). But the question is: what threshold for satellite-derived mean monthly chlorophyll-*a* is sufficient to promote enhanced larval survivorship of CoTS? Recall that the critical threshold associated with nutrient enrichment *at a point* in time and space is  $\sim 0.8 \mu\text{g/L}$  chlorophyll *a* (**Figure 3**) and that we estimate that the rate at which these conditions occur is about once every four years (Section 3). We need to find the percentile of the average (wet season) monthly concentration of chlorophyll *a* reported by satellite data that has an equivalent expected return time. The satellite data collated for our regression analysis is summarised in **Figure 11**. Note that *mean* concentrations never exceeded the lab-based critical threshold of  $0.80 \mu\text{g L}^{-1}$  over the observation period.

A return time of 4 years equates to a per annum probability of exceedance of 0.25. If we denote  $p$  as the per (wet season) month probability of exceedance, the probability of insufficient nutrient enrichment in any one month is  $(1 - p)$ , and the probability of insufficient enrichment over  $m$  months is  $(1 - p)^m$ . Now for time  $m = 4$  wet season months per year, solving for  $p$  in the equation

$$1 - (1 - p)^m = 1/4 = 0.25,$$

can be used to estimate the probability of

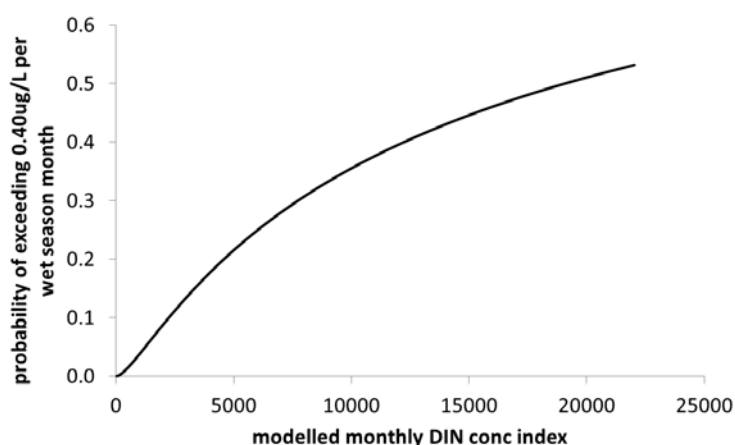
we obtain  $p = 0.07$ . So the critical percentile for our monthly wet season satellite data is the 93<sup>rd</sup> percentile, or approximately  $0.40 \text{ ug L}^{-1}$ . That is, the threshold for mean monthly wet season chlorophyll a, as measured by satellite, where we estimate nutrient enrichment conditions are sufficient to trigger a primary outbreak is  $0.40 \text{ ug L}^{-1}$ .

We can now use our regression analysis to plot the probability of exceeding this threshold for any value for our modelled monthly DIN concentration index (**Figure 12**).

Using historical flow data from Wet Tropics

ivers and the changes in DIN loads reported in **Table 5**, we used a Monte-Carlo simulation to model monthly variability in DIN concentrations in the initiation zone for each of our eight candidate options, and then derived monthly probabilities of exceeding our critical threshold of  $0.40 \text{ ug L}^{-1}$  for mean satellite chlorophyll a concentration. Monte Carlo simulations comprised 10,000 runs for each option and included an account of inter-basin correlations in flows (**Table 6**).

Treating these probabilities as binomial trials, we were able to estimate the return time of nutrient enrichment conditions under each of our candidate options for land use and



**Figure 12.** Probability of exceeding the critical threshold for sufficient nutrient enrichment of  $0.40 \text{ ug/L}$  chlorophyll a as a function of modelled monthly DIN concentration.

**Table 6.** Correlations in flows between Wet Tropics river basins, derived from wet season monthly flow data, 2003 – 2015.

	Barron	Daintree	Herbert	Johnstone	Mossman	Russell-Mulgrave	Tully-Murray
Barron	1.00						
Daintree	0.73	1.00					
Herbert	0.72	0.72	1.00				
Johnstone	0.80	0.82	0.69	1.00			
Mossman	0.59	0.77	0.73	0.71	1.00		
Russell-Mulgrave	0.79	0.76	0.78	0.89	0.64	1.00	
Tully-Murray	0.45	0.65	0.90	0.65	0.62	0.73	1.00

management practice, and by extension, the return times of CoTS outbreaks. Results are shown in **Table 7**.

Unsurprisingly, the best performing options were S2 and S3, where the return time of outbreaks would be extended from about 16 years to 20 years. Both these options involved retirement of all cane production in the Wet Tropics. The feasibility and costs of these options, alongside that of the full set of options is explored below, as part of a more

complete analysis of the merit of candidate interventions. We emphasise that results reported in **Table 7** rest on an absolute conviction that nutrient enrichment is a necessary condition for triggering CoTS outbreaks. Some scientists are distinctly less convinced of the credibility of the nutrient enrichment hypothesis. In Section 7 we relax the assumption of absolute belief to explore the sensitivity of outcomes to uncertainty in the effectiveness of management options.

**Table 7.** *The performance of candidate options for land use and practice change described as reductions in the probability of nutrient enrichment and extensions of the return time of CoTS outbreaks relative to the status quo (do nothing) base case.*

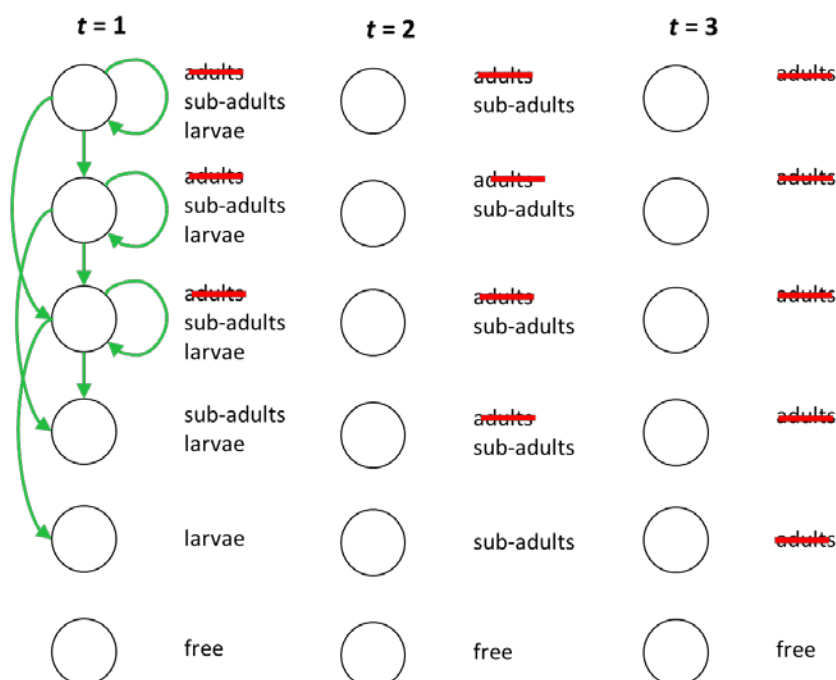
Option	Per wet season month probability of exceeding 0.40 ug L <sup>-1</sup>	Annual probability of exceeding 0.40 ug L <sup>-1</sup> in one or more wet season months	Mean return time of CoTS primary outbreaks (years)
Status quo	0.069	0.25	15.9
S1	0.058	0.21	17.3
S2	0.039	0.15	19.7
S3	0.038	0.15	19.7
S4	0.042	0.16	19.3
S5	0.060	0.22	17.1
S6	0.051	0.19	17.8
S7	0.042	0.16	19.1
S8	0.040	0.15	19.3

## 5. Return Time of CoTS Outbreaks Under Surveillance and Cull Options

Complete arrest of an incipient outbreak of CoTS through a surveillance and culling operation is difficult, but not impossible. Conventional surveillance methods rely on the detection of adult starfish, occasionally indicated by feeding scars on corals (GBRMPA 2016). Technology to detect larval signatures of outbreaks is under development (Uthicke et al. 2015a).

Culling technology is also for now restricted to the adult life stage. Culling operations that first detect and cull adult starfish in a cluster

at outbreak densities subsequently needs to 'catch-up' with offspring that have already dispersed downstream or that live as cryptic juveniles inside the reef. **Figure 13** shows a model of what is required for a successful cull operation of a pest capable of directional dispersal and self-recruitment of discrete habitat patches. If there are two life-stage transitions to adulthood (larvae to sub-adults, sub-adults to adults) each of which takes one time step, it will take three consecutive successful culling operations (at time  $t = 1$ ,  $t = 2$ , and  $t = 3$ ) to stop the outbreak wave progressing. Importantly, the model assumes perfect detection and cull of adults, which is optimistic. Under that assumption, the requirement for three consecutive successes need not occur in the first three years of an outbreak wave, but the resources required and the scale of the undertaking can become very large very quickly if success is not secured early on.



**Figure 13.** Conceptually, a culling operation can stop an incipient outbreak wave, but adult densities need to be reduced to below outbreak densities wherever they occur over multiple consecutive time steps.

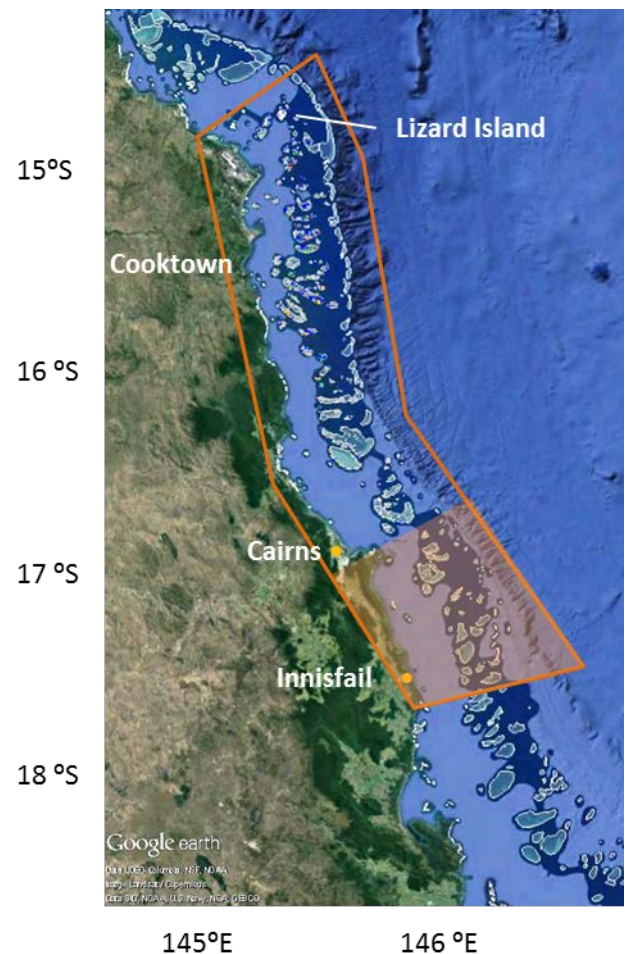
In section 5 above we outlined 675 options for surveillance and cull operations. Those options entail different levels of resourcing for three key elements:

- Continuous pre-outbreak detection surveillance in the initiation zone
- Post-outbreak surveillance and culling operations in the initiation zone
- Post-outbreak surveillance and cull operations in the Innisfail sector, immediately south of the initiation zone to clean up starfish spreading beyond the initiation zone ('mop up' operations).

In our analyses, we asked, what is the probability of successfully arresting a COTS outbreak wave before it progresses beyond the Innisfail sector within a 30-year time horizon? To succeed, an operation needs to:

- Reduce the density of adults at all reefs in the initiation zone to densities below outbreak levels (i.e.  $< 7.1 \text{ CoTS ha}^{-1}$ ; Babcock et al. 2014), **and**
- Successfully reduce the density of adults at all reefs in the Innisfail sector to densities below outbreak levels.

To estimate the probability of successful arrest of incipient outbreak wave we developed a synthesis model that captured hydrodynamic connectivity and biological understanding. We outline these elements below, along with a field work component that helps inform the feasibility of arresting an incipient outbreak wave.



**Figure 14.** The Innisfail sector (red shaded polygon), immediately south of the Initiation zone (blue polygon).



## 5.1 CoTS Connectivity

*Analyses were conducted by Dr Karlo Hock and Prof Pete Mumby. University of Queensland*

### *Model and simulations of larval connectivity patterns*

Connectivity patterns of COTS larvae for the reefs in initiation box spanning the Cairns-Cooktown region of the GBR were obtained by running Lagrangian dispersal simulations with the help of Connie2 particle dispersal modelling tool (Condie et al. 2012, Hock et al. 2014). Reefs in the region were represented as polygons using the coordinates provided by the GBRMA zoning plan (GBRMPA 2004). These dispersal simulations were used to establish the potential for larval exchange among the reefs due to prevailing hydrodynamic forces, which can then be combined with the population models that determine the larval output. Dispersal of COTS larvae was simulated by releasing the larval particles in the late afternoon and evening hours at 1<sup>st</sup> and 15<sup>th</sup> of December and January over the course of four GBR summers (2008-09, 2010-11, 2011-12, and 2012-13), for a total of 16 spawnings per reef and with 10,000 particles released per spawning from each reef. While the exact timing and regime of spawning for COTS is uncertain, it is generally assumed that most of the spawning occurs at this time of the season (Pratchett et al. 2014).

Repeating the process for all the reefs in the initiation zone and then combining the results for each source-sink pair resulted in a regional spawning-specific connectivity network. By retaining the asymmetry of source-sink relationships, i.e. a defined direction of the connectivity link, the resulting networks took the form of weighted digraphs, with weights of the links corresponding to the relative

amount of larvae exchanged between respective source-sink pairs. All links between the reefs in the initiation zone were represented in the networks, while only links leading from the initiation zone reefs to the Innisfail region were included. This ensured that all colonisation/outbreak events in the Innisfail region were the result of larvae coming from the initiation zone, making any such colonisation/outbreak an indication that the CoTS has spread beyond the bounds of the initiation zone. Before further analysis, the networks for the individual spawning dates were combined to provide an average connectivity pattern for that spawning. The strengths of individual links between specific source-sink pairs from these 16 networks were then fitted using beta distribution. The parameters of the beta distribution characteristic of a specific link in the connectivity network could then be used to draw random samples of link strength for use in population simulations or to obtain the average strength of connectivity for a particular link.

### *Estimating the spread of COTS with the help of the connectivity network analysis*

The connectivity networks produced were used to identify reefs which, if they were to host a breeding stock of adult COTS, could be responsible for spreading COTS larvae beyond the initiation zone. While the actual output of from the COTS population breeding on a reef will depend on many factors such as the population size etc., it is nevertheless possible to determine where connectivity could play a major role in future dispersal. As an example, a reef with strong connections to many other reefs will be more likely to spread COTS larvae for a particular size of the source population, as its larvae are predicted to reach its sink reefs at favourable times when many of the released larvae are both

surviving and competent to settle. This makes it possible to compare the relative importance of all reefs in the region according to their putative role as potential sources of COTS larvae that can seed other reefs directly (Hock et al. 2014, Hock et al. 2016).

The strength of the connectivity link will also be indicative of the number of competent and surviving larvae that will reach the sink reef. Therefore, assuming local environmental conditions at the sink reef are favourable, stronger incoming links from sources of COTS larvae can be assumed to be more likely to lead to colonisation of the sink reef. The mechanism how this can be determined has been described in detail in Hock et al. (2016). To summarize, the belief that a *de novo* colonisation will take place at the sink reef can therefore be expressed as a combination of a) the belief, or prior probability, that the source reef has a population of COTS capable of colonising the sink, b) the belief, or prior probability, that the sink reef has already been colonised, and c) the belief that the larvae will be transported from the source reef to the sink reef. This relationship can be determined for any link that connects a source-sink pair in the connectivity network. Furthermore, this calculation can be performed for all outgoing and incoming connections to a reef to determine the cumulative probability that a reef will colonise its potential sinks, or become colonised from its potential sources (Hock et al. 2016).

By performing this calculation iteratively to simulate successive colonisation events, it is also possible to estimate the probability that the presence of a COTS population on a specific source reef will eventually lead to a colonisation of reefs to which it cannot directly provide larvae, i.e., that a colonisation of reefs in some distant region will occur indirectly over several successive steps with intermediary reefs in between acting as stepping stones for this colonisation. For

example, consider three reefs, A, B and C, of which reef A has a known COTS population but reefs B and C are currently free of COTS (or have minimal background, non-outbreaking levels of COTS). If a link exists between reef A with a known population of COTS and reef B, COTS can spread from reef A to reef B directly. In contrast, if another link also exists between reef B and reef C, but there is no direct link between reefs A and C, there is no possibility that reef C will be colonised directly from reef A. However, it is possible that reef B will be colonised first and the COTS will then later spread from reef B to reef C, or, in other words, that the COTS population on reef A will eventually result in reef C being colonised indirectly. Obviously, this cannot occur without reef B being colonised first and therefore reef C cannot be colonised in a single generation. It is therefore possible to estimate the probability that COTS population from reef A would have indirectly caused colonisation of reef C after a certain number of years in two steps: first, by using the strength of the link from reef A to reef B to estimate the probability that reef B is colonised after a specified number of years, and second, using the belief that reef B is colonised at this time and combining it with the strength of the link from reef B to reef C. The mathematical details of these processes have been provided in recent publications that explore calculations of multi-step colonisation probabilities (Hock and Mumby 2015, Hock et al. 2016).

Estimating the probability that COTS populations will colonise reefs beyond direct connections from the primary source(s) is especially important when considering the spread of COTS outbreaks at large spatial scales, which makes it unlikely that there will be direct links between presumed initiation region and the region of interest. For example, COTS populations from primary outbreaks in the Lizard Island area will be likely to colonise other reefs in the Cooktown-Cairns region first before

eventually reaching the reefs in the Innisfail region. Therefore, the calculations that can estimate how colonisation could occur over successive steps are necessary to estimate which reefs are likely to act as intermediary steps during the propagation of COTS outbreak waves. Importantly, such ranking can also reflect both the prior beliefs that particular reefs are more or less likely to host outbreaks (Hock et al. 2016). This is achieved by setting the prior probabilities that reefs have COTS populations to reflect the initial bias towards reefs in some regions, such as those in the vicinity of Lizard Island, being more likely to experience primary outbreaks.

Performing this calculation for all reefs in the region makes it possible to rank those reefs based on the expectation that a COTS population on a reef will spread to certain reefs of interest, e.g., reefs outside of the initiation box that indicate outbreaks have now escaped this region, and do so over a specific time horizon, e.g. after a proposed number of years or colonisation steps. For the purpose of this project, these calculations entailed estimating the potential impact that putative COTS populations on each of initiation box reefs could have on reefs in the Innisfail region. Given that the paradigm this project explored was to prevent COTS outbreaks from spreading beyond the initiation box, the metric used was based on estimated chance that even a single reef in the Innisfail region will be affected. As such, reefs in the initiation box can be ranked according to the expectation that a COTS population on them will have non-zero impact (that is, cause at least one colonisation event) on the reefs in the Innisfail region. This expectation of non-zero impact (or ENZI) can then be used to for decision making in surveys and decide which reefs should be prioritised for surveys if their current state of COTS population is unknown. Analogously, once the location of COTS populations and/or outbreaks become known, either from prior knowledge or through novel detection, these estimates can

help decide which of the known populations/outbreaks would need to be prioritised for eradication in an attempt to minimise the spread of its larvae and therefore its future regional impact.

A key advantage of the described approach is that the priorities for surveys and eradication efforts can be dynamically updated as the situation in the field continues to develop (Hock et al. 2016). As such, targeting priorities can be adaptively updated to reflect the newly discovered COTS populations, which could occur either because surveys have now detected them or because the populations have continued to spread to new reefs over time. To do this adaptive updating of management targets in the current project, the ENZI value of the newly discovered populations can simply be compared to those that have been known previously to decide whether these newly detected COTS populations are expected to have a higher probability of spreading on to the Innisfail region and should therefore be prioritised for eradication. This ensures that the populations on reefs that are expected to have the most significant influence on the management objective (i.e., spreading beyond the initiation box to the Innisfail reefs) are always preferentially targeted regardless of which, or how many, reefs currently have COTS populations.

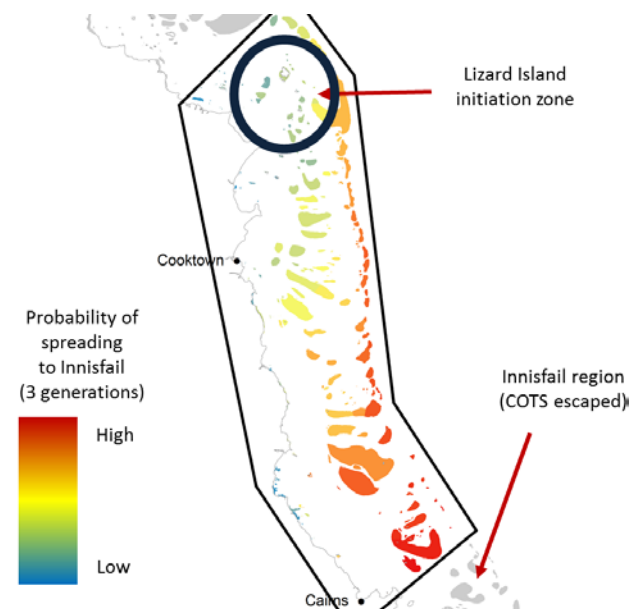
Mathematical details of the ENZI calculations, as well as a method on how such metric could be used to guide adaptive targeting solutions of COTS populations, has been recently published by Hock et al. (2016). However, although the fundamental calculations and the principles of adaptive targeting of source populations are the same, there are also some differences between the published work and the calculations performed for the purpose of the current project. Notably, fewer dispersal simulations to derive the connectivity estimates were available at the time of the preparation of the published study. Also,

rather than estimating the impact of reefs on the Innisfail region, management decisions in simulations by Hock et al. (2016) were guided by the expected impact of putative COTS populations on a set of reefs estimated to possess high-value tourism sites, most of which are located in the Cooktown-Cairns region. Finally, although Hock et al. (2016) provided the mathematical means to implement priori beliefs that certain reefs will be more or less likely to host COTS populations, there were no a priori assumptions about primary outbreaks being more or less likely to be initiated on specific reefs, as the aim of the published study was to derive ENZI values exclusively from the dynamic spread of the COTS populations in the system.

For the purpose of this report, the ENZI calculations were performed in concordance with the stated project goals. This entailed using the prior probabilities that were analogous to those used for the initiation of primary outbreaks in the population models; specifically, this meant giving 50x higher weight to mid-shelf reefs around the Lizard Island and Green Island (see population model parts of this report). The objective was to estimate the probability that COTS will spread from a focal reef to at least one reef in the Innisfail sector using mean beta estimates for each link connecting a respective source-sink pair. Given that around 6 years have passed since local COTS outbreaks in the current wave begun to appear around Lizard Island (around 2010) and the present when outbreaks are becoming established in the Innisfail region (2016), and considering that it usually takes 2 years for COTS settlers on a newly colonised reef to mature and reproduce (Pratchett et al. 2014), the ENZI probabilities were calculated over three successive colonisation steps.

The results indicating that the probability that COTS could spread from each of the reefs in the initiation box to at least one reef in the

Innisfail region are shown in **Figure 15**. As indicated in the risk map, this probability will naturally be the highest for reefs in the southern parts of the initiation box that are the closest to the Innisfail region and therefore likely to have strong direct connections to reefs in the Innisfail region. The probability is generally reduced with the geographical distance and the presence of reefs in between a focal reef and Innisfail region; reefs further north appear less likely to directly contribute to the colonisation of the Innisfail reefs.



**Figure 15.** Probability that a reef could supply COTS larvae to the Innisfail region in 3 generations (dispersal possible among all reefs).

While the overall trend of decreasing probability of larval supply with increasing distance from the Innisfail region, it is also possible to use this approach to distinguish among reefs that are approximately the same distance away from the Innisfail region. Judging by the current surveys of COTS in the region (MacNeil et al. 2016), these estimates should preferentially focus on determining survey and control priorities among mid-shelf reefs in the

initiation box region. A strategy that adaptively and preferentially targets reefs with higher estimated probability of spreading the COTS populations towards the Innisfail region can therefore be devised in order to determine priorities for surveys and eradication efforts. An example of how such strategy can be devised and implemented is given by Hock et al. (2016).

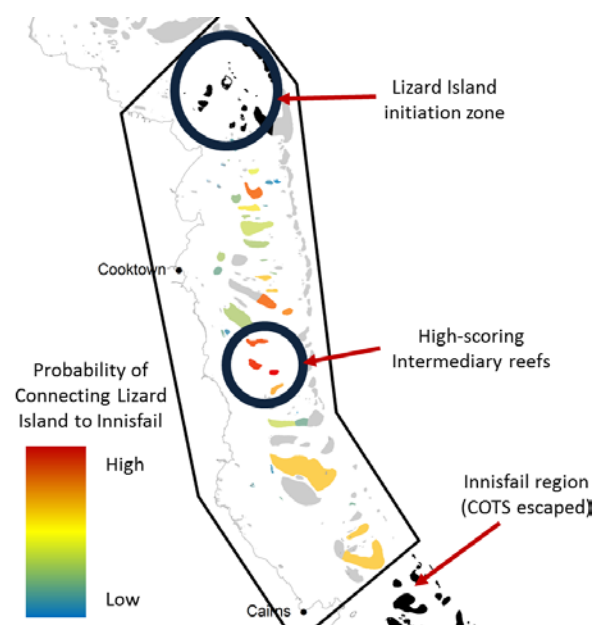
The results also indicate that the outer shelf reefs further north tend to have direct links to Innisfail reefs. This likely occurs because of the prevalent southward direction of the oceanic currents that run parallel to the outer reef matrix (Mao and Luick 2014) and consequently the unobstructed transport of larval particles over open water as opposed to transport through a dense reef matrix. However, although connectivity patterns suggest that transport to Innisfail from outer shelf reefs as far north as Lizard Island is certainly possible, field surveys indicate that these outer shelf reefs have been largely free of COTS. As such, the known absence of COTS populations on outer shelf reefs makes it unlikely that Innisfail region will be colonised via these reefs.

The results of connectivity analyses shown in **Figure 16** underpinned surveillance and cull options involving an ENZI search pattern. We note that extensions to the connectivity network analysis enable a richer understanding of connectivity, including:

- Identifying the likely routes of COTS dispersal through the initiation box (Figure 16).
- Estimating the connectivity risks when COTS larvae disperse only via mid-shelf reefs (Figure 17).

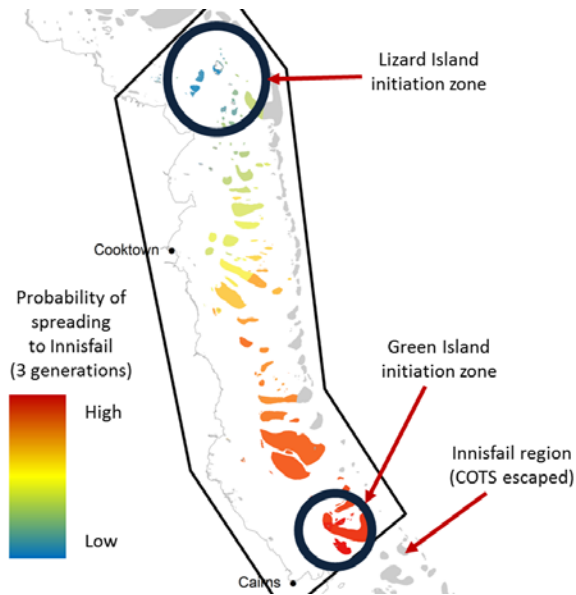
- Estimating risk of directly transmitting COTS larvae to Innisfail reefs (Figure 18).
- Estimating the number of stepping stones from the Lizard Island initiation zone (Figure 19).

Applications of these extensions are the subject of ongoing research.

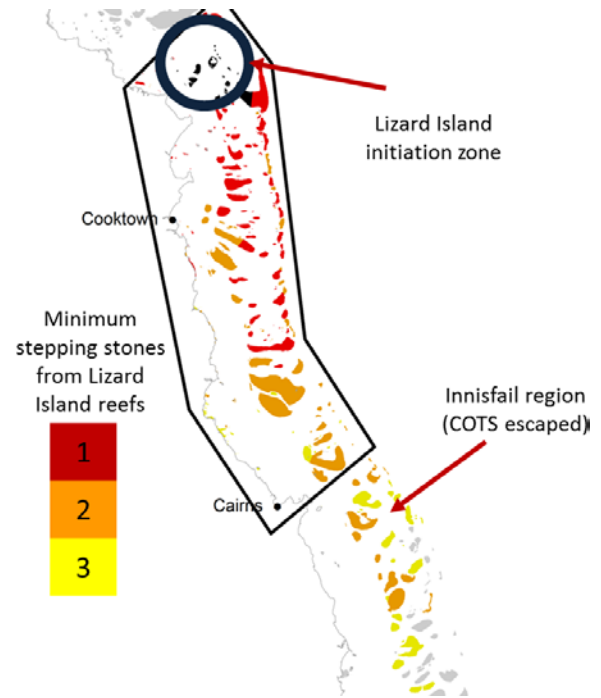


**Figure 16.** Probability that a reef could will act as an intermediary connecting reefs in the Lizard Island initiation zone with reefs in the Innisfail region. This is based on the probability of a reef being an intermediary on any of the ten most likely paths from each Lizard Island reef to any of the Innisfail reefs. The figure also highlights a region approximately half-way though the initiation box that contains many of the high-scoring intermediary reefs.

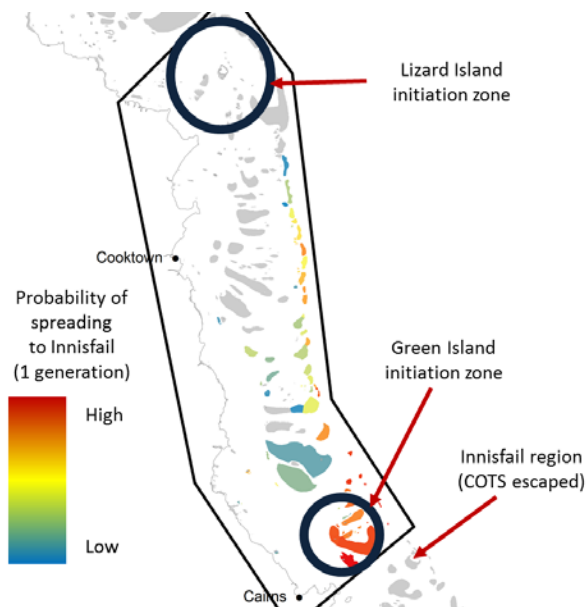




**Figure 17.** Probability that a reef could supply COTS larvae to the Innisfail region in 3 generations if dispersal is only possible among the mid-shelf reefs.



**Figure 19.** Minimum number of stepping stones needed to reach a reef from any of the reefs in the Lizard Island initiation zone. This is equivalent to the worst-case scenario when all connectivity links will result in colonisation regardless of their strength, and therefore each stepping stone represent the distance to which COTS can spread in a single generation.



**Figure 18.** Probability that a reef could supply COTS larvae to the Innisfail region in a single generation (dispersal possible among all reefs).

## 5.2 Synthesis Model

*The model was developed and analyses conducted by AIMS and the University of Queensland.*

We developed a stochastic dynamic ensemble model to simulate the prospects for successful arrest of outbreak waves under different surveillance and cull operations. Success was defined here as prevention of CoTS spread beyond (south of) the Innisfail sector. After simulating a primary outbreak within the initiation zone, at each monthly time step the model reports the number of reefs at outbreak densities among (a) the 284 reefs within the initiation zone, and (b) the 122 reefs within the Innisfail sector.

For reefs within the initiation zone, the model was composed of four elements:

- CoTS population dynamics
- Coral response modelled as a predator-prey relationship,
- CoTS detectability and
- Surveillance and cull operations (**Figure 20**).

### CoTS Model

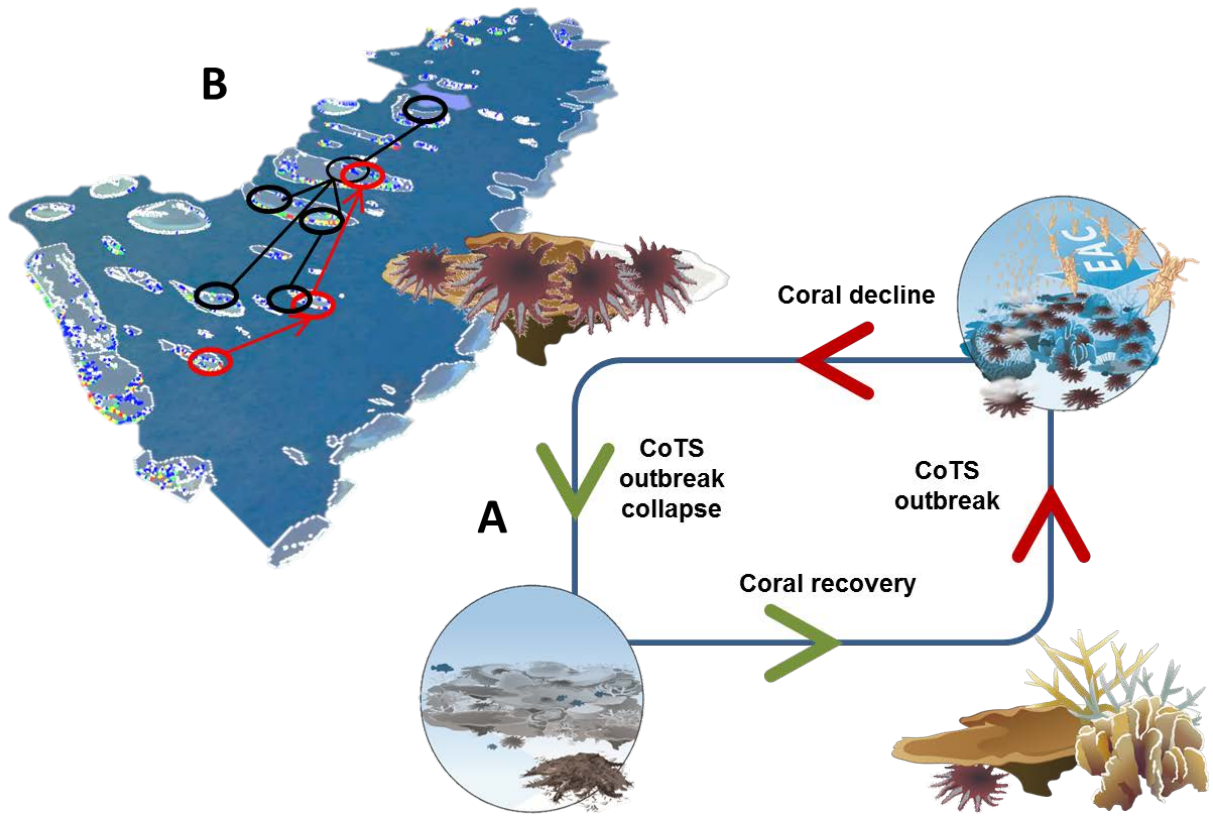
The CoTS model implemented here was an empirical semi-individual agent-based model structured around the current scientific understanding of the species. It was modified to accommodate computational overheads associated with spatial dynamics in a domain with over 284 reefs in the initiation zone, including the connectivity matrices described above. We adopted a cohort-based approach,

whereby starfish that recruited to the same reef at the same time were pooled into a single cohort and each cohort is an agent in the simulation. This significantly shortens time required to complete each simulation.

We assumed self-recruitment was 20% of fertilized eggs produced by starfish on any one reef, and the remaining 80% dispersed to other reefs according to the CoTS connectivity model described above. The same approach as in Hock et al. (2014) was used to simulate the COTS reef connectivity network for the 2008/2009, 2010/2011, 2011/2012 and 2012/2013 spawning seasons. The results of preliminary analyses of the four year connectivity data show a strong temporal variation in reef connectivity. Therefore to account for the temporal variation in the simulation, the COTS connectivity between any two reefs was assumed to have a beta distribution with parameters estimated from the available data.

### Coral Response Model

Although more sophisticated models have been developed for the modelling of coral dynamics, a simple coral response model was developed for the purpose of this study. The model simulates the changes in monthly coral cover as was found to be sufficient. Here, we assumed the rate of changes in monthly coral cover was a function of net coral growth (growth and natural mortality) and the predation pressure from CoTS. The amount of absolute change in coral cover depended on the rate of change and the initial coral cover of a reef.



**Figure 20.** Schematic overview of the dynamic synthesis model used in this study to assess surveillance and cull options. The overall simulation model is composed of four elements, CoTS, coral, detectability and simulations of surveillance and cull operations. The model was constructed as a simplified predator-prey model that accounted for the life-history characteristics of predominantly CoTS (A), and integrated with CoTS connectivity and abundance and the operation of surveillance and cull teams (B). The black paths and circles on the map (B) indicate a few of the many paths of connectivity. Red arrows and circles represent one of many potential search patterns taken by a surveillance and cull boat. The model simulation was carried out for 284 reefs in the initiation zone. See also **Figure 5**

Coral growth and predation pressure from CoTS were modelled differently for outbreak and non-outbreak conditions. We used a piecewise regression model to describe the rate of change in coral cover, such that:

$$\begin{aligned} \Delta C_{rt} | d_{rt}, a_0, b_0, a_1, b_1 &\sim N(\mu_t, \tau) \\ \mu &= 0.001(a_0 + b_0 d_{rt}), \text{ if outbreak} \\ \mu &= 0.001(a_1 + b_1 d_{rt}), \text{ if non outbreak} \end{aligned}$$

where  $\Delta C_{rt}$  and  $d_{rt}$  are rate changes in coral cover and the density of CoTS on reef  $r$  at time  $t$ ;  $a_0$  and  $a_1$  are the baseline changes and parameters  $b_0$  and  $b_1$  are the effect size of CoTS density during the outbreak and non-outbreak period, respectively.

Coral cover for time  $t+1$  for reef  $r$  is thus

$$C_{rt+1} = C_{rt} + \Delta C_{rt} C_{r0}$$

where  $C_{r0}$  is the initial coral cover for reef  $r$ .

CoTS adult density,  $d_{rt}$ , is described as

$$d_{rt} = \frac{N_{rt}}{N_{r0}}$$

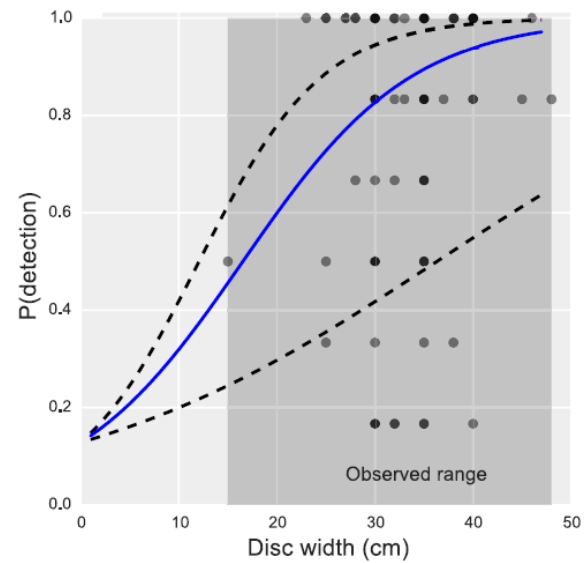
where  $N_{rt}$  and  $N_{r0}$  are the population size of CoTS above 200mm disc width at time  $t$  and time 0.

For each reef  $r$ , we assumed the initial coral cover,  $C_{r,0}$ , was the same as that in

2008/2009, immediately preceding the current outbreak wave. For reefs within the initiation zone where 2008/09 coral cover data was available, those data are used directly in the simulation as the initial coral cover ( $c_{r,0}$ ). For locations where 2008/09 coral cover was not observed, we used the mean coral cover of reefs in the same shelf position to infer the initial coral cover (i.e. inner, mid and outer reef). The mean estimates for inner, mid and outer reefs were 19%, 15% and 20% respectively.

### Detectability of COTS Outbreaks

Even under permanent and constant surveillance, the likelihood of detecting a COTS outbreak varies with the starfish population density, population size structure and individual detectability. In a study under this Partnership, MacNeil et al (2016) estimated the relationship between size of individuals and their detectability (**Figure 21**). The authors found the probability of detecting animals larger than 300mm was close to 1, but declined exponentially with a decrease in starfish size. We used findings of MacNeil et al (2016) to characterise the probability of false negatives, whereby surveillance operations visit a reef at outbreak density but declare it at non-outbreak density because of failure to detect sufficient adults. The probability of a false negative diminishes with surveillance sampling effort. Similarly, the model included culling down to a *perceived* threshold delineating outbreak and non-outbreak conditions (7.1 CoTS ha<sup>-1</sup>). The size class distribution of non-detected adults that remained after a culling operation was a function of the pre-cull size class distribution and the detectability function shown in **Figure 21**.



**Figure 21.** Estimated relationship between starfish size( median) and detectability (blue line), with 95% uncertainty intervals (dashed lines). Source: MacNeil et al. (2016)

### Surveillance and Cull Operations

We used an individual reef as the unit of operation. For each month of operation, we assumed  $N_s$  reefs were visited no more than once by surveillance or cull teams (i.e.  $N_s \leq 284$ ). Also, we assumed that the probability that a reef  $r$  is visited in any month  $t$  depended on the search methods deployed.

Three adaptive search methods were included and compared in this study. Firstly, the naïve (uninformed) search method was used as the baseline, which assumed searching without prior knowledge of CoTS dispersal mechanisms, hence all reefs in had equal probability of being visited. Secondly, we used a stratified random and thirdly an ENZI search method (see above under connectivity). Both were based on existing knowledge of broad-scale north-south dispersal and an understanding of finer scale hydrodynamics affecting CoTS outbreak spread.

## Simulation Settings, Additional Assumptions and Sensitivity Analysis

Although the exact number of reefs involved in triggering a primary outbreak within the initiation zone is unknown, we used the AIMS long term monitoring data to inform a coarse estimate. In any single model run we used a random sample from a Poisson distribution with  $\mu = 7$  to simulate the number of reefs associated with a primary outbreak in the first time step. Mid-shelf reefs within an 18km radius of Lizard or Green Island were assigned a higher probability of being selected as primary outbreak reefs in simulations than other reefs (by a factor of 50). For each of the 675 surveillance and cull options, 5000 simulations were run over 360 months (30 years). All simulations were implemented in C++, and run on the Australian Institute Marine Sciences high performance cluster computer, an Intel Xeon E52690 v2 3.0GHz.

performing options at five discrete levels of resourcing (**Table 8**).

We found that a high sampling effort (50 ha covered per reef) was more effective than low sampling effort at all resourcing levels. In general, scheduling of surveillance and cull operations based on connectivity dynamics (ENZI) performed well, especially as the level of resourcing increased.

Modelling results showed that the deployment of even modest surveillance and cull resources can reduce the spread and intensity of outbreaks, but that the prospects for actually stopping an outbreak wave are modest, at best. Unsurprisingly, the probability of simultaneously reducing all reefs within the initiation zone to below outbreak densities increases with level of resourcing, but even with 20 surveillance boats and 40 cull boats operating in the initiation zone the best estimate for the time it takes to achieve these conditions is 24 years (**Table 9**).

### 5.3 Model Results

Digesting outcomes for all 675 options is unnecessary. Here we present only the best

**Table 8.** The best performing surveillance and cull options for five discrete levels of resourcing within the initiation zone.

Label	Number of surveillance boats	Number of cull boats	Sampling effort	Search pattern
1 - 6	1	6	50 ha	random
5 - 10	5	10	50 ha	north emphasis
10 - 20	10	20	50 ha	ENZI
20 - 20	20	20	50 ha	ENZI
20 - 40	20	40	50 ha	ENZI



**Table 9.** Prospects for simultaneously reducing all reefs within the initiation zone to densities below outbreak levels under five discrete levels of resourcing

<i>Option</i>	<i>Probability of successful treatment within the initiation zone</i>	<i>Median time to successful treatment within the initiation zone (years)</i>
1 - 6	0.098	30
5 - 10	0.175	25
10 - 20	0.325	24
20 - 20	0.307	24
20 - 40	0.384	24

Successful treatment of the initiation zone is only part of the solution. Mop-up operations need to locate and successfully cull starfish on reefs at outbreak densities within the Innisfail sector. In general, the number of reefs at outbreak densities in any one year among the Innisfail sector's 122 reefs was small (typically five or fewer). But outbreak waves would only be arrested if (a) all reefs could be found and (b) all reefs successfully treated in each and every one of the years in which one or more reefs were at outbreak densities. We estimated 25 boats are necessary to visit all 122 reefs in the Innisfail sector within a single year and perform surveys of sufficient intensity that the chance of a false negative can essentially be ignored. Might it pay to deploy fewer mop-up boats than the 25 required for a census?

The answer is no. We assumed a hyper geometric distribution for detection of outbreak reefs in the Innisfail sector under various levels of resourcing for mop-up operations. Scenarios involving 10 or fewer

mop-up boats almost never succeeded in detecting all reefs at outbreak densities in the Innisfail sector each and every year (implying inevitable downstream spread of the outbreak wave). Deploying one boat fewer than that required for a full census, or 24 boats, results in a dramatic halving of the probability of successful detection of all outbreak reefs (**Table 10**). The number of annual culling operations required in the Innisfail sector is less than the number of years to successful treatment within the initiation zone reported in **Table 9** because it takes several years for the outbreak wave to reach the Innisfail sector. The number of years required for (successful) mop-up culling operations for our subset of best performing options is shown in **Table 11**. The probability of consistently succeeding over  $m$  mop-up years is  $p_s^m$ , where  $p_s$  is the per annum probability of success.

**Table 10.** Probability of successful detection of all reefs at outbreak density in the Innisfail sector throughout a 30-year surveillance and cull operation under five options for resourcing in the initiation zone and three options for resourcing of mop-up operations in the Innisfail sector.

Option	Number of mop-up boats deployed		
	10 boats	24 boats	25 boats
1 - 6	0.0002	0.50	1.00
5 - 10	0.0003	0.51	1.00
10 - 20	0.0004	0.52	1.00
20 - 20	0.0005	0.53	1.00
20 - 40	0.0005	0.53	1.00

**Table 11.** Sensitivity of overall mop-up success to  $p_s$ , the per annum probability of successful mop-up.

Option	Number of years required for mop up	Overall probability of successful mop up in the Innisfail sector		
		$p_s = 0.50$	$p_s = 0.80$	$p_s = 0.90$
1 - 6	23	0.000	0.006	0.089
5 - 10	19	0.000	0.014	0.135
10 - 20	18	0.000	0.018	0.150
20 - 20	18	0.000	0.018	0.150
20 - 40	18	0.000	0.018	0.150

Using an upper bound on what might be considered plausible, we assign a value of  $p_s = 0.9$  and use the last column in **Table 11** in our calculations of the overall probability of successful arrest of outbreak waves within the Innisfail sector and within 30 years of initiation. Results in **Table 12** for ‘probability of success’ are the product of

- The probability of successful treatment within the initiation zone (**Table 9**)
- The probability of successful detection within the Innisfail sector, assumed to be 1 for 25 boats (**Table 10**)

- The probability of successful mop-up cull treatment, assuming  $p_s = 0.9$  (**Table 11**).

These results are at the optimistic end of plausible bounds for the effectiveness of surveillance and cull operations. Where detectability is less than perfect and only partial culling if possible, surviving adults can quickly grow to a reproductive size that potentially negates the initial cull effort via a *Hydra* effect (Abrams 2009). We can identify four failure pathways that have been ignored or treated favourably in our modelling:

- Primary outbreaks that subsequently lead to waves of downstream streams

outbreaks may originate outside the delineated initiation zone.

- CoTS adults may aggregate at depths beyond the reach of free swimming cull divers at sufficient densities for downstream spread of an outbreak.
- The per annum effectiveness of cull operations in the Innisfail sector ( $p_s = 0.9$ ) is unlikely to be as high as we have estimated in analyses presented here (**Figures 22 and 23**)
- Larvae may disperse beyond the Innisfail mop-up sector in sufficient numbers for

downstream spread of an outbreak (**Figure 24**).

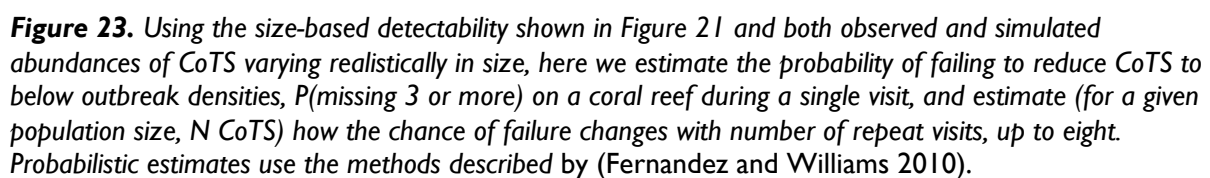
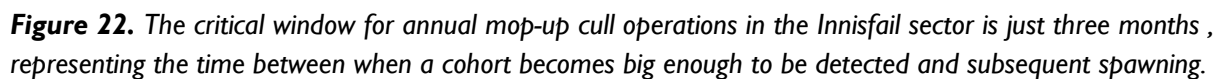
In section 9 below we assign collective non-zero probabilities to these failure modes to explore the sensitivity of outcomes to uncertainty.

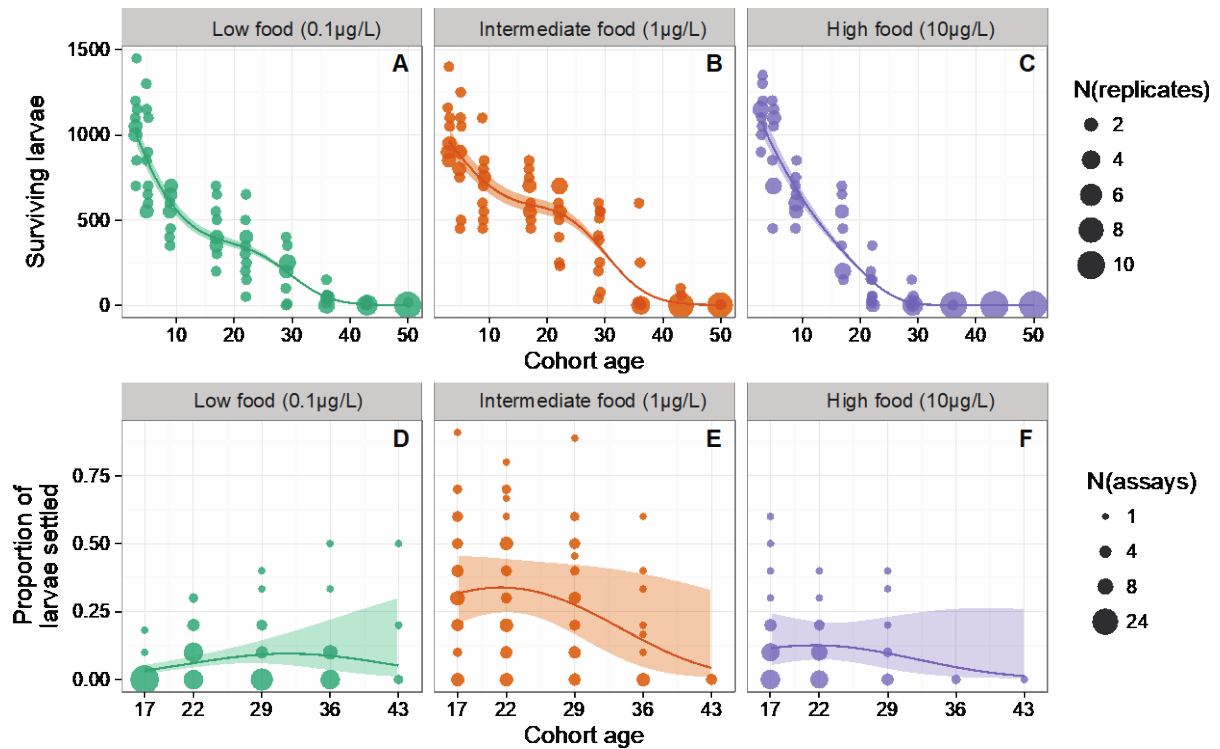
**Table 12.** The overall probability of successfully arresting outbreak waves in the initiation phase under five resourcing options. Outcomes for all options include 25 boats used in mop-up operations in the Innisfail sector and high effectiveness of mop-up operations ( $p_s = 0.9$ ).

Option	Probability of success	Success rate	Mean return time of outbreaks
			that escape
1 - 6	0.009	1 in 115 outbreaks	16.0 years
5 - 10	0.024	1 in 42 outbreaks	16.3 years
10 - 20	0.049	1 in 20 outbreaks	16.7 years
20 - 20	0.046	1 in 22 outbreaks	16.7 years
20 - 40	0.058	1 in 17 outbreaks	16.9 years

The potential gains to be made through improved detectability are underscored in **Table 13**. We repeated simulations for two of the better surveillance and cull options under conditions of perfect detectability. Even under relatively low levels of resourcing, the return time of outbreaks could potentially be extended beyond 26 years – a superior outcome to any of the options we explored for change in land use and management practice (see **Table 7**). Of course, attainment of *perfect* detectability is unachievable. But if technological advances can make material gains in improving detectability, then surveillance and cull operations may contribute to arresting outbreaks in the initiation phase.

Option	mean return time of CoTS primary outbreaks (years)
Status quo	15.9
1 surveillance boat, 6 cull boats, <i>imperfect detectability</i>	16.0
20 surveillance boats, 40 cull boats, <i>imperfect detectability</i>	16.9
1 surveillance boat, 6 cull boats, <i>perfect detectability</i>	26.4
20 surveillance boats, 40 cull boats, <i>perfect detectability</i>	36.7





**Figure 24.** As part of this Accelerate Partnership, the ARC Centre of Excellence for Coral Reef Studies explored larval survivorship under various levels of chlorophyll concentration. Results suggest past estimates have underestimated the length of time larvae can survive pre-settlement, and by extension, have also underestimated the distances larvae can disperse. Source: (Pratchett et al. 2017).



## 6. Benefit-cost analyses

*The framework for benefit-cost analyses was led by Natural Decisions and AIMS.*

There are many investments that can lead to better environmental outcomes, but not all such investments are worthwhile. Sound public decision-making rests on credible benefit-cost analyses, where ratios greater than 1 imply net benefit. Using improvement in expected coral cover as an indicator of benefit, we developed a framework for benefit-cost analysis that enabled direct comparison of both land use and surveillance and cull options.

### 6.1 Costs of implementation

#### *Land Use and Management Practice*

*The costs of options for land use and management practice was led by Natural Decisions with significant contributions from Terrain NRM*

The Investment Framework for Environmental Resources (Pannell et al., 2012) was used to condition benefits and estimate costs associated with land use change and adoption. The INFFER framework has been developed specifically for environmental projects involving investment in the use of private land and the behaviour of private landholders. It includes judgments of

- technical feasibility,
- landholder adoption,
- socio-political risks,
- lag-times in project implementation and the realisation of benefits, and
- uncertainty of long-term funding

For the land use and management options the lag time until full benefits occur was assumed to be 10 years. We have assumed a constant rate of funding and implementation for management practice and land use change to be spread across the 10 years. Costs were discounted at a rate of 5% over our 30 year time horizon.

Farm size can have an important impact on cost because practice change is cheaper on large farms compared with small ones (van Grieken et al. 2014). Using data available on farm size from Paddock to Reef adoption work we ascribed the proportion of farm areas into three categories; below 100 ha (classified as small farms), 101-200 ha (medium farms) and >200 ha (large farms).

The summary unit costs assumed for the management practice change and land use change options (**Table 14**) were assigned to different basins in the Wet Tropics corresponding to each of the eight options.

#### *Land-management practice change to at least all B*

The assumptions we made were for upfront costs:

- The total upfront costs were assumed to be spread evenly across the 10 year project implementation period
- Regulations were applied to land in D practice. The only cost applied was the \$13.33/ha applied every year for the 10 year project implementation period.
- For land moving from C to B practice (namely all land that started in both C and D practice), the upfront costs were applied once, with extension costs applied annually for the 10 year implementation phase.

And for maintenance costs:

- Maintenance costs (assumed to be 10% of upfront costs) were applied annually
- It was assumed that only half the extension resources would be required to provide on-going advice to farmers.

### *Land-use change*

For upfront costs:

- As for the practice change scenarios the initial project implementation phase was assumed to be 10 years.
- For conversion to low-input grazing, land was assumed to be compulsorily purchased for \$12,000/ha (applied to all farm sizes), covenanted and then re-sold. The income from re-selling the land was estimated to be \$5,866/ha and thus the net cost/ha was calculated as \$6,184/ha.
- For conversion to biodiversity conservation, a purchase price of \$12,000 was assumed. A further \$18,750/ha was also assumed to rehabilitate and replant cane land for biodiversity habitat.
- Personnel costs to enable people to work with landholders as part of the land acquisition process were assumed to be at the same level as the extension costs associated with land management practice change during the 10-year implementation period.

### *For maintenance costs*

- For low input grazing land no maintenance costs were assumed on the basis that land was covenanted and only allowed to be used for low input grazing. Purchasers would be well aware of this and there was assumed to be no need for public investment to subsidise maintenance.

- For the biodiversity conservation scenario, the land is assumed to be publicly owned and to not generate an income. Maintenance costs were assumed to be \$160/ha/year being required annually for pest and weed control.

**Table 15** shows the improvement in return times of outbreaks alongside costs for the eight options for change in land use and/or management practice.

**Table 14.** Summary of assumed costs (\$/ha) used for cane farms in the Wet Tropics (unit costs were taken from (Alluvium 2016) <sup>B</sup>.

(a) Small farms (<100 ha)

<b>Management change</b>	<b>Upfront</b>	<b>Maintenance</b>	<b>Extension</b>	<b>Regulation</b>
Move from D to C practice	0	0	0	13.33
C to B practice <sup>A</sup>	345	34.50	21.82	0
Land use change to low input grazing	6,134 <sup>B</sup>	0	21.82	0
Land use change to biodiversity conservation	30,750 <sup>B</sup>	160	21.82	0

(b) Medium sized farms (100-200 ha)

<b>Management change</b>	<b>Upfront</b>	<b>Maintenance</b>	<b>Extension</b>	<b>Regulation</b>
Move from D to C practice	0	0	0	13.33
C to B practice <sup>A</sup>	114	11.40	21.82	0
Land use change to low input grazing	6,134	0	21.82	0
Land use change to biodiversity conservation	30,750	160	21.82	0

(c) large farms (>200 ha)

<b>Management change</b>	<b>Upfront</b>	<b>Maintenance</b>	<b>Extension</b>	<b>Regulation</b>
Move from D to C practice	0	0	0	13.33
C to B practice <sup>A</sup>	78.00	7.80	21.82	0
Land use change to low input grazing	6,134	0	21.82	0
Land use change to biodiversity conservation	30,750	160	21.82	0

<sup>A</sup> Noting that this cost was also applied to areas originally in D practice once regulation has been used to move to C practice.

<sup>B</sup> Except that land purchase price used in the Alluvium report was \$10,000 assessed as market value. For this project land was assumed to be compulsorily acquired and we therefore added a premium of 20%.

**Table 15.** Improvements in return times of outbreak waves and their monetary costs for options dealing with change in land use or practice. Costs are discounted at 5% over a 30 year time horizon.

<b>Option</b>	<b>mean return time of primary outbreaks (years)</b>	<b>cost of implementation (\$M)</b>
S0	15.9	na
S1	17.3	\$51.9
S2	19.7	\$696.5
S3	19.7	\$3,517.4
S4	19.3	\$639.3
S5	17.1	\$207.7
S6	17.8	\$244.6
S7	19.1	\$623.8
S8	19.3	\$629.3

S1 (practice change only involving all cane land to at least B practice) was estimated to generate a material improvement in return times at modest cost. Whilst the extension of the return time of primary outbreaks was not as high as for the other options, the cost were much lower, as were the socio-political risks compared with the land use change options (S2 and S3). Conversely, whilst both land use change options were estimated to produce the most overall impact, they are expensive due both to costs (particularly land use change to biodiversity conservation) and high socio-political risks.

Among the best of the mixed management practice and land use change options appears to be S6, which involved land use change for Johnstone and Tully-Murray catchments to low input grazing and at least B practice for

the remaining catchments. Reasons for this being the best of the four mixed scenarios were due largely to the lower costs for targeted land use change in the selected Johnstone and Tully-Murray catchments compared with more broad scale land use change. The Herbert basin contains the largest cane area (estimated 63,945 ha of the total Wet Tropics cane area of 142,005 ha) and any scenario involving land use change in this catchment resulted in efficiencies simply due to costs associated with scale.

## Surveillance and Cull

*The costs of surveillance and cull options were based on data and judgments involving current operations supplied by GBRMPA, AMPTO and the Queensland Parks and Wildlife Service.*

On the basis of current operations, we estimate the annual cost of a single surveillance boat is \$2.09 million. For a single cull boat, the cost is estimated to be \$2 million each year. Below we estimate how much surveillance and culling effort is 'purchased' for these amounts. These estimates were used in the synthesis model described above.

We estimated a single surveillance boat in any one year would be deployed for 22 trips of 10 days duration. In any one year, we allowed five days to be lost to bad weather and 22 days (or one day per trip) would be dedicated to travel. In total, we estimated

$22 \times 10 - (5 + 22) = 193$  days of surveillance per boat per year.

We assume 18 fixed area plots of 5m radius is used to survey a single hectare, and a single diver is capable of searching 1.17 ha per day. If we have 6 divers operating in-water each day, a single boat can search  $6 \times 1.17 = 7$  ha per day. Now,  $(7 \text{ ha} \times 193 \text{ days})/12 \text{ months} = 113$  ha per boat per month.

The number of reefs that can be surveyed per boat per month depends on the sampling effort used:

- 10 ha per reef – 11.30 reefs per month
- 20 ha per reef – 5.65 reefs per month

- 50 ha per reef – 2.26 reefs per month

Over 284 reefs within the initiation zone, we estimate the average area available for treatment on a single reef is 167 ha.

On average, we estimated a boat comprising a team of  $4 \times 2 = 8$  divers each doing 4 dives per day could treat 16 ha in a single day.

As for surveillance, the total number of days available to culling per boat per year was estimated to be 193.

Now,  $(16 \text{ ha} \times 193 \text{ days})/12 \text{ months} = 257$  ha per boat per month. If a single reef averages 167 ha of treatable area, the number of reefs treated per boat per month  $= 257/167 = 1.5$  reefs per boat per month.

In **Table 16** we collate return times for outbreaks and costs for the subset of options for surveillance and cull that performed best at discrete levels of resourcing. When compared alongside results shown in Table 16 for change in land use and practice, we see that none of the surveillance and cull options are competitive. The lowest cost option for surveillance and cull (\$199.5 million) buys a mean return time of 16 years. The lowest cost option for change in land use or proactive (considerably cheaper at \$51.9 million) provides a return time of 17.3 years.

In subsequent analyses we retain surveillance and cull options 1-6 and 20-40 for illustrative purposes.



**Table 16.** Improvements in return times of outbreak waves and their monetary costs for the better performing surveillance and cull options. Costs are discounted at 5% over a 30 year time horizon.

<i>Option</i>	<i>Mean return time of primary outbreaks (years)</i>	<i>Cost of implementation (\$M)</i>
Do nothing	15.9	nil
1 - 6	16.0	\$199.5
5 - 10	16.3	\$395.8
10 - 20	16.7	\$721.6
20 - 20	16.7	\$1,051.7
20 - 40	16.9	\$1,373.0

## 6.2 Benefits

The return time of outbreaks is an indirect descriptor of the benefits of management intervention. Here we outline a model we developed to translate extensions of return times to GBR-wide estimates of coral cover under climate change uncertainty.

### *Modelling coral cover into the future under different management options for CoTS and uncertainty in system response to bleaching*

To estimate future coral cover over time we developed a Gompertz-based model for inter-annual dynamics that incorporates disturbance and subsequent rates of recovery. Rather than assuming linear rates of change between years, we quantified the rates of coral recovery from 1995 to 2015 among AIMS LTMP-surveyed reefs, and using our mechanistic model assumed coral cover on each reef ( $HC_i$ ) grows according to a constant intrinsic rate of increase ( $r_i$ ) between any

given year  $t$  and  $t+1$ . The model also estimates average effect sizes ( $\beta$ ) for five classes of disturbance (DIS) - storms, coral bleaching, CoTS outbreaks, and unknown (coral losses exceeding 5% between years with an unknown source) - that contribute to coral loss over the time series.

Because coral cover is bounded between 0 and 100, we adopted a hierarchical modelling method developed by (Fukaya et al. 2010) for benthic cover that includes a reef-specific density dependence parameter ( $\alpha_i$ ):

$$\ln(HC_{i,t}) = r_i + (1 - \alpha_i) \ln(HC_{i,t-1}) + \sum_{i,t} \beta_d DIS_{i,t}$$

A unique feature of the data of the reefs within the LTMP time-series includes 15 individual transects ( $j$ ) that are repeatedly sampled year on year, providing high power to detect temporal change in coral cover on specific transects. In addition, we allocated LTMP reefs into one of four habitat types ( $k$ ) - Acropora-dominated, mixed corals, soft corals, and Poritidae/Alcyonidae-dominated - that characterize the coral community within each reef.

To estimate the various parameters of the Gompertz model, we adopted a Bayesian approach, allowing easy specification of the nested transect<sub>i</sub>-reef<sub>j</sub>-habitat<sub>k</sub> structure:

$$\begin{aligned}
HC_{kij,t} &\sim \text{BIN}(100, l\mu_{kij,t}) \\
l\mu_{kij,t} &= \frac{e^{\mu_{kij,t}}}{100} \\
\mu_{kij,t} &= r_{ki} + (1 - \alpha_{ki}) \ln(HC_{kij,t-1}) \\
&\quad + \sum_{d=1}^5 \beta_d DIS_{ij,t} \\
\beta_{d1,...,d5} &\sim N(0, 1000) \\
r_{ki} &\sim N(r_k, \sigma_r) \\
\alpha_{ki} &\sim N(\alpha_k, \sigma_\alpha) \\
\sigma_r, \sigma_\alpha &\sim U(0, 100)
\end{aligned}$$

where  $DIS_{ij,t}$  represents one of the five disturbance types occurring at each reef between observation years  $t-1$  and  $t$ . Note that because of an LTMP sampling program change from annual to biennial surveys in 2008, only data from adjacent years was used in the most recent part of the time series.

To estimate the probability of any given reef experiencing a disturbance event in any given year, we developed a second set of probability models, using the same LTMP dataset, estimating coarse (average) disturbance probabilities from transects throughout the entire dataset. Specifically, we estimated the probability of a given reef ( $i$ ) within each habitat ( $k$ ) having ( $DIS_d=1$ ) or not having ( $0$ ) a given disturbance ( $d$ ) in each year:

$$\begin{aligned}
DIS_{dki,t} &\sim \text{BER}(p_{dk}) \\
\text{logit}(p_{dk}) &= \beta_{dk} \\
\beta_{d1k1,...,d5k4} &\sim N(0, 10)
\end{aligned}$$

This structure provided habitat-specific probabilities of each disturbance type in any given year.

The per annum probability of disturbance in any single habitat was assumed constant over time for storms, coral bleaching, and unknown sources. The per annum probability of CoTS disturbance varied according to a Weibull distribution estimated for each management scenario.

The rate of disturbance estimated for bleaching from past data is unlikely to be representative of the incidence of bleaching in the future. Based on Logan et al. (2013) we included three global bleaching projection scenarios that increase the annual probability of bleaching under RCP 4.5 (IPCC 2014). Specifically these are:

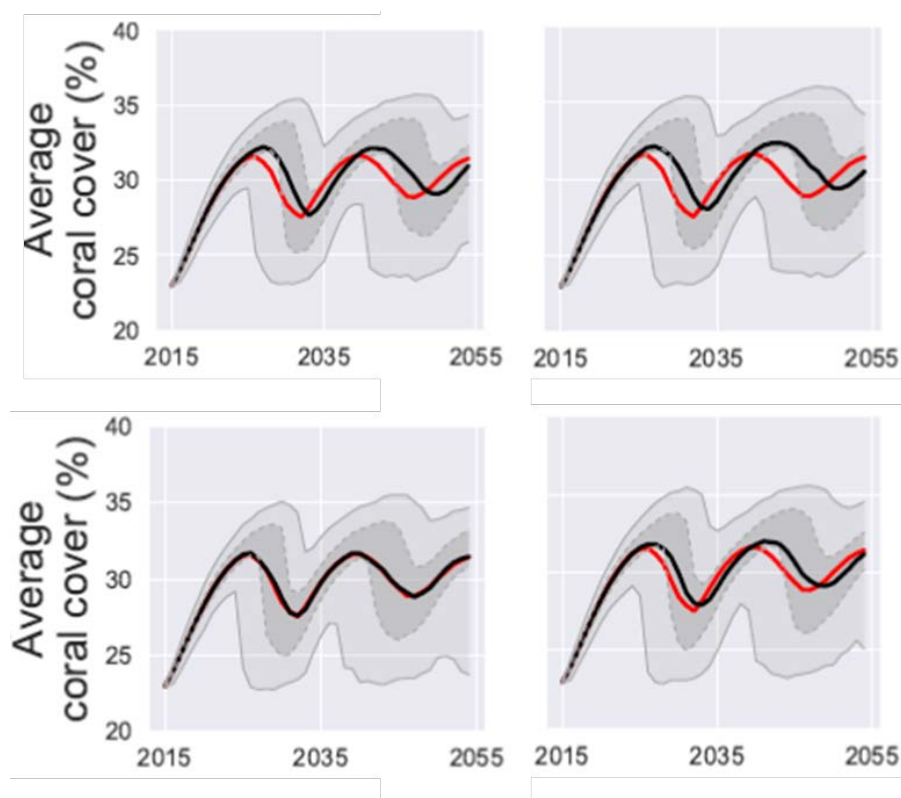
- No adaptive response - this is a projection of the NOAA GFDL ESM2M model out to 2100.
- Rolling climatology window - this is a projection that simulates corals' ability to adaptively respond to recent thermal history over an 80 year window.
- Temporary increase in bleaching threshold - this is a projection that simulates symbiont shuffling given a  $+1^\circ\text{C}$  threshold and a 5 year return time.

For each of these scenarios data from (Logan et al. 2014) were used, scaling the current rate of bleaching estimated from the LTMP data to the relative increases predicted globally.

We implemented the recovery rate and disturbance probability models using the PyMC3 package (<https://github.com/pymc-devs/pymc3>) for the Python programming language. Models were run for 10,000 iterations using the No-U-Turn Sampler

(NUTS) or Metropolis-Hastings where appropriate. All parameter traces were visually examined for evidence of convergence, and evaluated using Geweke's Z-scores comparing traces from multiple

chains. Examples of model outputs are shown in **Figure 25**.



**Figure 25.** Examples of the estimated temporal trajectory of coral cover under options for land use change (a and b) and surveillance and cull (c and d). Red lines show the ‘do nothing’ trajectory. Black lines show the improvement under the specified option, with 50% and 95% uncertainty intervals. Note that these trajectories make the naïve assumption that the incidence of bleaching in the future will be the same as that of the past and ignore impacts of climate change beyond bleaching. Although the absolute level of coral cover changed according to the assumed adaptive response to coral (results not shown), the contrast between options and do nothing (the red and black lines) was similar across all three bleaching response scenarios.

- (a) S1: land practice change, At least all B practice (A remains as A as default)
- (b) S6 : land use and practice change, Retire all Johnstone and Tully-Murray to low input grazing and at least B practice everywhere else
- (c) I – 6: surveillance and cull, 1 surveillance boat, 6 cull boats
- (d) 20 – 40: - surveillance and cull, 20 surveillance boats, 40 cull boats

### 6.3 The Value of the Great Barrier Reef and Implied Benefit-Cost Ratios

Stoeckl et al. (2014) assessed the total commercial (market) and wellbeing (quality of life) value of the entire GBR to be between A\$15 billion and A\$20 billion per year, much larger figures than previously estimated. These authors described their method as a 'whole ecosystem' approach to assessing both the importance (to overall quality of life) and the monetary value of various community-defined benefits, some of which align with various ecosystem services. They used a survey method to assess life satisfaction (LS) and then related this to a broad range of services. A lower bound on the value of the GBR is the \$4 billion per year associated directly with tourism (Deloitte Access Economics 2013). More recently, Deloitte Access Economics (2017) estimated a net present value of \$56 billion for the GBR using a time horizon of 33 years and a discount rate of 3.7%, equating to approximately \$3 billion per year. The authors emphasise that a valuation of \$56 billion is conservative

because it does not include the value Traditional Owners place on the Reef, ecosystem services, or existence value to the international community. We use \$15 billion per year as a point estimate for value of the GBR, and \$4 billion and \$20 billion as plausible lower and upper bounds, respectively.

We assumed a linear function relating coral cover and monetary value. At a value of \$15 billion per year, this value function equates an absolute change in coral cover of 1% to an annual loss (or gain) of \$652 million. The corresponding estimates for valuations of \$4 billion and \$20 billion were \$174 million and \$870 million, respectively.

We used the discounted difference in projected coral cover between do nothing and each option (**Figure 25**) to calculate total benefit. As for costs, a discount rate of 5% was used over our 30 year time horizon.

Results of benefit-cost analyses are shown in **Table 17**.

**Table 17.** Benefit-cost ratios for ten options for intervention aimed at reducing the impact of CoTS via prevention (change in land use or practice; S1 – S8) or arrest of outbreak waves (surveillance and cull; 1-6 and 20 – 40). Analyses use a 5% discount rate and assign a value to the GBR of \$15 billion per year. The best alternatives for change in land use and practice are highlighted, along with the two best options for surveillance and cull.

<b>Option</b>	<b>Bleaching response scenario</b>		
	<b>no adaptation</b>	<b>rolling window</b>	<b>temporary increase</b>
S1	67.1	80.6	82.3
S2	15.0	15.0	15.3
S3	2.9	3.0	3.2
S4	14.5	15.3	16.0
S5	15.6	17.0	15.7
S6	22.2	22.1	22.3
S7	14.3	14.6	14.6
S8	15.4	16.3	15.6
1- 6	2.7	4.4	2.2
20 - 40	2.5	2.6	2.5

## 7. Policy choice under uncertainty

The results shown in **Table 17** all report a benefit-cost ratio (BCR) greater than 1, implying that the net benefit of the subset of listed options makes investment in any one option worthwhile. However, the magnitude of the ratio varies widely among options. Of the options involving reductions in the incidence of nutrient enrichment, the BCR of S1 (all B practice) was highest with a range of 67 to 82 across the three scenarios for bleaching response under climate change. S6 (retire all Johnstone and Tully-Murray to low input grazing and at least B practice everywhere else) was second best, with a BCR of 22 that was essentially invariant across bleaching scenarios. The efficiencies of surveillance and cull options were relatively modest, with BCRs of 2 to 4.

The seemingly small extensions in the return times of outbreak waves reported in **Tables 15 and 16** may seem inconsistent with the finding of BCRs being uniformly greater than 1. Likewise, the consequential improvements in coral cover over a 30 year time horizon (e.g. Figure 25) may seem small. But this impression fails to comprehend the scale of improvement in outcomes associated with the better performing options. The GBR extends over a vast 348,000 km<sup>2</sup>. So for example, under the no adaptation scenario over a 30 year time horizon, the average annual improvement in coral cover relative to the do nothing option equates to an additional 1,500 km<sup>2</sup> of coral cover under S1, and an additional 2,400 km<sup>2</sup> for S6. These improvements represent very substantial benefits.

The BCRs reported in **Table 17** assume an annual value of \$15 billion for the whole GBR and an absolute conviction in a belief in

nutrient enrichment as a necessary condition for primary outbreaks (for options S1 to S8) and an absolute conviction in the adequacy of coverage of surveillance and cull operations (for 1 – 6 and 20 – 40). At the end of section 6.3 (dealing with land use) and 7.2.6 (surveillance and cull) we cautioned against absolute conviction. In **Tables 18 and 19**, we explore the sensitivity of outcomes to these assumptions. For S1 and S6, a valuation of the reef as low as \$4 billion and a belief in nutrient enrichment as a cause of primary outbreaks can be as low as 0.06 and 0.17, respectively, and still the benefit-cost argument for implementation is maintained (**Table 18**). That is, should decision-makers carry a belief that as few as 1 in 16 outbreaks are associated with enhanced nutrient enrichment, the benefits of implementing S1 would still cover costs. For S6, the critical threshold is a 1 in 6 belief. This key insight here is that options S1 and S6 are robust to uncertainty in the cause of primary outbreaks.

The same cannot be said for surveillance and cull options. Under a valuation of \$4 billion, the BCRs for the two options shown in **Table 19** are less than one, even where absolute conviction is maintained in a belief in adequate coverage. Where the valuation is \$15 billion, and consideration of the plausible failure modes listed at the end of section 7.2.6 leads to a belief less than about 0.4 in the adequacy of coverage, the costs of implementation outweigh benefits.

The outcomes of analyses for policy-makers are unambiguous:

- At a cost of \$52 million, implementation of option S1 (all B practice throughout Wet Tropics basins) is clearly worthwhile.
- Where fiscal policy is more accommodating, the greater benefits stemming from implementation of S6 (retire all Johnstone and Tully-Murray to low input grazing and at least B practice



everywhere else) at a cost of \$245 million is also clearly worthwhile.

Incorporation of benefits, costs and risks associated with investment is important to inform policy decision-making. Changing land management practices has the potential to generate the best value for money if it is well-resourced and targeted. The amount of money required to implement option S1 (upfront cost of around \$46 million and on-going maintenance payments of 10% upfront costs plus extension) is greater than incentives provided in the Wet Tropics region currently. Targeting of land management practice changes within Wet Tropics basins has potential to further increase cost-effectiveness.

We note that the INFFER analysis used here does not, for example, assume full ongoing adoption of B level practice throughout all areas of sugarcane production within the Wet Tropics. After accounting for probabilistic judgments associated with incomplete adoption, socio-political risks and risks to long-term funding, results for S1 assume only 42% of sugarcane production successfully

progresses to level B practice. Nevertheless, there are some major social and political challenges associated with land management practice change. To provide outcomes at the levels suggested here may require underpinning regulations or compliance to be strengthened. Provision of incentives at current levels and inadequate auditing is unlikely to achieve outcomes at the scale documented here.

Where the objective is reduction in the frequency of outbreak waves, intense surveillance and cull operations in the initiation phase using currently available technology is a relatively low-return investment.

While investment in land use change and improved management practice in Wet Tropics basins will reduce their frequency, outbreak waves are likely inevitable. *Once an outbreak is established and the objective of management switches to reducing downstream damage*, surveillance and cull has a key role to play in asset protection and limiting the rate and intensity of spread.

**Table 18.** Sensitivity of the two best land-use options to alternative valuations of the Great Barrier Reef and belief in the underpinning proposition, that nutrient enrichment is a necessary condition for primary outbreaks. Critical thresholds refer to the point at which the benefit-cost ratio is reduced to 1, expressed as a probabilistic belief in the underpinning proposition. Analyses use coral cover projections under the 'no adaptation' scenario.

Management option	GBR valuation		
	\$4B per year	\$15B per year	\$20B per year
S1	0.06	0.02	0.01
S6	0.17	0.05	0.03

**Table 19.** Sensitivity of the two best surveillance and cull options to alternative valuations of the Great Barrier Reef and belief in the underpinning proposition, that operations in the delineated initiation box represent adequate coverage. Critical thresholds refer to the point at which the benefit-cost ratio is reduced to 1, expressed as a probabilistic belief in the underpinning proposition. Analyses use coral cover projections under the ‘no adaptation’ scenario.

<b>Management option</b>	<b>GBR valuation</b>		
	<b>\$4B per year</b>	<b>\$15B per year</b>	<b>\$20B per year</b>
S9	-	0.38	0.28
S10	-	0.40	0.29

## 8. Future directions

This Partnership has shown that the challenge of preventing and combatting CoTS outbreaks in many ways resembles that of preventing and managing wildfires (Thompson and Calkin 2011). As CoTS outbreaks or wild fires start and develop, management objectives shift from a focus on protecting the whole landscape/seascape to one of prioritising high-value sites (Kalabokidis et al. 2012, GBRMPA 2016). When an outbreak or a fire is fully developed, the decision problem becomes one of triage (Bottrill et al. 2008) with an increase in risk of damage relative to available resources. From this perspective alone, it can be argued that investment in prevention and early detection/response, as was the focus of this Accelerate Partnership, should be the primary objective of any CoTS management strategy (Hoey et al. 2016), control program for species invasions (Pysek and Richardson 2010) or fire management service (Thompson and Calkin 2011). Efforts to contain or suppress outbreaks should thus be considered a plan B, but are nonetheless critical for the protection of key assets such as high-value tourism sites on the GBR.

Optimal management across all outbreak stages should be informed by uncertainties associated with the most important state-dependent processes, the probability of CoTS detection, and the likelihood of success (Chadès et al. 2011). Reference points that specify when management efforts move from one objective to another need to be identified and agreed upon in advance by co-managers (Irwin and Conroy 2013). Importantly, as an outbreak progresses, the logistical difficulty, coral damage, and costs of control efforts increase while the likelihood of protecting the ecosystem (the main objective) diminishes.

This places a high priority on prevention or early intervention as the a priori preferred strategy. In other words, early prevention represents the decision point with the greatest payoff in terms of ecosystem protection, analogous to the hierarchy of priorities in fire-fighting strategies (Alonso-Betanzos et al. 2003). However, if early prevention is infeasible or unlikely due to limited resources or technologies, a next step should be to decide on which of the competing hypotheses are most likely to drive further population spread (**Table 20**) and then invest accordingly in monitoring and cull technologies. Here, an active adaptive management approach that informs decisions via an integrated research and development strategy may enhance effectiveness (Williams et al. 2011). In the following, we first discuss opportunities around developing new technologies for CoTS surveillance and control and then propose some merits of an integrated and adaptive CoTS science and management strategy.

### 8.1 Opportunities for Innovative R&D to Increase CoTS Management Effectiveness

The CoTS problem on the GBR has no obvious Achilles heel - much research is still required to determine why outbreaks begin and end. However, integrated and model-assisted monitoring and cull operations, improved detectability, and enhanced effectiveness of control campaigns are three priority areas where research and development could immediately improve management outcomes.

## Early warning

If the nutrient-enrichment hypothesis is accepted as one mechanism contributing to primary outbreaks, then outbreak initiation can potentially be predicted from environmental conditions conducive to larval survival (Fabricius et al. 2010a, Brinkman et al. 2014) and the modelling of larval drift trajectories and likely points of settlement (Hock et al. 2014). However, despite significant advances in this research field under this Partnership, predictions of hotspots for CoTS outbreaks remain uncertain. Improved understanding of CoTS larval biology, more detailed insight into oceanographic factors, including El Niño–Southern Oscillation conditions (Wooldridge and Brodie 2015) that drive larval dispersal, and the role of other environmental variables in facilitating larval settlement and survival into young adults, would all increase the capacity to predict with accuracy where outbreaks start.

Again, early warning systems used in prevention planning and containment of wildfires could add value to CoTS outbreak prevention. Specifically, these are GIS-based decision-support systems providing a synoptic outlook of fire risks, updated in near real time based on a combination of remote sensing, on-ground environmental data and monitoring (Kalabokidis et al. 2012). The modelling efforts under this Partnership to integrate environmental data layers in the prediction of chlorophyll concentrations that drive trigger high CoTS larval survival and connectivity is a step towards identifying hotspots for outbreak initiation and spread. Lastly, to be effective as an early-warning and operational decision-support tool, such models need to be integrated with up-scaled and targeted reef monitoring, including the capacity to monitor juvenile CoTS.

## Starfish Detectability

Detecting CoTS on coral reefs is a challenge at every stage of the outbreak cycle. Juveniles or sub-adults situated inside the reef matrix are particularly difficult to detect using standard survey techniques (Pratchett et al. 2014) and represent a major source of uncertainty about the state of any outbreak. In surveys, detectability of large (>30 cm) adult starfish is around 90%, dropping to less than 20% for starfish <10 cm (MacNeil et al. 2016). Therefore, the development of monitoring methods that can identify aggregations of juveniles before they emerge as sub-adults represents a key research priority. However, high-resolution conventional monitoring of juvenile and sub-adult CoTS in large initiation zones would be slow and cost-prohibitive. One way to overcome this problem could be by using a three-tiered approach that (1) predicts the location of juvenile hot-spots using environmental / oceanographic models, (2) uses innovative, quantitative eDNA techniques (using CoTS-specific PCR primers, Uthicke et al. 2015a) to detect the presence of juvenile or cryptic sub-adults on reefs where (3) monitoring programs cannot detect adult CoTS.

## Effectiveness of control methods

A major gap in control efforts for CoTS is that small starfish cannot be controlled effectively using manual culling methods. Here, chemical signals offer a possible avenue for control enhancement as marine invertebrates respond to a spectrum of chemical cues (Hay 2009). Attractants, for example used in traps, have the potential to increase the density of small CoTS and increase the effectiveness of monitoring and control efforts targeting early outbreak stages. This approach has been used effectively to control sea lamprey (*Petromyzon marinus*) in North America (Wagner et al. 2006) and development of such methods for CoTS represents a key R&D opportunity (Hall

et al. 2017). Importantly, the efficiency gains in culling campaigns resulting from technology developments could, in combination with better intelligence analogous to those used in fire prevention and management, enable the suppression of starfish densities to below threshold levels. This would increase the likelihood of containing an initial outbreak, or alternatively, reducing the frequency of culling campaigns required to protect priority sites.

### *Can CoTS outbreaks be made to collapse earlier?*

Detailed knowledge of population weaknesses is an important principle for integrated pest management, and is used to guide resource allocation in time and space as well as to develop more specific control methods. New technology developments in the biosciences mean that processes involved in local CoTS population collapses can now be addressed in much more detail than previously possible. For instance, our knowledge of the echinoderm immune system is increasing rapidly with advances in molecular technologies (e.g. Fuess et al. 2015), and recently the CoTS genome sequence was released (Hall et al. 2017). This opens new avenues for identifying CoTS specific genetic pathways and traits, for studying potential dependencies of CoTS immune competence and disease susceptibility on environmental and physiological factors, and for studying CoTS immune responses to coral-produced chemical deterrents. Also, rapid developments in our ability to study the composition and function of CoTS-associated viromes and microbiomes enable studies of organisms that are not easily cultured in the laboratory, and this can inform the design of specific isolation strategies if required. Investment in these types of studies would fill an important knowledge gap in our understanding of natural CoTS population cycles. Importantly, such advances could also lead to the development

of safe, species-specific approaches for early induction of local CoTS population collapse. This would potentially represent an efficiency gain, as discussed above, and could be particularly effective if used during primary outbreak stages. Obviously, thorough safety testing and multi-disciplinary discussions of possible ecological and legislative implications would be required before new approaches could be tested in the field. However, the initial knowledge generation is risk-free and should be initiated during the current CoTS outbreak on the GBR.

## **8.2 An integrated strategy for CoTS management and research**

The complexity of CoTS outbreaks and uncertainties facing state-dependent decision-making at each outbreak stage requires an adaptive and integrated management and research strategy – ideally as a partnership with coordinated objectives and activities. Multiple hypotheses potentially explain environmental and biological processes driving different outbreak stages, and there are significant knowledge gaps in all parts of the outbreak cycle (**Box 1**). To be able to detect and contain primary outbreaks, for example, will require testing of: (1) techniques to detect and cull juvenile CoTS; (2) improved population models that provide accurate early warning of primary outbreaks based on environmental information and biological processes identified as key drivers; and (3) active filling of knowledge gaps that limit (1) and (2). By using an adaptive and integrated research and management approach, varying beliefs in different working hypotheses can be confronted with evidence and updated through active monitoring and research to support effective management decisions and actions. For example, resolving multiple alternative or linked hypotheses (e.g. nutrient-enrichment and predator-removal) will

require clarity in problem formulation among co-managers, willingness to experiment in speculative actions, and a monitoring program of sufficient resolution and power. Lastly, on the GBR, the alignment of the chain of CoTS management objectives during outbreaks with the different research and management challenges at different outbreak stages provides a blueprint for research and management prioritisation and integration. Where knowledge gaps mean high uncertainty around processes that drive outbreak dynamics, investments and efforts should be allocated to narrow such gaps. Likewise, where management effectiveness is limited by methodology, innovation around improved technologies should drive investments, again in a coordinated and adaptive science and management partnership.

## *Conclusion*

CoTS outbreaks on Indo-Pacific reefs resemble epidemiological and pest-control problems (Pysek and Richardson 2010), where the damage and challenge of containment escalates once the population becomes a primary outbreak. From this perspective, investment into early detection and containment is the preferred strategy if main

the objective is to protect or limit damage to the broader reef ecosystem and minimise cost. However, early containment is limited by (1) imperfect starfish detection, (2) low predictive power of CoTS population models, (3) uncertainty in the prediction of initiation hot-spots from environmental data and (4) risks of reinfestation following control campaigns due to limited cull effectiveness.

Technological innovations to overcome these bottlenecks including detectability represent key research and development priorities, best delivered through an adaptive and integrated management and research partnership. Although early detection, prevention or containment of primary outbreaks is a key objective for ecosystem protection, improved methods to control CoTS in later outbreak stages continue to be relevant for the protection of high-value locations where containment of primary outbreaks fail. Lastly, given the cryptic nature of CoTS and the spatial scale of the problem on the GBR, the identification of means to compromise CoTS disease resistance and to safely promote early population collapse from endemic pathogens needs to be an active area of research.



**Table 20.** Elements of an integrated CoTS monitoring, management and research strategy across the decision points of outbreak stages and associated management objectives.

Outbreak stages	Management objectives	Intelligence/ monitoring	Management tools	Research products	Knowledge gaps
Latency	Prevent outbreak conditions.	Meteorology, water quality, oceanography, juvenile and adult starfish densities.	Land-use management, fisheries management.	Early warning system based on environmental information and effective CoTS monitoring.	Threshold conditions for primary outbreaks.
Initiation	Contain aggressively.	Early warning, identification of primary outbreak reefs via targeted monitoring and modelling.	Cull sub-adults. Frequent site re-visitation in and outside of initiation zone. Improved techniques to detect and monitor juvenile CoTS.	High-resolution monitoring techniques. Real-time models of population status.	Role of ENSO, coral cover, release of predation pressure and larval nutrition. Identify settlement sites/habitats
Early outbreak stage	Suppress population at source reefs. Proactively prepare containment at sink reefs.	Targeted CoTS monitoring. Identify key source and sink reefs.	Focused and coordinated cull campaigns. Alarm signals, attractants to increase cull efficiency.	Dynamic population models. Strategies for spatial and temporal allocation of control efforts.	Dispersal dynamics in connectivity models. Role of coral cover in spread dynamics.
Late outbreak stage	Protect priority reefs. Promote early collapse.	Targeted CoTS monitoring. Identify key source and sink reefs.	Focused and coordinated cull campaigns.	Alarm signals, attractants to increase cull efficiency. Low-risk agents to promote disease in <i>Acanthaster sp.</i>	Risk analyses associated with different biological control options.

## 9. Key Partnership Achievements

This project has delivered achievements on a number of fronts: science advancements, career advancements, improved institutional capacity and new linkages between science and environmental management organisations. A key project achievement is new and clear policy advice around an environmental issue that threatens the Great Barrier Reef (GBR): Crown-of-Thorns Starfish (CoTS). By providing clarity around the efficacy and cost efficiency of a suite of management options, our project can guide investment strategies to protect a reef that supports over 64,000 jobs and contributed \$6.4 billion to the Australian economy in 2015 - 16 (Deloitte Access Economics 2017).

### 9.1 Science Advancements

The project has brought together a large multidisciplinary team of biologists, mathematical and statistical modellers, oceanographers, GIS specialists, economists and decision scientists. The team produced a strong scientific basis for exploring solutions to the CoTS problem.

The team developed analyses to help GBR managers assess the capacity for changed agricultural land-use and manual control efforts to prevent or arrest CoTS outbreak and at what costs and levels of effectiveness. Specific Partnership achievements were:

- New network model of CoTS connectivity between GBR reefs (Dr Karlo Hock). Model identifies gateway reefs for CoTS dispersal from initiation zone to all of the GBR. The model is now a decision support tool for GBRMPA to plan and guide CoTS control campaigns.
- New high-resolution model of how CoTS populations start and grow as an outbreak, how CoTS interact with corals and potentially when the outbreak collapses (Dr Carla Ewells).
- First CoTS detectability model to inform how many CoTS of different sizes are likely missed by monitoring and control campaigns (Dr Aaron MacNeil). Critical information to understand sources of uncertainty around CoTS surveillance and management effectiveness.
- Together, Hock's, Ewell's and MacNeil's models provide new insight into the capacity for CoTS control programs to contain or suppress CoTS at the early outbreak stage, including the importance of structured search and cull strategies and the importance of repeat culls.
- Economic study of the costs and benefits of different land-use options for GBR catchments as a means to prevent future CoTS outbreaks. The study is the first to directly compare the pay-offs of investments in land-use change and control campaigns as options for CoTS management and policy.
- Improved quantitative, biological under-standing of the role of water quality in driving larval survival and settlement of CoTS larvae (Prof Morgan Pratchett).
- Improved understanding of the linkages between land-use change and the risk of CoTS outbreak initiation via improved satellite oceanography and models linking nutrient loads to chlorophyll.
- Decision support framework for early CoTS outbreak prevention and control, building on key principles from fire and pest management strategies. The framework guides how science and management can be integrated adaptively to close knowledge gaps and enhance CoTS management effectiveness.

## 9.2 Clear Policy Advice on a Complex Priority Issue

CoTS is a wicked problem and progress towards solutions has been hampered by uncertainty around the multitude of processes implicated in driving outbreaks. This project has synthesised the current state of the knowledge, and has filled key knowledge gaps critical for guiding management and policy decisions.

Three key take-home messages from this project are:

- Arresting or containing outbreaks in the initiation zone is very difficult using conventional control methods once a full primary outbreak is underway.
- The mechanisms preventing such containment are:
  - Very low detectability of juvenile CoTS, which are highly cryptic within the 3-dimensional reef structure;
  - Control efforts required to suppress adult densities to below the outbreak threshold and prevent larval spread beyond the initiation zone requires near perfect detectability and multiple repeat control campaigns on individual reefs.
- Analogous to the management of wildfires, prevention of the conditions that spark and fuel an outbreak is a more cost-effective investment. One avenue here is via the reduction of nutrient run-off into the initiation zone, assuming the survival of larval CoTS is food (phytoplankton) limited.

In addition, the project has produced:

- New insight into end-of-catchment nutrient loads that represent threshold conditions for CoTS outbreak initiation.

## 9.3 Career Developments

The project promoted the careers of three promising young researchers.

- Dr Karlo Hock, postdoctoral fellow, Marine Spatial Ecology Lab, The University of Queensland
- Dr Carla Ewells, postdoctoral fellow, Australian institute of Marine Science (AIMS) and ARC Centre of Excellence for Mathematical and Statistical Frontiers.
- Dr Aaron MacNeil, early career researcher with AIMS and Dalhousie University, Canada

Also, the project supported the work of postdocs and PhDs working with Prof Morgan Pratchett, ARC Centre of Excellence for Coral Reef Studies.

## 9.4 Enhanced Institutional Capacity

The project has strengthened the capacity of particularly AIMS, UQ and TropWater to provide science-based support for decisions around the highly complex issue of CoTS management and

policy. The Partnership has developed a deeper quantitative understanding of risk and uncertainty across ecological, economic and social dimensions of the CoTS issue. This has built a transparent basis for policy advice and consideration of investment strategies to reduce the environmental and economic risks of CoTS outbreaks. The collaboration of the science teams with GBRMPA, QPWS, RRRC, AMPTO and Terrain NRM has laid the foundation for an integrated science and management strategy that can materially limit the damage caused by CoTS in the future. Lastly, the collective capacity built under this Partnership has positioned the team to better assist in the mission of supporting Reef 2050 Plan and RIMReP.

## 9.5 Publications and Reports Arising from this Project

1. **Anthony KRN.** 2016. Coral reefs under climate change and ocean acidification - challenges and opportunities for management and policy. *Annual Review of Environment and Resources* 41:59–81.
2. **Anthony KRN, MacNeil MA, Hall MR, Høj L, Schaffelke B, Uthicke S and Walshe T.** In review. The Crown-of-Thorns Starfish challenge — research to support evidence-based management on the Great Barrier Reef. *Journal of Applied Ecology*
3. **Chen C et al.** In press. Understanding ecological processes underpinning population dynamics through integration of complex individual agent based model and statistical parameter estimation. *Ecological Modelling*
4. **MacNeil MA, Mellin C, Pratchett M, Sweatman H, Cheal A, Miller I, Hoey J, Fonnesbeck C, Cowan Z-L and Anthony KRN.** 2016. Joint estimation of Crown-of-Thorns (*Acanthaster planci*) densities on the Great Barrier Reef. *PeerJ* 4: e2310
5. **MacNeil, M., K. Chong-Seng, D. Pratchett, C. Thompson, V. Messmer, and M. Pratchett.** 2017. Age and Growth of An Outbreking *Acanthaster cf. solaris* Population within the Great Barrier Reef. *Diversity* 9:18; doi:10.3390/d9010018
6. **Hock K & Mumby PJ.** In prep. Connectivity of Crown-of-Thorns Starfish in the Lizard Island-Cooktown-Cairns region of the Great Barrier Reef, with implications for management and targeting of survey and control efforts.
7. **Pratchett ,M.S., Dworjanyn, S., Mos, B., Caballes, C.F., Thompson , C.A. and Blowes, S.** 2017. Larval survivorship and settlement of crown-of-thorns starfish (*Acanthaster cf. solaris*) at varying chlorophyll concentrations. *Diversity*. 9, 2; doi:10.3390/d9010002

### Accelerate Partnership manuscripts currently in preparation

8. **Chen CM, K Anthony, K Hock and T Walshe.** Informed surveillance design improves the detection of outbreaking pests
9. **Chen CM, K Anthony, K Hock and T Walshe.** Optimised surveillance and cull design enhances control effectiveness of crown-of-thorns starfish on the Great Barrier Reef.
10. The pay-off of change in land use and management practice to reduce the incidence of primary outbreaks of Crown of Thorns Starfish on the Great Barrier Reef
11. Surveillance and cull of Crown of Thorns Starfish - a synthesis model for decision support

12. The implications of imperfect detectability for surveillance and cull operations aimed at reducing the spread of primary outbreaks of Crown of Thorns Starfish on the Great Barrier Reef
13. Informing policy on management of future outbreaks of Crown of Thorns Starfish on the Great Barrier Reef



*Acanthaster cf solaris*. Photo credit:  
Christian Miller

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