Resolving the nature and variability of the cost-effectiveness of tourism-led coral replanting on the Great Barrier Reef

by Rachael Isabella Scott

Thesis submitted in fulfilment of the requirements for the degree of:

Master of Science (Research)

Under the supervision of Dr. Emma Camp and Professor David Suggett.

University of Technology Sydney
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Certificate of original authorship

I**, Rachael Isabella Scott** declare that this thesis is submitted in fulfilment of the requirements for the award of Master of Science (Research), in the Faculty of Science at the University of Technology Sydney.

This thesis is wholly my own work unless otherwise referenced or acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

This document has not been submitted for qualifications at any other academic institution.

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Thesis acknowledgements

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Thesis abstract

Coral restoration approaches, including the asexual propagation and outplanting of corals, have gained global traction to mitigate declines in coral health and assist ecosystem recovery, including on Australia's Great Barrier Reef (GBR). While researchers and restoration practitioners have documented the biological feasibility and ecological outcomes of restoration initiatives, project costs remain rarely – and/or inconsistently – reported. This lack of cost data challenges effective project budgeting and broader understanding of the economic feasibility, context-specific suitability, and scalability of reef restoration projects, thereby hindering effective management and investment decisions critically contingent on cost-effectiveness. Cost-effectiveness data are particularly lacking for restoration practices on the GBR, where both reef restoration research and implementation has accelerated in the last five years, largely in collaboration with reef stakeholders such as the reef tourism industry. Therefore, to address uncertainties around restoration costs on the world's largest continuous reef system, the goal of this thesis was to identify and understand the cost-effectiveness of tourism industry-driven coral propagation and outplanting efforts across diverse high-value sites and modes of tourism operations on the GBR.

Goals of this thesis were examined through the Coral Nurture Program (CNP) to understand coral restoration cost-effectiveness across diverse environmental and tourism operational contexts. The CNP is a collaborative tourism industry-research partnership aimed at assisting the recovery of high-value tourism reef sites using low-cost techniques and workflows and approaches underpinned by research. After four years of scaled operations in the Cairns and Port Douglas tourism hub, the CNP was adopted in the Whitsundays region, enabling evaluation of restoration costs at different phases of program establishment and in different reef environments. I first evaluated the 'operational' costs of coral outplanting and propagation by five tourism operators through the CNP on the northern GBR (Cairns-Port Douglas), relative to outplant survivorship ('realised' costs). I opportunistically accessed existing data captured over the previous three years (required for permit reporting purposes) to derive 'operational' program costs. In parallel, I evaluated outplant survivorship at sites over space and time via a novel metal-detector based survey approach for locating established coral outplants that had been attached to reef substates using a metal device (Coralclip®). From these data, I demonstrated that costs for outplanting corals ranged from US\$0.81 - \$5.74 coral^{-1} trip^{-1}, but that costs increased 2-to-7-fold (mean US\$5.79 - \$16.33) when

essential non-outplanting restoration activities (e.g., training, nursery maintenance) were additionally conducted and costed. I further demonstrated that costs increased by 25-70% (mean US$2.94 – $21.23) when outplant survivorship (ranging on average 68-88%) across reefs was accounted for. As such, whilst opportunistic reporting of restoration activity can be utilised to examine project costs, I highlight that adopting a more comprehensive, 'whole life' restoration costing framework is essential to support project budgeting and investment decisions.

I next examined early-phase restoration implementation and associated costs of CNP activity via three tourism operators at three inshore reefs of the Whitsundays (CNPW). I evaluated activity costs over 18 months from program planning to restoration initiation and monitoring, including 'in-kind' costs that are often unquantified in restoration cost reports. I further evaluated the initial nine-month survivorship of coral outplants across the three sites via triplicate fate-tracked plots. From these data, I demonstrated total costs for CNPW implementation of $57.36 coral^{-1} ($10.63 coral^{-1} for outplanting-only activity), but that total cost estimates decreased to $32.44 coral^{-1} (-44%) when in-kind costs associated with program overheads (the largest cost category) were excluded. I further demonstrated variable outplant survivorship across sites (ranging 23-48%, 267 days post outplanting), which resulted in increased 'realised cost' estimates ranging $33 - $180 coral^{-1} depending upon site, survivorship assessment approach and inclusion of in-kind costs. My findings therefore demonstrate the importance of accounting for program overheads and in-kind contributions in project cost-assessments to avoid inflating cost-efficiency estimates. My results further demonstrate that 'effectiveness' (as assessed as coral outplant survivorship) is highly site-specific, and that implementation of existing restoration practices in new locations requires adaptation to novel ecological and operational contexts that may carry additional cost implications.

Collectively, my findings illustrate the complexities of extrapolating coral restoration costs and outcomes across different contexts, underscoring the importance of reporting primary cost data for restoration activity. I have delivered the first multi-site cost assessment of stakeholder-led coral propagation and outplanting practices on the Great Barrier Reef, identifying inherent context- and method-dependent variability in restoration cost-effectiveness; for example, lower outplant survivorship in inshore reef environments, and higher restoration costs where activity cannot be integrated within routine tourism operations.

In doing so, I have highlighted key environmental and operational factors – such as vessel size, moorings, baseline reef site condition, proximity to shore - that interact to influence costs and cost-effectiveness of CNP-type activity. I propose recommendations to guide future reef restoration research, practice, and stakeholder-funder communications to improve sufficient resourcing, transparent cost-reporting and move towards evaluation of costs relative to restoration benefits (e.g., ecosystem services). My work has demonstrated different approaches with which project cost evaluations can be conducted, highlighting the need to consider the cost-implications of specific socio-ecological contexts in restoration planning.

Thesis Structure

This thesis is comprised of an introductory chapter (**Chapter 1**), two data chapters (**Chapters 2 and 3**) in the form of journal manuscripts for peer-review, and a synthesis chapter (**Chapter 4**). At the time of this thesis submission, one data chapter (**Chapter 2**) was under review, and the other data was in preparation for submission (**Chapter 3**).

Chapter 1:

General introduction of background literature.

Chapter 2: An amended version of this chapter has been published in *Restoration Ecology* since submission:

Scott, R.I., Edmondson, J., Camp, E.F., Agius, T., Coulthard, P., Edmondson, J., Edmondson, S., Hosp, R., Howlett, L., Roper, C. and Suggett, D.J. Cost-effectiveness of tourism-led coral outplanting at scale on the northern Great Barrier Reef. *Restoration Ecology, e14137.* https://doi.org/10.1111/rec.14137

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Chapter 3: This chapter is presented as a full article prepared for journal submission.

Scott, R.I., Suggett, D.J., Hayward, C., Edmondson, J., Gillette, G., Howlett, L., Roper, C., Strudwick, P. and Camp, E.F. Early-stage outcomes and cost-effectiveness of implementing tourism-led coral propagation and outplanting in the Whitsundays (Great Barrier Reef) *

Chapter 4: General discussion, synthesis of results from both data chapters and recommendations for future research and/or practice.

*Please see title page for Chapter 2 and 3 for signed authorship declarations.

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Chapter 1: General introduction and thesis outline

1.1 Coral reef management under anthropogenic pressures.

Tropical coral reefs globally host the greatest biodiversity of marine organisms on Earth but are undergoing unprecedented decline under a suite of intensifying anthropogenic threats (Hughes et al., 2017; Souter et al., 2020; Eddy et al., 2021; Bozec et al., 2022). At local scales, accelerating development of coastlines and their watersheds has elevated nutrient pollution and sediment loads to reef systems (Burke et al., 2011; Andrello et al., 2022), compromising key reef ecosystem functions (e.g., reduced coral fecundity, propagule quality and recruitment (Fabricius, 2005); reduced herbivory by reef fishes; (Moustaka et al., 2018)), and the tolerance of corals to other chronic stressors (Silbiger et al., 2018; MacNeil et al., 2019; Donovan et al., 2021). At global scales, climate change is rapidly altering physicochemical conditions (e.g., temperature, pH, dissolved oxygen concentration) and disturbance regimes in the ocean, exceeding evolutionary thresholds of coral stress tolerance and recovery (Van der Zande et al., 2020; Hughes et al., 2021; Johnson & Watson, 2021). For example, warming tropical waters strengthen cyclones that decimate complex reef architecture into unstable rubble fields with limited potential for juvenile corals to settle and survive (Cheal et al., 2017). More frequent and intense marine thermal anomalies have particularly resulted in severe recurrent mass coral bleaching events (Eakin et al., 2019; Hughes et al., 2021; van Woesik et al., 2022), where prolonged metabolic stress can impair coral growth, reproduction, and recruitment (Richmond et al., 2018; Suggett & Smith, 2020) and eventually result in widespread coral mortality. For example, recurrent bleaching events in 2016-17 on Australia's Great Barrier Reef (GBR) resulted in 30% loss of corals across the 2,300 km reef system (Hughes et al., 2017) with subsequent declines of 50-90% in systemic larval supply (Cheung et al., 2021) and recruitment (Hughes et al., 2019) compared to the previous two decades. Such compromised ecological recovery of depleted coral populations (Ortiz et al., 2018; Hughes et al., 2019) risks irreversibly transitioning reef assemblages to highly altered states with diminished three-dimensional habitat complexity, productivity, and ecological diversity (Graham & Nash, 2013; Bellwood et al., 2019; Hughes et al., 2017).

The scale and severity of global coral reef degradation poses cascading and potentially devastating consequences for coastal communities that depend upon their many ecosystem services (Costanza et al., 2014; Woodhead et al., 2019). With close to one billion people living within a 100 km radius (Sing Wong et al., 2022), coral reefs are inherently socio-ecological systems (Cinner et al., 2009; Kittinger et al., 2012; Mcleod et al., 2019) whereby the health of reef ecosystems and prosperity of adjacent social systems and economies are profoundly interconnected (e.g., Cinner et al., 2012; Marshall et al., 2017; Reuter et al., 2022).The biodiversity and structural integrity of coral reefs underpins their estimated US$9 trillion total annual value to human society as a source of food, livelihood, biological resources, cultural heritage, identity, recreation, and coastal protection (de Groot et al., 2012; Costanza et al., 2014; O'Mahoney et al., 2017). Globally coral reefs support the livelihoods of an estimated six million small-scale fishers (Teh et al., 2013), who in turn, sustain the food security of almost four million people in developing reef-adjacent communities (Donner & Potere, 2007; Selig et al., 2019). Living structures of coral reefs provide habitat and breeding grounds for over a third of *named* global marine species (Fisher et al., 2015), and are hence foundational to the growing blue economy (e.g., Grafeld et al., 2017; Cziesielski et al., 2021; Fairoz, 2022). Reef-associated tourism alone is valued at nearly $US36 billion/year, and in many small-island jurisdictions, contributes over 10% to national Gross Domestic Product (Spalding et al., 2017). The capacity of coral reefs to sustain these ecosystem service values is diminishing under intense anthropogenic pressures (Eddy et al., 2021). Consequently, 'reactive' measures (Hein et al., 2021) are needed to sustain their value and equip reef-dependent communities with additional ways to adapt and respond to changing reef systems (Mcleod et al., 2019; Saunders et al., 2020; Hein et al., 2021; Kleypas et al., 2021).

Prior to pan-global bleaching events, coral reef management largely centred on mitigation of localised stressors through pollution mitigation via catchment management (Beher et al., 2016; Kroon & Brodie, 2009) and the enforcement of marine protected areas to counter overfishing practices (Strain et al., 2019). However, such conventional approaches, whilst essential to prevent synergistic stress on corals (Suggett & Smith, 2020; Andrello et al., 2022), are insufficient to safeguard even the most highly protected and pristine reefs on the planet against the pervasive impacts of climate change (Hughes et al., 2021; Johnson & Watson, 2021). Furthermore, whilst the future survival of functioning coral reefs hinges upon rapid decarbonisation of global economies (Kleypas et al., 2021), ongoing coral population losses are projected irrespective of climate action, due to ocean warming already ‘locked in’ by greenhouse gas emissions to date (van Hooidonk et al., 2016; Beyer et al., 2018; McWhorter et al., 2022). These stark realities of a changing ocean have shifted reef governance, management and research strategies (Anthony et al., 2017; Bellwood et al., 2019; Suggett et al., 2023) toward consensus that reactive ‘restoration and adaptation’ interventions will be instrumental to supporting the persistence of key reef ecosystem functions and services until global temperatures stabilise under a lower carbon economy (Duarte et al., 2020; International Coral Reef Initiative, 2021; Knowlton et al., 2021). Faced with omnipresent threats beyond their control (Gibbs & Newlands, 2022; Knoester et al., 2023), reef management authorities and local reef communities are thus implementing more reactive management strategies to locally assist recovery of depleted coral populations and strengthen reef resilience (Rinkevich, 2019; Shaver et al., 2022). For example, on Australia’s GBR, repeat mass bleaching events triggered the marine park’s central management authority to adopt policies aimed at incorporating reactive restoration-based intervention into existing mitigation and monitoring-based management (Great Barrier Reef Marine Park Authority (GBRMPA), 2017, 2020).

1.2. Global proliferation of reef restoration

The next two decades represent a time-critical window to trial and implement reactive reef restoration and adaptation interventions *before* ecosystems approach irreversible phase-shifts or tipping points (Anthony et al., 2017; Hardisty et al., 2019; Knowlton et al. 2021). In recognition of this urgency and the centrality of marine ecosystem services to human wellbeing, the United Nations (UN) General Assembly declared 2021–2030 the ‘UN Decade of Ecosystem Restoration’ alongside the ‘UN Decade of Ocean Science for Sustainable Development’ (Knowlton et al., 2021). The coupling of these decadal priorities highlights the need to not only rapidly scale marine restoration efforts but ensure that approaches are guided by science to upscale effectively (Waltham et al., 2020; Vardi et al., 2021; Shaver et al., 2022). Furthermore, during my thesis candidature in December 2022, the landmark Kunming-Montreal biodiversity framework was adopted at COP17 (Conference of Parties to the Convention on Biological Diversity), establishing targets that by 2030, 30% of all degraded (marine and terrestrial) ecosystems will be under effective restoration and US$200B/year will be mobilised to finance biodiversity conservation (Convention on Biological Diversity, 2022; Suggett et al., 2023). Concurrently, strategies to map and quantify ecosystem service flows to guide investment and mobilise funding have recently been proposed and/or developed through the creation of national ecosystem service accounts (or "natural capital" accounts) (e.g., Cziesielski et al., 2021; Australian Bureau of Statistics, 2022; Global Ocean Accounts Secretariat, 2023; The White House, 2023).

Such global commitments have strengthened growing investment (US$258 million over the last 15 years) into reef restoration research and implementation (Hein & Staub, 2021; Victurine et al., 2022). In recent years, large-scale national (e.g., Australia's Reef Restoration and Adaptation Program ("RRAP"; McLeod et al., 2022) and international (e.g., G20 Coral Research and Development Accelerator Program (CORDAP)) research and development programs have been funded to expedite development of effective techniques at larger scales (Hein & Staub, 2021; Suggett et al., 2023). Moreover, novel financing mechanisms have emerged (e.g., parametric insurance schemes (Schelske et al., 2021) and blended-finance instruments (e.g., Global Fund for Coral Reefs (Meyers et al., 2021)) which are critically expanding capacity for sustained restoration beyond short-term, small-scale research projects (summarised in Suggett et al., 2023). Despite increases in funding, the growing scale of reef degradation means investment prioritisation is necessary (Anthony et al., 2020). However, effective, and informed investment remains fundamentally challenged by persistent uncertainties about the cost and feasibility of reef restoration interventions (Bayraktarov et al., 2015; Hein & Staub, 2021).

Marine restoration is in its infancy in comparison to more well-established terrestrial practices (Bayraktarov et al., 2015; Rinkevich, 2019**)**, with coral restoration only gaining momentum over the last 40 years (Omori, 2019; Saunders et al., 2020). During this time frame, adoption has accelerated to over 56 reef-associated countries at varying scales (0.01 - 1.5 ha), with goals of repopulating coral assemblages or recovering reef structure and habitat (Bayraktarov et al., 2019; Boström-Einarsson et al., 2020).The most widely employed approach has been via asexual coral propagation and 'outplanting' (previously colloquially referred to as "coral gardening" (Rinkevich, 2006; Vardi et al., 2021), which involves transplantation of coral fragments using either naturally fragmented coral material ("Corals of Opportunity") or material propagated in *in-situ* or *ex-situ* nurseries (Rinkevich, 2006). Other approaches aim to enhance coral larval recruitment success through macroalgal removal (Kittinger et al., 2016; Smith et al., 2021) and stabilisation of coral rubble substrate (e.g., Williams et al., 2019; Ceccarelli et al., 2020). Despite several decades of practice in key reef regions worldwide (e.g., the Caribbean; Lirman & Schopmeyer, 2016; Coral Triangle; Razak et al., 2022) such techniques have only been implemented in the last five years in Australia, with activity on the GBR now spanning ~20 in-water projects and the $100 million RRAP program to test, improve and scale up reef restoration methods (reviewed in McLeod et al., 2022).

Global focus for coral restoration has more recently shifted to scaling coral restoration approaches and enhancing genetic diversity of restored populations via sexual propagation. Here, coral gametes are collected during spawning events to either be grown up in ex-situ aquaria for later outplanting (e.g., Villanueva et al., 2012; Guest et al., 2014), deployment on "seeding" devices (Chamberland et al., 2017; Randall et al., 2021, 2023), or contained in net enclosures for several days until they reach competency, and are settled on fixed areas of the reef (i.e., "larval enhancement") (dela Cruz & Harrison, 2017; Harrison et al., 2021). As natural reef recovery capacity becomes increasingly impaired, the reef restoration 'toolbox' (*sensu* Rinkevich, 2019) is necessarily expanding to encompass innovative techniques aimed at enhancing the adaptive potential of the coral 'holobiont' to repeat and chronic stressors (reviewed in Voolstra et al., 2021; Suggett & Van Oppen, 2022). However, whilst many of these emerging techniques hold promise for enhancing coral stress resistance (e.g., Buerger et al., 2020), few are validated beyond small-scale laboratory or field trials (Kleypas et al., 2021). Thus, different restoration interventions are at varying stages of maturity as feasible, 'shovel-ready' management tools for responding to current reef decline (Hein et al., 2021; Suggett & Van Oppen, 2022).

Widespread recognition that reef ecosystems are already fundamentally in transition has shifted the goalposts of coral reef restoration (Rinkevich et al., 2019; Sheaves et al., 2021; Kleypas et al., 2021; Voolstra et al., 2021), from restoring to pre-disturbance baseline states to more realistic goals of conserving priority ecosystem values, functions and services (Bellwood et al., 2019; Anthony et al., 2020; Saunders et al., 2020; Hein et al., 2021). Different restoration and adaptation interventions aim to address different goals (Hein et al., 2021), which range from rebuilding populations of specific species (Ware et al., 2020), improving structural complexity of reef habitat (Fadli et al., 2012; Williams et al., 2019), enhancing coral thermal tolerance (Humanes et al., 2021), or maintaining ecosystem service delivery, such as a reef site tourism or storm protection value (Roelvink et al., 2021; Howlett et al., 2022). Thus, no single approach is suitable or feasible for all reef sites (Anthony et al., 2020; Hein et al., 2021). Recent efforts are helping to inform the 'complex decision challenge' (*sensu* Anthony et al., 2020) inherent in prioritising projects, sites, and methods, to effectively deliver restoration interventions (Fig. 1.1). These include the development of coral species-selection frameworks (e.g., Baums et al., 2019; Madin et al., 2023), spatial prioritisation frameworks based on ecological models of reef larval connectivity and identified thermal refugia (Beyer et al., 2018; Doropoulos & Babcock, 2018; Selmoni et al., 2020; Camp, 2022; Quigley & van Oppen, 2022) and socio-spatial maps of human uses and ecosystem service values (Levine & Feinholz, 2015; Spalding et al., 2016; Storlazzi et al., 2019). Nevertheless, meeting reef restoration goals at ecologically or socially meaningful scales means that future project feasibility, prioritisation and investment allocation are underpinned by the need to justify the cost-effectiveness of any given approach (Okubo & Onuma, 2015; Omori, 2019; Anthony et al. 2020; Suggett et al. 2023). However, at present, though they represent critical decision support tools for restoration planning, cost data on coral restoration interventions is fundamentally lacking (Fig. 1.1A-B).

1.3 Money Matters: costs and cost-effectiveness of coral restoration.

Reef restoration projects have collectively documented and shared learnings over the past decade via published studies and reviews, dedicated restoration guides, and consortiums (e.g., Shaver et al., 2020; Vardi et al., 2021). However, socioeconomic evaluations of project costs needed to gauge and/or justify cost-effectiveness remain sparsely and inconsistently reported (Spurgeon & Lindahl, 2000; Edwards et al., 2010; Iacona et al., 2018; Bayraktarov et al., 2019) (see also Table 1.1). In a review of 87 coral reef restoration studies, Bayraktarov et al. (2019) found that <30% of projects reported cost estimates, and of these, only 28% reported on different cost components (Fig. 1.1B). Fewer still reported costs across all phases of restoration projects, spanning planning, training, nursery installation and maintenance, coral deployment, and post-deployment monitoring (e.g., 'whole life costs'; *sensu* Spurgeon, 2001). Furthermore, 'in-kind' contributions of researcher/volunteer time or resources, upon which the financial viability of many reef restoration programs depends (e.g., Hein et al., 2018), are rarely quantified (Edwards et al., 2010). As such, where costs have been reported, they are likely an underestimate of real total project costs or 'true costs' (Iftekhar et al., 2017; Bayraktarov et al., 2019; Hein & Staub, 2021).

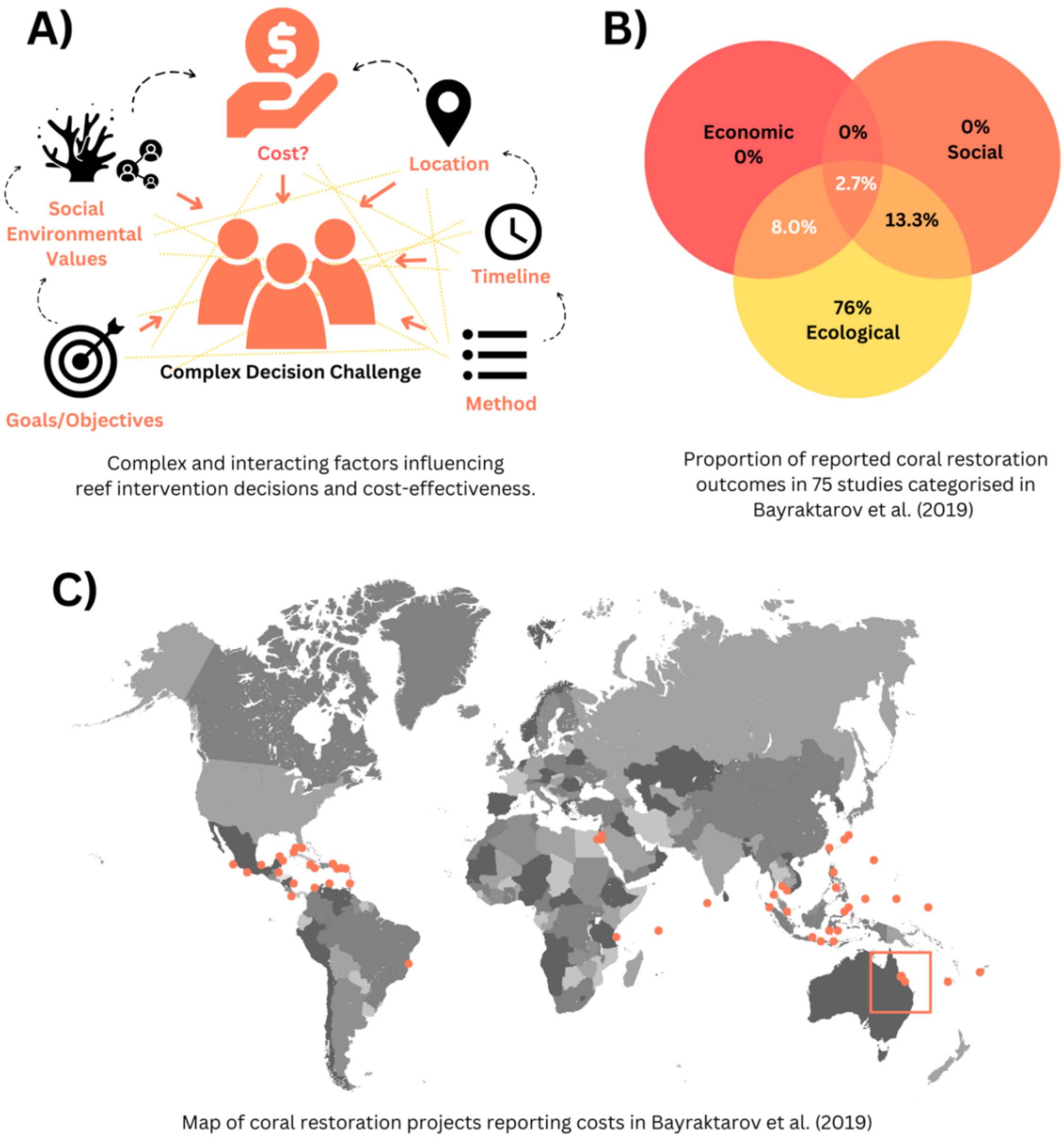

Figure 1.1. (A) Decisions facing coral reef managers regarding coral restoration are underpinned by complex and interacting factors, and questions remain around effective siting, timing and method-selection tailored for particular social and ecological contexts and restoration objectives (Anthony et al., 2020). The mismatch between the scale of intervention required and the availability of resources to effectively address restoration goals means investment prioritisation is necessary. However, **(B)** Decisions are challenged by uncertainty around restoration costs, as <10% of restoration projects (n = 75) (as reviewed in Bayraktarov et al., 2019) have reported on socioeconomic outcomes. **(C)** Of these, only one project was located on Australia's Great Barrier Reef (reported in Spurgeon & Lindahl, 2000). Since then, only one other study (Suggett et al., 2020) has reported coral restoration costs for the region. Figure 1.1 (B) and (C) adapted from Bayraktarov et al., 2019.

Such gaps and inconsistencies in reporting primary cost data can hamper the ability of managers, funders, and practitioners to understand the cost-benefits and trade-offs of different restoration approaches (Eger et al., 2022) or the cost implications of different restoration actions, contexts, and outcomes (Edwards et al., 2010). This in turn challenges informed investment and budgeting decisions to ensure restoration goals can be met (Edwards et al., 2010; Hein & Staub, 2021; Suggett et al., 2023). Given that project costs underpin feasibility and hence investment confidence, reporting this data are thus now considered a conservation priority (Hein & Staub, 2021; Iacona et al., 2018; Eger et al., 2022; Suggett et al., 2023). Notably, due to the relative novelty of reef restoration on Australia's GBR, costing data are especially underrepresented on the world's largest continuous reef system (e.g., Fig. 1.1C, Table 1.1; see also Bayraktarov et al., 2015, 2019; Hughes et al., 2023).

Whilst understanding direct costs is important for formulating restoration budgets, benchmarking costs against outcomes is essential to demonstrate what project objectives (e.g., key outcomes, benefits, and goals) can be achieved within available resources (i.e., 'cost-effectiveness' or 'cost-benefit') (Cook et al., 2017; Anthony et al., 2020; Hein et al., 2021). This in turn ensures that 'success' or justification for investment is not simply measured by the scale of deployment (e.g., number of fragments planted) (Suggett et al., 2023; Hein & Staub, 2021). In the context of this thesis, I specifically examine 'cost-effectiveness' and define key terms as follows (Fig. 1.2): The cost of an intervention includes the capital and operational expenditure required to deliver it (Spurgeon, 2001; Bayraktarov et al., 2015; Iacona et al., 2018), and this overall cost is often expressed relative to the scale of activity or output (e.g., "cost-efficiency"; cost per coral outplanted) (Fig. 2). Whereas cost-effectiveness compares the cost of an intervention (in monetary units) relative to an outcome (in non-monetary units) that is related to program objectives (Hughey et al., 2003; Cook et al., 2017). In both terrestrial and marine restoration, outcomes have typically been related to growth and survival of propagules (e.g., Hein et al., 2017; Ferreira et al., 2022), however also include change in cover of restored populations (Kimball et al., 2015; Mostrales et al., 2022). Cost-benefit analyses compare intervention costs with outcomes and benefits that have been assigned monetary values through natural capital accounting or ecosystem service valuation (Spurgeon, 2001; Hughey et al., 2003). These different methodologies exist within a continuum of increasing difficulty in accounting (requiring increasing data depth), but equally, increasing utility to managers and investors as decision support-tools (Iftekhar et al., 2017; Fig. 2). Assessments of cost-effectiveness and cost-benefit can help to determine where and when different approaches are feasible and beneficial, and ultimately whether they should be initiated, continued, or terminated (McLeod et al., 2019).

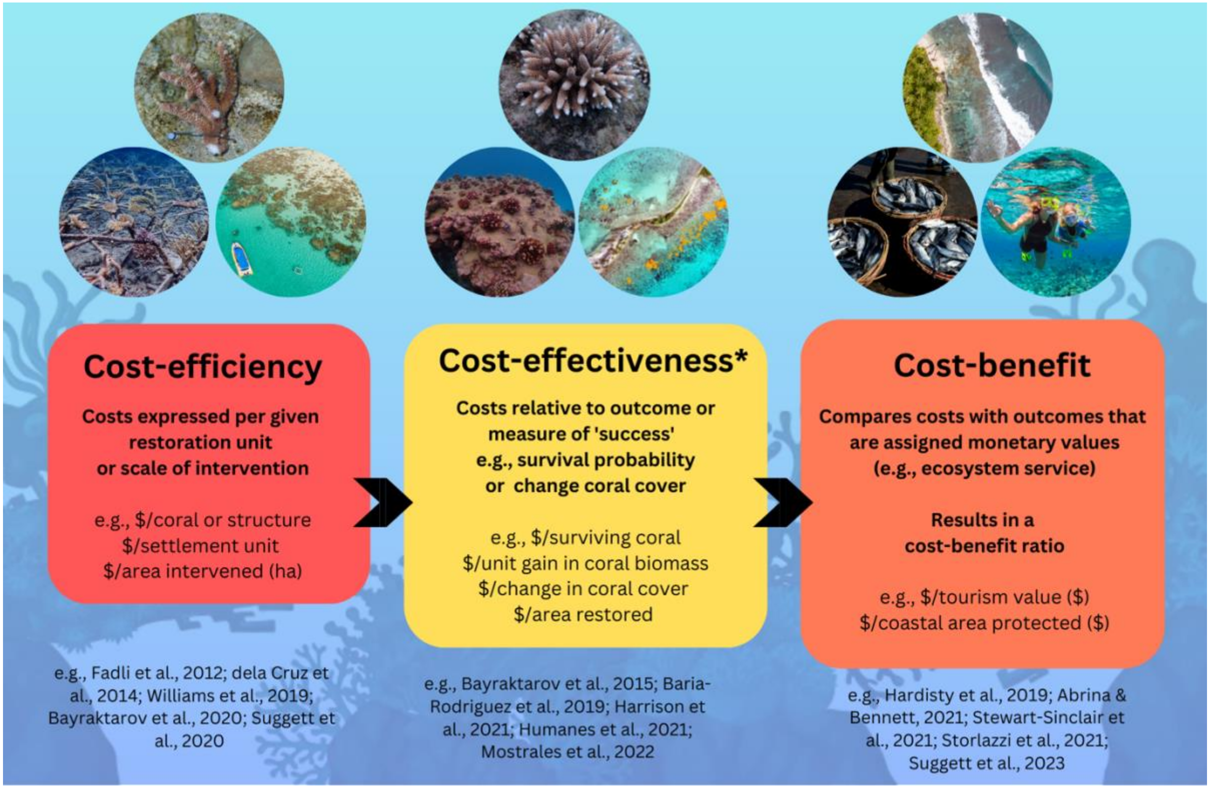

Figure 1.2. Key terms as defined in this thesis that are used in economic evaluations of interventions to aid resource allocation decisions in fields spanning conservation to healthcare. Examples from the coral restoration literature are given, where the type of cost-evaluation reported is categorised accordingly below each term. *Indicates the economic evaluation specifically addressed in this thesis. Arrows represent the increasing difficulty in evaluation, owing to increasing data depth and valuation of non-market values required in cost-benefit analyses.

Just as cost accounting across projects is influenced by restoration context and costing approach (Bayraktarov et al., 2019; see also Table 1.1), approaches for assessing cost-effectiveness are similarly diverse: Table 1.1 outlines several variable approaches for cost accounting and evaluating cost-effectiveness across methods and locations. For example, some projects have evaluated restoration costs based upon specific stages of intervention (e.g., outplanting or deployment only; Chamberland et al., 2017; Suggett et al., 2020), whereas others have undertaken comprehensive life-cycle cost analyses of all phases (e.g., Villanueva et al. 2012; projects reported in Bayraktarov et al., 2020; Humanes et al., 2021). Some projects report the costs of intervention only (e.g., Mbije et al., 2013; Cruz et al., 2014;

Williams et al., 2019), whereas others benchmark costs based upon defined outcomes, such as survivorship or gain in coral biomass, over specific timeframes to evaluate cost-effectiveness or 'realised' costs (costs less losses) (Chamberland et al., 2017; Toh et al., 2017; Baria-Rodriguez et al., 2019; Forrester et al., 2019; Harrison et al., 2021; Humanes et al., 2021). Other studies evaluate cost-benefit by relating costs to the retention of (or gains in) ecosystem service value. Approaches include social-ecological systems (SES) frameworks (e.g., tourism value, Suggett et al., 2023), contingent valuation (Abrina & Bennett, 2021) or value transfer methods (Stewart-Sinclair et al., 2021) to assign economic values to non-market values (e.g., coral cover, fish abundance and diversity; Abrina & Bennett, 2021). Economic evaluations are thus highly dependent on the metrics, scales, and timeframes over which 'effectiveness' is evaluated. Furthermore, owing to cost and scale bottlenecks associated with outplanting capacity (Edwards et al., 2010), few projects have operated at a scale large enough, or over sufficient timeframes to meaningfully evaluate cost-effectiveness.

Table 1.1: The diversity of approaches for estimating (a) cost-efficiency and (b) cost-effectiveness/cost-benefit (see Fig. 1.2) for coral restoration coral from across 18 studies. Note, this list is not exhaustive, and more extensive reviews have been conducted in Bayraktarov et al. 2016; 2019 and Hughes et al (2023). Data was extracted either directly from the study supplementary material or where possible, summarised values in Bayraktarov et al (2019). All costs reported in $USD. Inclusions detail the cost attributes that were accounted for in reported costs, whereas 'Exclusions' detail the key cost attributes that were not included in reported costs.

A) Project costs or cost-efficiency reported only							
Study/Project	**Location**	**Costing Method**	**Timeframe**	**Restoration Method**	**Working size**	**Costing approach (inclusions, exclusions, and assumptions)**	**Cost**
12 restoration projects in Latin American countries and territories reported in (Bayraktarov et al., 2020)	(Several) Colombia, Costa Rica, Dominican Republic, Mexico,	Cost efficiency	Median 3 years (one project 17 years)	Mostly asexual propagation and planting (with in situ nursery rearing), and one ex-situ larval propagation project	0.06 - 8.39 hectares	Costs are converted to 2018 USD and scaled to cost/hectare/year. <u>Inclusions:</u> installation and operation including planning, construction of nurseries, labour and boat use, equipment, SCUBA gear, monitoring and maintenance based upon cost-accounting framework in Iacona et al (2018) <u>Exclusions:</u> Specific breakdown of costs not available as these estimates were provided to the authors by correspondence.	US$10,000 - $331,802 ha/year (2018) ($US93,000 median)
Community-based, low-tech method of restoring a lost thicket of Acropora corals. (dela Cruz et al., 2014)	Santiago Island, Bolinao Anda Reef Complex, Philippines	Cost efficiency	19 months (2010-2012)	Asexual transplantation and planting in sand/rubble substrate using bamboo stakes and wire	450 fragments transplanted in an area of 96m^2	Cost estimates scaled to 1 hectare (62, 500 fragments) for scenarios with and without community volunteer participation. <u>Inclusions:</u> (a)With community participation - boat rental, labour, and snorkel gear all assigned a value of $0 (volunteers and in-kind contributions); materials for coral collection and planting; fuel (b) Without community participation: Boat rental and gasoline ($8/day for large boat,	With volunteer involvement: US$9,198.40 per ha ($0.9/m^2 or US$0.15 per coral) (2010) Without volunteer involvement: US$22,839.74 per ha (or $2.28/m^2 or $0.36/coral)

						\$121.95/day for large boat), labour (US\$13/person/day), and snorkel gear/person/day, materials for coral collection and planting. Exclusions: In-kind researcher costs, monitoring costs, exchange rates, SCUBA gear not required (planting done by snorkelling)	
The role of habitat creation in coral reef conservation: a case study from Aceh, Indonesia. (Fadli et al., 2012)	Pak Dodent artificial reef project, Palau Weh, Indonesia	Cost efficiency	3 years (2006-2009)	Artificial reef: asexually propagated coral fragments attached to concrete blocks.	260 concrete modules covering an area of 250m^2 (0.0250ha)	Inclusions: cost of materials to construct modules, labour costs for construction, transport costs (e.g boat). Exclusions: cost for fragment collection, monitoring and maintenance, currency conversion rate, year of conversion *After observing increases in recruitment, coral cover, fish diversity and abundance, and tourism activity after 3 years, a severe bleaching event resulted in almost 100% mortality of corals on the structures.	US \$45/module or US\$11,700 total (US \$2006) (US \$2010) \$437,648/ha (Bayraktarov et al., 2019)
A first endeavour in restoring denuded, post bleached reef in Tanzania. (Mbije et al., 2013)	Changuu, Zanzibar and Kitutia Reefs, Tanzania	Costs only	2 years (2008-2009)	Asexual propagation and planting of nursery-grown coral colonies. Attachment: wedging into dead coral spaces OR drilled holes filled with epoxy	14,022 corals in total 1296 m^2 area	Costs for transplantation phase only Inclusions: boat hire cost (\$80/day), volunteer labour cost (\$10/day), materials (epoxy, small tools including knives, cutters) for transplantation. Exclusions: labour, boat and dive costs for nursery installation and maintenance, post-transplant monitoring, diving costs. Unclear if labour accounts for time to collect fragments from nursery and transport to transplantation site. Unclear if cost for drill is included.	US\$2,020 total or \$0.14/colony (Year of currency conversion not stated)
Large-scale coral reef restoration could assist natural recovery in Seychelles, Indian	Cousin Island Marine Reserve, Seychelles	Costs only	2011-2014 (3.5 years)	Asexual propagation and planting of nursery-grown coral fragments.	24,431 coral colonies transplanted over	Costs are reported in Bayraktarov et al (2019) and include total cost for carrying out a complete coral reef restoration project over 3.5 years consisting of nursery, outplanting and monitoring.	US\$1,429,893/ha \$33.40/coral (overall reported project cost divided by number

Ocean (Montoya-Maya et al., 2016) Costs reported in (Bayraktarov et al., 2019)					5,225m^2 (0.52ha)	Costs account for overheads and a research & development component of ca. US$300,000. Total cost of the project is scaled to the cost to restore 1 ha (with a planting density of 4 corals/m^2) in US$(2010).	of colonies planted) OR $43.42/coral (based on 70% survival rate reported in Montoya-Maya et al. 2016
Coralclip®: a low-cost solution for rapid and targeted out-planting of coral at scale. (Suggett et al., 2020)	Great Barrier Reef (Australia)	Cost efficiency	7 months (2018-2019)	Asexual coral propagation and planting of primarily corals of opportunity	3 sites, one reef - 4,580 fragments	<u>Inclusions:</u> staff time per day (US$ 168), Coralclip® device cost. <u>Exclusions:</u> boat cost ($0, boat cost absorbed by existing tourism trips), Nursery infrastructure equipment and installation costs, planting equipment (brushes, hammer, basket) - assumed negligible across several thousand deployments. Labour costs did not include time to travel to site only time for Coralclip® deployment (e.g, cost per unit effort).	US$0.6–3.0/coral deployed A commercial scenario costing was also performed incorporating a boat cost of US$1,750/day and labour costs of US$300/day, yielding a cost-estimate 12.25x greater (i.e US$11.40)
Large-scale coral reef rehabilitation after blast fishing in Indonesia (Williams et al., 2019)	Palu Badi, Indonesia	Cost efficiency	2 years (2013-2015)	Rubble stabilisation and planting: Attaching corals of opportunity or nursery-grown fragments to MARS Reef Stars	Reef stars cover an area of 7000m^2	Estimated cost based on installation costs and some maintenance costs. <u>Inclusions:</u> materials, construction labour, truck and boat transportation to deployment site (rates given per 100 spiders, rather than day rates), coral attachment and installation labour, maintenance costs estimated at US$3 per spider. Unclear if labour includes the efforts of 36 trained islanders or only 4 divers. <u>Exclusions:</u> Currency conversion details, monitoring costs.	Cost per spider estimated at US$15.76 (2016 USD)) or US$24.85/m^2
(b) Costs benchmarked against measure of ‘effectiveness’ or ‘benefit’							

Study/Project	Location	Costing Method	Timeframe	Restoration Method	Working size	Costing approach (inclusions, exclusions, and assumptions)	Cost
A benefit-cost comparison of varying scales and methods of coral reef restoration in the Philippines. (Abrina & Bennett, 2021)	Bolinao, Philippines	Cost-benefit analysis using Choice-Modelling and Willingness to Pay methodologies	n/a	Cost-benefit analysis comparing asexual coral gardening versus sexual larval enhancement	1-65ha (hypothetical)	Compared the cost of a local-scale coral gardening project (de la Cruz et al, 2014) and a larval enhancement project (de la Cruz & Harrison, 2017) with methods employed at national scales (Filipinnovation on Coral Restoration and a hypothetical large-scale larval enhancement project). These costs were compared to an estimation of benefits based on Choice Modelling or Willingness to Pay for gains in coral cover, fish abundance and diversity by Filipinos living near reefs and in the city (Abrina & Bennett, 2018).	The benefit-cost ratio for local scale coral gardening was 37.3 versus 87.4 for local scale larval enhancement
Performance and cost effectiveness of *Acropora granulosa* juveniles compared with asexually generated coral fragments in restoring degraded reef areas. (Baria-Rodriguez et al., 2019)	Bolinao Anda reef complex, Philippines	Cost-effectiveness based upon survivorship	2 years (2010 - 2012)	Asexual propagation and sexually propagation and planting (both include an intermediate hatchery and in-situ nursery phase)	150 asexual and sexual propagules each planted to 5 bommies (4m in diameter)	Realised cost/coral surviving 13 months post out-plant. Costs calculated from time of propagule production (from collection to in-situ nursery to out planting) <u>Inclusions:</u> materials, consumables, boat rental and fuel ($94/day), hired labour (split into salaries US$10.50 - $28/day), scuba gear and air tank rental. <u>Exclusions</u>: post-out plant monitoring, attachment material cost.	<u>Asexual</u> Production cost = US$7.05/colony Realised cost (20.4 months survivorship of 4.67%) = US$88.15/colony <u>Sexual</u> Production cost = $12.7/colony Realised cost (20.4 months survivorship of 18%) = US$20.01/colony
New seeding approach reduces costs and time to outplant sexually propagated corals for reef restoration.	Curacao, Caribbean	Cost-effectiveness based upon survivorship	2014 (1 year)	Sexual propagation and planting: Seeding of sexual recruits onto tetrapods in ex-situ nurseries and planting 3-week	160 seeding units in a 0.150 m^2 area	Realised cost: Cost/seeding unit (SU) remaining with settlers surviving after 1 year for a hypothetical 10 000 SUs. <u>Inclusions:</u> materials cost for production of SUs (US$0.5), materials needed to secure SUs to reef (epoxy, nails, cable ties,	US (2010)$1-2.50 per SU remaining with settlers surviving after 1 year

(Chamberland et al., 2017)				old juveniles to reef		pneumatic drill); tanks for divers, planting labour ($6.63 per person/per hour - calculated to US dollars based on median global GDP). Exclusions: expenses for colony collection, spawning, larval rearing, nursery construction and maintenance; boat usage costs; labour costs did not include time to travel to site only time for SU deployment (e.g., cost per unit effort).	
Comparing the efficiency of nursery and direct Transplanting methods for restoring endangered corals. (Forrester et al., 2019)	Harris Ghut and Muskmelon Bay, British Virgin Islands	Cost-effectiveness based upon survivorship and biomass gain.	15 months (2013-2014)	Asexual propagation and planting: comparing direct transplantation with intermediate nursery phase	400-800 m^2	Return on investment and return on effort (time) calculated as a function of the number of surviving corals and the gain in living coral biomass after 64 weeks - calculated separately. Inclusions: For return on financial cost, only materials cost was included (e.g. nursery installation materials, coral attachment). Exclusions included: SCUBA gear, labour for all stages, boat costs and materials for monitoring. Return on time investment included all phases of project (coral collection, attachment, nursery installation, materials purchase, transfer from nursery to reef). Nursery maintenance and monitoring were not included. Labour was not expressed in $ terms.	3 living corals per $1 invested for direct transplantation versus 1 living coral per $1 invested for nursery-grown corals (as a result of greater time and materials investment required for little observed return in terms of enhanced growth/survivorship)
An ounce of prevention: cost-effectiveness of coral reef rehabilitation relative to enforcement. (Haisfield et al.,	Komodo National Park, Indonesia	Cost-effectiveness based upon predicted coral coverage.	2002-2009 (7 years)	Substrate stabilisation: artificial reef consisting of locally quarried rocks after chronic blast fishing damage, relied on natural recruitment	0.6430m^2	From 7 years of coral growth data, generated a cost-effectiveness model for predicted coral coverage gained from investment in rehabilitation versus the coral damage averted by investment in enforcement of blast fishing. Installation cost: 76 boat trips, 910 truck loads, 6430 m^2 covered area, rock cost at	To achieve 100% coral cover over 1m^2, would cost US$52.92 or it would require 11m^2 of rock installation to get 1m^2 of coral cover in the future. For enforcement,

2010)						US$ 3.11/m³, boat cost at US$ 266.67/trip, Ranger + meals (labour) at US$46.67/trip.	saving 1 $m^2$of 100% coral coverage would cost an estimated $9.64.
Increased coral larval supply enhances recruitment for coral and fish habitat restoration. (Harrison et al., 2021)	Magsaysay Marine Protected Area, Philippines	Cost-effectiveness based upon survivorship	3 years (2016 - 2018)	Larval enhancement: culture and settlement of 7-day old coral larvae in enclosed mesh nets	75 m^2 (3 x 25m^2 plots)	Realised cost reported as the cost/coral surviving after 10 and 34 months. Costs divided into colony and spat collection, larval rearing, site preparation and settlement, exchange rate. <u>Inclusions:</u> Materials (SCUBA tanks, equipment hire, larval enhancement nets, hatchery facility use), boat rental/fuel ($105.4 – 210.80/day), hired labour at all stages ($16.45 - $23.77/day). <u>Exclusions:</u> follow-up monitoring costs, in-kind researcher time (only hired labour)	US$13.73/coral for 285 sexual recruits alive at 10 months US$17.79/coral for 220 living colonies at 34 months age (77 colonies gravid).
An experimental framework for selectively breeding corals for assisted evolution. (Humanes et al., 2021)	Palau	Cost-effectiveness based upon survivorship	2017-2020 (3 years)	Sexual propagation and planting (assisted evolution): selectively bred corals raised in ex-situ nursery and outplanted after 5 or 11 months.	384 colonies at 2.5 colonies/m^2 across 16x10m transects	Costs divided into capital, consumable and labour costs for all stages including colony collection and spawning, larval culturing, rearing, out planting and monitoring. <u>Inclusions:</u> Materials (SCUBA tanks and diving equipment, all materials and consumables for all phases including an underwater drill costing $1,386 (pro-rated by number of uses/years), boat rental/fuel ($375/day plus $25 fuel/trip), labour (divided into different pay grades, $US 5.77 - $28.37/hour) at all stages. <u>Exclusions:</u> laboratory bench fees, cost of experiment to characterise heat tolerance ($9,000)	US$227 per surviving 2.5-year-old coral planted at 5 months ex-situ nursery rearing (6% survivorship) US$49 per surviving 2.5 year old coral planted at 11 months ex-situ nursery rearing (30% survivorship).
Evaluation of the performance and cost-effectiveness	Talim Bay, Luzon Island,	Cost-effectiveness based upon	Nov 2018-May 2019	Microfragmentation on artificial reef habitats	3 x (8x5m) clusters at 3 sites.	Using a model that assumed a target coral cover of 20% in 1 ha of reef (Feliciano et al., 2018), micro-	Total cost to achieve target 20% hard coral cover:

of coral microfragments in covering artificial habitats (Monstrales et al., 2022)	Philippines	survivorship and coral growth and predicted coral coverage	5-week 'ex-situ' nursery period followed by 5 months deployed at site			fragmentation performance (as measured by projected coral cover, overall cost and required number of colonies) was compared to (a) coral gardening and (b) larval enhancement values reported in other studies. Models were run using production costs, and accounting for coral growth rates and mortality. Exclusions: Costs presented by stage and cost category however personnel time (person-hours) and boat time not indicated. Details on currency conversion (year) and standardisation between methods unclear. Personnel costs do not include salaries.	Without/(with) mortality: Coral Gardening US$122,240.00 (US$188,061.54) Larval Enhancement: US$272,000.00 (US$824,424.42) Micro-fragmentation: US$210,160.00 ($US404,153.85)
Spatial cost–benefit analysis of blue restoration and factors driving net benefits globally. (Stewart-Sinclair et al., 2021)	Global (29 studies)	Cost-benefit analysis based upon published ecosystem service values	Various (converted to 2010 $US)	Several	Several - however costs were scaled to 1 hectare/year	Cost values for studies previously reported in Bayraktarov et al (2019) were converted to 2010 ($US) Benefit values (in 2010 $US/hectare/year) for respective locations were extracted from Ecosystem Service Value Database (ESVD) published in de Groot et al.. (2012). Costs were paired to benefits using a value-transfer approach	Benefit-Cost ratio for coral restoration studies was 4:1.
An integrative framework for coral reef restoration	Opal Reef, Great Barrier Reef, Australia	Cost-benefit analysis based upon reef tourism value (via SES	2021 (using 2018-2019 survivorship and	Asexual coral propagation and planting	Costs estimated for 24.7km^2 reef area (where 17% reef area	Using mean planting costs, survivorship and coral growth rates previously reported for Opal Reef in Suggett et al (2020) and Howlett et al (2022), the cost of replanting at	Aerial coral recovery cost-efficacy =71cm^2/$/yr

(Suggett et al., 2023)		framework)	costings)		is hard coral)	sufficient quantities to offset 2021 rates of coral cover decline (and thus retain the Reef's US$15 million/year tourism value as per Spalding et al (2017)) was calculated.	Cost to retain 2021 coral cover = $1.61M (10% of tourism service value) Therefore every US$1 spent on restoration retains US$10 in tourism value
A cost-effective approach to enhance the scleratinian diversity on artificial shorelines (Toh et al., 2017)	Singapore	Cost-effectiveness based upon survivorship and unit area of coral biomass	2014 - 2015 (18 months)	Asexual propagation and planting (Intermediate nursery phase) onto a seawall	213 colonies transplanted to an area of $60m^2$	Realised cost per transplanted colony alive 6 months post-transplant or cost per unit gain in areal coral biomass. <u>Inclusions:</u> labour (some labour was volunteer labour), boat trips, consumable costs (dive gear), tank and scuba rental, planting equipment, nursery materials for all phases including the collection of coral material, nursery installation, maintenance and rearing, transplantation, and monitoring.	US$115.70 per coral surviving after 9 months nursery rearing and 6 months post-outplant or US$0.13/$cm^2$. If no volunteer labour included cost increased 6.2% to $122.87/coral
Growth and survivorship of juvenile corals outplanted to degraded reef areas in Bolinao-Anda Reef Complex, Philippines (Villanueva et al., 2012)	Bolinao-Anda Reef Complex, Philippines	Cost-effectiveness based upon survivorship	1 year (2010 - 2011) 6-months rearing in land-based hatchery 6-month survivorship on reef	Sexual propagation and outplanting using seeded juvenile recruits reared in ex-situ nurseries	221 juvenile corals seeded onto plastic masonry wall plugs outplanted to an area of $12m^2$	<u>Inclusions:</u> Equipment, consumables, and labour costs ($1.31-$3.5/hour) required in each stage: pre-restoration surveys and collection of gravid colonies, ex-situ hatchery-rearing over 6 months, deployment of colonies into in-situ nurseries, outplanting coral juveniles to the reef, and monitoring of outplants. Based upon costing framework in Edwards et al (2010). Vessel costs: $44.4/day + Dive gear $20.00/person/day	Outplanted cost for 221 juvenile corals: US$5.3 (2011 $USD) Realised cost for remaining outplanted colonies surviving after 6 months (61% survival): US$11.20.

1.4 Value of stakeholder-led coral restoration and the reef tourism industry

Upscaling is a key challenge for all restoration efforts with automation being considered as a potential solution for reef-scale intervention (Gibbs, 2021). Currently however, it is networks of practitioners that are operating at increasing local scales through collective action and the application of low-cost tools and workflows (e.g., Unsworth et al., 2021; Howlett et al., 2022). Stakeholder-led, in-water coral propagation and outplanting programs have shown promise in facilitating targeted recovery of locally impacted reef areas (e.g., Montoya-Maya et al., 2016; Hein et al., 2020; Peterson et al., 2023). Such programs further deliver the substantial co-benefit of equipping local stakeholders with new tools to actively manage the reef sites they steward (Kittinger et al., 2016; Hesley et al., 2017; Hein et al., 2019; Williams et al., 2019; Howlett et al., 2022; Palou Zúniga et al., 2023). As a result of such community partnerships, several of these efforts represent some of the longest running and largest scale reef restoration programs globally (see Bayraktarov et al., 2019; Boström-Einarsson et al., 2020).

A large proportion of restoration programs globally to date have been led or implemented by the reef tourism industry, primarily through asexual propagation and outplanting approaches (e.g., in the Caribbean and Eastern Tropical Pacific (Bayraktarov et al., 2020; Virdis et al., 2021; Blanco-Pimentel et al., 2022) and the Indo-Pacific (Okubo & Onuma, 2015; Hein et al., 2020; Howlett et al., 2022; Razak et al., 2022). Whilst such methods are not feasible for intervention at the scale of reef systems (Boström-Einarsson et al., 2020; Vardi et al., 2021; DeFilippo et al., 2022; McLeod et al., 2022), they are suitable for targeted intervention at socio-economically 'high-value' reef tourism sites (e.g., Spalding et al., 2017). At this targeted and local scale, restoration-based assisted recovery of coral assemblages is a vital first step towards maintaining or regenerating coral cover and habitat value (Hein et al., 2020), which in turn underpins the reef site's tourism value (Abrina & Bennett, 2018; Suggett et al., 2023). As such, stakeholder-led restoration represents an important means to sustain local tourism, promote reef stewardship, respond to, and mitigate reef decline in the near term (Hein et al., 2020, 2021; Howlett et al., 2022).

In Australia, tourism operations underpin ~90% of the economic revenue and jobs created by the GBR, which contributes $6.4 billion to the Australian economy every year (O'Mahoney et al., 2017). Reef sites near the GBR's major tourism gateways, the Cairns-Port Douglas and

Whitsundays regions are among the most heavily used in the GBR Marine Park, drawing 90% of the reef's annual visitors (GBRMPA & Queensland Government, 2021). Such sites can generate upwards of US$1 million/km^2/year in tourism and recreational value (Spalding et al., 2016; 2017), and are therefore priority reefs for targeted rehabilitation interventions. As for many locations, reef tourism operators on the GBR represent a highly skilled and knowledgeable workforce that are increasingly playing an active role in the conservation and preservation of the reef sites they steward (Howlett et al., 2022; Bartelet et al., 2023). Consequently, the research community, management authorities and funding agencies are increasingly investing in and collaborating with tourism operators and other stakeholder groups, such as reef Traditional Owners, to co-deliver stewardship-based monitoring and reef restoration activities (Hein et al., 2020; GBRMPA, 2021; Australian Institute of Marine Science (AIMS), 2022; McLeod et al., 2022). One such program, built upon a collaborative partnership between researchers and reef tourism operators – the Coral Nurture Program (CNP) (described in Howlett et al., 2022) – has achieved a scale of coral propagation activity that is yet unprecedented in Australia. Preliminary cost assessments by Suggett et al. (2020) at one CNP site demonstrated an estimated outplanting cost of as little as US$1/coral (Table 1.1). However, since this assessment, activity has scaled to >100,000 corals outplanted by nine high-standard tourism operators at 30 'high-value' reef sites in the Cairns-Port Douglas and Whitsundays regions (Fig. 1.3).

1.5 Thesis roadmap, aims and objectives.

Increasing government commitments to preserve and rebuild reef ecosystem health alongside rapidly growing financing opportunities to scale ecosystem restoration efforts (Saunders et al., 2020; Hein & Staub, 2021; Suggett et al., 2023) has created a time-sensitive need to resolve the cost-effectiveness of coral restoration activities. Despite rapid acceleration of reef restoration projects worldwide (e.g., Boström-Einnarsson et al., 2020; Vardi et al., 2021) few have reported activity costs. It remains unclear how coral restoration projects can meet their goals within available resources, and ultimately provide an economic justification for activity to continue (or not), let alone scale (e.g., McAfee et al., 2021; Suggett et al., 2023). Such constraints are particularly evident for Australia, where growing restoration investment and research (Anthony et al., 2020, McLeod et al., 2022) as well as stakeholder and management appetite for restoration implementation (GBRMPA 2017; Howlett et al., 2022), has not been evaluated relative to activity cost-effectiveness. No cost-evaluation has been performed for multi-site coral outplanting efforts in Australia, where higher wage costs, strong environmental regulation, and large distances to reef sites (30-50 km) preclude direct comparison with - or indeed application of – costs reported in other reef restoration programs globally to inform project planning and investment. As reef restoration efforts in Australia develop and scale, assessments of coral restoration techniques, workflows, and cost-effectiveness are critically needed to improve understanding of the region- and site-specific suitability and socioeconomic viability of stakeholder-led coral propagation approaches within broader reef management strategies (McLeod et al., 2022). **The overall aim of this thesis is therefore to identify and understand the cost-effectiveness of tourism industry-driven coral outplanting-based restoration efforts across diverse high-value sites and modes of tour operations on the GBR**. In doing so, I aim to critically inform the ongoing application and future scaling of coral restoration across the diverse socio-ecological contexts along the GBR (Burrows et al., 2019).

To address this overarching goal, two data chapters deliver the following aims and hypotheses, which were conducted across the two major tourism hubs for the GBR (Fig. 1.3); specifically, within (1) ongoing restoration activities by the CNP on the mid-outer shelf reefs in the Cairns-Port Douglas region, during the CNP "scaling phase" (from one operator at one site in August 2018 to five operators at 23 sites by December 2021) (Howlett et al., 2022) (Chapter 2) and (2) establishment of CNP activity in the inshore fringing reefs of the Whitsundays region ("launch phase", three sites and operators, August 2022 – June 2023) (Chapter 3) (Fig. 1.3).

> **Aim 1 (Chapter 2): Determine the range and variability in cost-effectiveness of established coral outplanting activities by the tourism industry on the northern Great Barrier Reef (Cairns-Port Douglas).**

I first evaluated the cost-effectiveness of coral outplanting and propagation by five tourism operators through the CNP on the northern GBR (Cairns-Port Douglas), from its inception in mid-2018 to December 2021 (Fig. 1.3). Importantly, this enables better understanding of the 'operational' costs associated with stakeholder-led coral outplanting practices, and the current effectiveness of delivering new (surviving) coral biomass to the reef. Cost data – planting versus other essential activity considered central to regulating outplanting effectiveness (e.g., nursery maintenance, training)– was opportunistically derived from routine CNP activity reporting forms. Survivorship of outplanted corals was assessed *in-situ* across space (five Reefs) and over time (two sites, 9-12 months) to forecast ‘realised’ planting costs based on coral outplants retained on the reef. In doing so, the following hypotheses were tested:

(i) Integration of coral outplanting into routine tourism operations enables lower-cost coral outplanting.
(ii) Additional stewardship activities, that in turn regulate outplanting effectiveness, will increase costs owing to time investment away from outplanting.
(iii) Under the CNP approach, where staff costs represent the greatest expense, coral outplanting costs will be largely governed by planting output, which in turn is governed by staff time allocation to planting.
(iv) ‘Realised’ costs will increase over the first year of outplant establishment owing to decreasing survivorship.

An amended version of this chapter has been published:

Scott, R.I., Edmondson, J., Camp, E.F., Agius, T., Coulthard, P., Edmondson, J., Edmondson, S., Hosp, R., Howlett, L., Roper, C. and Suggett, D.J. Cost-effectiveness of tourism-led coral planting at scale on the northern Great Barrier Reef. *Restoration Ecology, e14137.* https://doi.org/10.1111/rec.14137

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Aim 2 (Chapter 3): Examine cost-effectiveness of early-phase CNP coral restoration implementation and operation via adoption in the Whitsundays (Great Barrier Reef) tourism hub.

Arguably *the* validity test for restoration interventions and stewardship models as effective reef recovery aids, once operational and effective at small scales, is whether wider adoption is feasible and cost-effective across more diverse reef environments and tourism operations. In August 2022, CNP coral propagation and outplanting was initiated at three inshore fringing-reef sites in the GBR’s other major tourism hub, the Whitsundays (Fig. 1.3). This posed a unique opportunity to examine the early adoption (including initial public consultation), operation and cost-effectiveness of CNP activity, relative to the underlying environmental (reef state, water quality) and operational (public moorings) conditions. Benthic surveys were conducted within pre-selected outplanting areas at three commonly visited tourism reef sites heavily impacted during Cyclone Debbie in 2017, prior to the inception of propagation activity. At each site, fate-tracked plots were then established to evaluate early-stage survivorship of coral outplants during the first 9 months of establishment. Costs of implementing activity during this entire process were evaluated to identify the successes, challenges, and complexities of adapting the CNP stewardship approach. In doing so, the following hypotheses were tested:

(i) Costs in the early phases of restoration program activity (Whitsundays) outweigh operational costs of ongoing activity previously determined for Cairns-Port Douglas (Chapter 2) via site-setup costs.

(ii) Coral outplant survivorship will be lower than for Cairns-Port Douglas (Chapter 2) given lower starting coral cover and environmental conditions (e.g., high sedimentation and eutrophication in in-shore reef environment) and less experienced personnel.

This chapter is presented as a fully drafted article prepared for submission to a peer-reviewed journal:

Scott, R.I., Suggett, D.J., Hayward, C., Edmondson, J., Gillette, G., Howlett, L., Roper, C., Strudwick, P., Camp, E.F. Early-stage outcomes and cost-effectiveness of implementing tourism-led coral propagation and outplanting in the Whitsundays (Great Barrier Reef).

Finally, the knowledge gathered through delivering these two aims is considered in **Chapter 4,** where I integrate my findings for CNP cost-effectiveness for the two tourism hubs of the Great Barrier Reef (Cairns-Port Douglas *vs* Whitsundays). In doing so, I identify future critical directions for the application of this research in evaluating the financial viability, feasibility, and cost-benefits of sustained or future investment and implementation of coral propagation and outplanting for active reef management on the GBR and elsewhere.

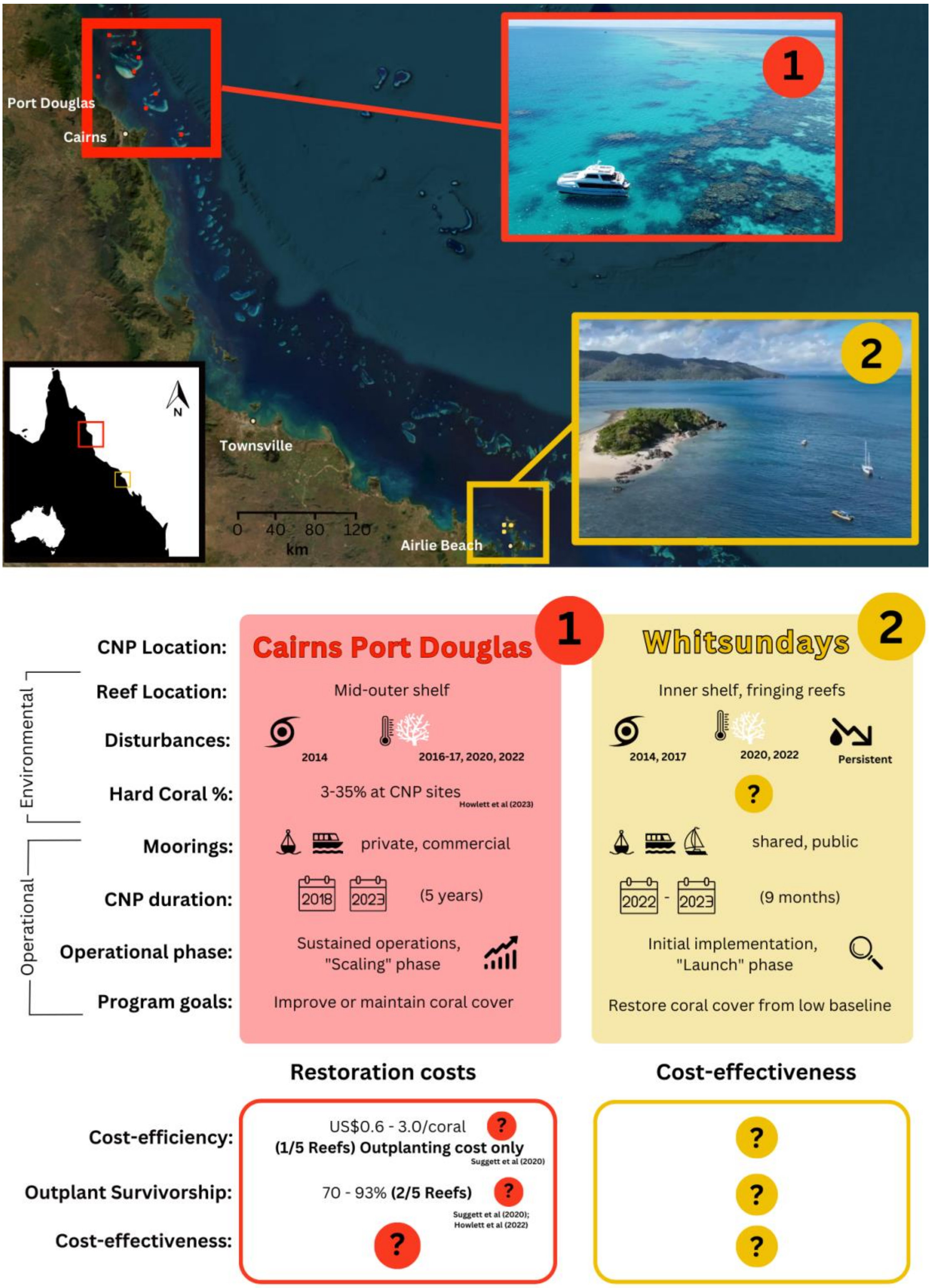

Figure 1.3. This thesis focuses on evaluating the cost effectiveness of tourism-led coral propagation activity on Australia's Great Barrier Reef via the Coral Nurture Program (CNP). Located in the GBR's two major tourism gateways – Cairns-Port Douglas **(Aim 1)** and the Whitsundays **(Aim 2)** - the two programs represent different ecological and operational contexts and phases of activity implementation. Question marks represent key areas

of knowledge that this thesis aims to progress, specifically; **Aim 1 (Chapter 2)** aims to extend previous outplanting cost and survivorship estimates by Suggett et al (2020) and Howlett et al (2022) to determine the range and variability in costs of coral outplanting activities by the tourism industry on the northern Great Barrier Reef (Cairns-Port Douglas); **Aim 2 (Chapter 3)** aims to examine cost-effectiveness of early phase CNP coral restoration implementation and operation via adoption in the Whitsundays (Great Barrier Reef) tourism hub.

Chapter 2: Cost-effectiveness of tourism-led coral outplanting at scale on the northern Great Barrier Reef

An amended version of this chapter has been published in *Restoration Ecology:*

Scott, R.I., Edmondson, J., Camp, E.F., Agius, T., Coulthard, P., Edmondson, J., Edmondson, S., Hosp, R., Howlett, L., Roper, C. and Suggett, D.J. Cost-effectiveness of tourism-led coral planting at scale on the northern Great Barrier Reef. *Restoration Ecology, e14137.* https://doi.org/10.1111/rec.14137

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Author Contributions: RIS, DJS, EFC, JE conceived the research; JE led the metal detector development for locating Coralclip®; JE, TA, PC, JE, ST, RH, and crew conducted coral outplanting, facilitated data collection; LH, CR, RH supported data collection; RIS conducted data analysis, drafted the manuscript, with subsequent editorials from all other authors.

All authors declare that the author contributions outlined above are a true reflection of the authorship of this thesis chapter:

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2.1 Abstract

Stakeholder-led coral reef restoration efforts, aimed at locally retaining or rebuilding coral populations, have rapidly grown over the last two decades. However, the cost-effectiveness – and in turn viability – of these coral restoration efforts remains rarely reported. We therefore evaluated coral outplanting cost-effectiveness across the first 3.5 years of the Coral Nurture Program (CNP), a coral restoration approach integrated within tourism operations on Australia's Great Barrier Reef. CNP operator activity reporting forms (63,632 corals outplanted, 5 tourism operators, 23 reef sites) were used to opportunistically calculate coral planting costs (PC, \$US coral^{-1} trip^{-1}) for routine outplanting activity versus when additional stewardship activities – that regulate outplanting effectiveness – were undertaken (e.g., nursery maintenance). Mean PC (± SE) was US\$2.30 ± 0.19 coral^{-1}-trip^{-1} (ranging \$0.81 - \$5.74, 5th - 95th percentile), but increased 2- to-7-fold on trips where nursery propagation, site maintenance or staff training was conducted to support outplanting efforts. The 'realised' cost (PC_R) of establishing coral biomass was subsequently determined by evaluating outplant survivorship across space (9 sites, n = 4,723 outplants) or over time (2 sites, n = 600 outplants). Outplant survivorship varied by both site (mean 68-88% across 5 reefs) and over time (mean 59-71% after 9-12 months), resulting in costs increasing from PC to PC_R by 25-71%. We demonstrate how integration of restoration activity into tourism operations creates potential for cost-effective coral outplanting at high-value reef sites and discuss important steps for improving cost-accounting in stakeholder-led restoration programs that may be similarly positioned to routinely determine their cost-effectiveness.

2.2 Implications for Practice

- Transparent cost-tracking of coral restoration efforts is critical to justify feasibility and investment.
- Coral outplanting led by tourism operators on the Great Barrier Reef enables low-cost coral outplanting (US\$2.30 ± 0.20 coral^{-1} trip^{-1}, mean (± SE)) via existing vessel infrastructure and personnel.
- Other essential – but infrequent – activities necessary for effective outplanting modify costs to US\$5.79 ± 0.81 coral^{-1} trip^{-1} when involving nursery propagation and maintenance, or US\$16.22 ± 5.15 when training staff.

- Accounting for outplant survivorship (by site or time) increases costs by 25-70% and is hence a necessary consideration where outplanting costs are used to justify restoration effectiveness.
- Accurate accounting of staff time dedicated to wider restoration activities that govern outplanting effectiveness is needed to improve cost estimates.

2.3 Introduction

Progressive declines in coral reef ecosystem health through climate change and localised impacts are driving modern reef management to implement proactive interventions alongside existing threat mitigation and habitat protection (Kleypas et al., 2021; McLeod et al., 2022). Global uptake of reef restoration interventions – particularly via in-water asexual propagation and outplanting (Boström-Einarsson et al., 2020) – has grown in the past decade as stakeholders attempt to boost coral population and reef recovery capacity at local scales (e.g., Bayraktarov et al., 2020; Hein et al., 2020; Howlett et al., 2022). Current efforts are rapidly accelerating as in-water nurseries (e.g., Howlett et al., 2021) and coral outplanting methods (e.g., Suggett et al., 2020; Unsworth et al., 2021) become cheaper and more efficient and as practitioners network to collectively learn (Quigley et al., 2022; Vardi et al., 2021). Furthermore, recent declarations of decadal priorities in restoration and ocean science have catalysed new financing mechanisms geared towards advancing the scale, equity, and sustainability of reef restoration efforts (Hein & Staub, 2021; Suggett et al., 2023).

Several programs have shown promise in facilitating targeted and scalable recovery of locally impacted reef areas (e.g., Montoya-Maya et al., 2016; Hein et al., 2020; Peterson et al., 2023) whilst simultaneously building the site stewardship capacity of reef stakeholders (e.g., Hesley et al., 2017 in Florida; Bayraktarov et al., 2020 in Latin America, and Howlett et al. 2022 on the Great Barrier Reef (GBR)). However, few restoration programs have reported costs needed to justify on-going investment and/or develop operational strategies to improve cost-efficiencies in practice (Bayraktarov et al., 2019; Quigley et al., 2022). As such, delivery of informed and sustained investment into restoration practices remains challenged by uncertainty regarding the cost and feasibility of different approaches (Bayraktarov et al., 2019; Hein & Staub, 2021; Suggett et al., 2023). Costs involved in coral restoration efforts are highly context-specific, spanning multiple activities that directly or indirectly carry monetary value. As such, where coral restoration costs have been reported, approaches have typically not been comprehensive, standardised, or transparent (Edwards et al., 2010; Iacona et al., 2018; Bayraktarov et al., 2019). Important contextual details underpinning costs are often absent; such as labour costs reported in local monetary values rather than comparable units of time (Edwards et al., 2010), currency conversion rates (Bayraktarov et al., 2019), disclosing where volunteer labour or in-kind contributions have been employed (Edwards et al., 2010), or factoring in project life cycle costs from planning through to monitoring (Spurgeon & Lindahl, 2000; Bayraktarov et al., 2019). Collectively such inconsistences in cost reporting can limit the ability of restoration practitioners and reef managers to evaluate ongoing cost-effectiveness, identify context-specific suitability, or develop realistic budgets for future implementation (Bayraktarov et al., 2015). Such lack of transparent reporting – and how it relates to activity goals – may further undermine efforts to provide trust and confidence to future investment opportunity (Suggett et al., 2023).

A novel coral propagation and planting approach on Australia's GBR driven by the tourism industry (Coral Nurture Program (CNP); Howlett et al., 2022) has shown promise in resolving several logistical and cost-efficiency constraints in rehabilitating coral populations at high-value reef sites (Suggett et al., 2020; Howlett et al., 2021, 2022, 2023). CNP was initiated in 2018 in response to the 2016-2017 mass bleaching events on the GBR and conceived to build capacity for high-standard reef tourism operators and other reef stakeholders. Activities were geared to assist recovery of hard coral cover at reef sites regularly accessed during tourism trips thereby harnessing existing resources deployed on the reef (i.e., vessels, trained personnel, equipment, and in-depth site knowledge). In parallel to the lower operational costs enabled by this approach, development of a novel coral attachment device (Coralclip®) has shown improved planting safety, speed and costs compared to previous methods (Suggett et al., 2020). However, these previously reported planting cost estimates for Coralclip® (US$0.6-3.0/coral) were based on only ~4,500 outplants at a single reef and therefore unlikely captured the diversity of coral outplanting costs, given the range of reef site ecologies and tourism operations across the CNP (described in Howlett et al., 2022).

Here we evaluate the broader cost of tourism-led coral outplanting operations using Coralclip® by examining over three years of CNP activity that resulted in 63,632 corals planted by five diverse tourism operations across 23 sites on seven reefs of the northern GBR (from 271 planting trips). We used CNP daily activity reporting forms to determine the range in outplanting costs under "routine" outplanting as well as other operational contexts, and in turn discuss the factors that influence costs of coral outplanting as part of assisted site recovery. To better resolve outplanting cost-effectiveness during this period, we further evaluated the survivorship of outplanted coral material. By adjusting outplanting costs with subsequent survivorship (Edwards et al., 2010), we determine the 'realised' cost of establishing new coral biomass on the GBR through tourism industry-driven site stewardship, providing the first cost assessment of targeted coral outplanting efforts at scale in Australia.

2.4.0 Methods

2.4.1 Coral Nurture Program operational context, activity, and data capture.

Our cost analysis focuses on CNP coral outplanting activities at sites on the northern GBR approximately 30-50km offshore from Cairns to Cape Tribulation, between August 2018 to December 2021 (Fig. S2.1, see also Howlett et al., 2022; 2023). Coral outplanting and propagation activity was initiated by one tourism operator at Opal Reef in August 2018 (under permit G18/40023.1), and after an initial validation phase was scaled to include four additional operators with commercial moorings at Mackay, Hastings, Upolu and Moore Reef from January 2019 to February 2020 (under permits G19/42553.1 and G20/43740.1) (Howlett et al., 2022). By the end of December 2021, tourism operators had outplanted corals at 23 distinct high-value tourism sites spanning seven diverse reefs on the GBR (Fig. S1.1). Activity remains ongoing at the time of publication. All outplanting used the metal attachment device, Coralclip® (Fig. 2.1, Suggett et al., 2020), for predominantly branching *Acropora* and *Pocillopora* species sourced largely as naturally detached fragments (Corals of opportunity (CoO)) (Howlett et al., 2022), supplemented by corals propagated on mid-water nursery platforms (see Fig. S2.2; Howlett et al., 2021), and occasionally from wild donor colonies within permit requirements.

Intensity and frequency of coral outplanting, propagation, and site maintenance activities (herein referred to as "CNP activity") was dependent upon operational factors (such as site access opportunities, trained personnel availability, tourism guest numbers, funding availability, operator preference) as well as local site conditions (e.g., availability of bare substrate or coral material for outplanting, nursery maintenance needs etc.) (Howlett et al., 2022). To maximise cost-effectiveness, CNP activity was originally conceived for integration into routine activity where additional paid staff (e.g., a dive-buddy pair) joined existing tourism day trips. In practice, this "routine" approach was periodically complimented by more intensive activity with vessel use "dedicated" to coral outplanting (i.e., non-tourism trip). Both "routine" and "dedicated" days could also include other stewardship activities such as maintenance of nursery structures and outplant areas, reef health surveys, or corallivore control (e.g., Crown of Thorns starfish (*Acanthaster planci*) or *Drupella* spp.) at site.

Figure 2.1. Example images of coral fragments of varying ages planted with Coralclip®, consisting of a stainless-steel spring-loaded clip secured into consolidated substrate with a masonry nail: Top row of images show *Acropora millepora* as a new outplant (left), ~12 months post-outplanting where the Coralclip® is no longer visible (middle) and colonies planted in June 2019 that spawned in November 2021. The middle row of

images show *Acropora intermedia* as a new outplant (left), 12 months post-outplant and fused to the substrate (middle), and colony >12 months old (right) (Note: images for both species are not of the same colony). Coralclip® is often still visible for established outplants of arborescent branching species, and thus in roving surveys, these corals would be counted by the visual surveyor. The bottom row shows the metal detector used in roving survivorship surveys on an established *Acropora sp*. outplant (left) and a reefscape with a mix of planted colonies and wild colonies demonstrating the difficulty in distinguishing between the two (right) (photos: J.Edmondson, R.Scott).

Many of the costs associated with coral propagation and outplanting (e.g., diving equipment and vessel use) were largely absorbed by operators where resources were already in use for tourism operations. However, costs for CNP staff, nursery materials, outplanting equipment and occasional dedicated vessel charters were compensated through funding sources; specifically; (i) from 2018 to the end of 2019 by the Australian Federal and Queensland Governments; (ii) during the COVID-19 pandemic in 2021 via the Great Barrier Marine Park Authority's "Tourism Industry Activation and Reef Protection Initiative" (GBRMPA, 2021; see Howlett et al., 2022); and (iii) from 2021 onwards by the Australian Government Reef Trust in partnership with the Great Barrier Reef Foundation, and private funding from Diageo Australia. No funding was available in 2020, and CNP activity was fully absorbed at the operators' own cost.

To meet permitting requirements, all CNP activity was recorded using a standardised CNP reporting form. Operators documented outplanting and propagation activity for each trip including the reef sites visited, the number of personnel conducting activity, and the quantity, taxonomy (identified to species, if possible, otherwise genus and morphology), and origin of coral outplants (CoO, nursery-propagated material, or wild donor colonies). Operators also recorded details on nursery maintenance (noting installation of frames, addition of coral material and occasional removal of biofouling organisms) and other site maintenance activities (e.g., corallivore removal, outplant and site monitoring). Given our aim was to quantify costs for coral outplanting, trips where reporting forms did not differentiate personnel time allocation to outplanting and non-outplanting activity were excluded from this analysis. As such, of the 63,632 corals planted across 271 trips during this period, only 67% (43,054 corals over 154 trips) were used for our costing dataset.

We next filtered our costing dataset to resolve coral outplanting costs under four different operational contexts: (a) Routine Planting Days – outplanting activity only during tourism day trips (30,556 outplants over 110 trips); (b) Propagation and Maintenance Days – mixed activity days where nursery maintenance and propagation, site maintenance and monitoring were reported in addition to outplanting (3,298 outplants over 30 trips); (c) Training Days – dedicated to training tourism personnel in outplanting with Coralclip®, coral identification and propagation techniques (848 corals outplants over 6 trips); and finally, (d) Dedicated Planting Days – non-routine tourism days where either vessel use was covered by external funding for the purpose of high-throughput outplanting, or were representative of trips dedicated to conducting stewardship activity at sites less desirable for tourism visitation (i.e., for rehabilitating degraded or storm-damaged sites (8,877 outplants over 8 trips)).

2.4.2 Quantifying the costs of coral outplanting under different operational contexts.

'Planting costs' were calculated per coral for each outplanting day trip (PC, US$ coral^{-1} trip^{-1}); specifically, PC is expressed as the sum of labour, materials, and vessel costs relative to the number of corals planted (Equation 1):

$$\textit{(1)}\quad PC = \frac{\left((\$S \cdot FTE) + \$D)*n_{(D)}\right) + \left(\$c \cdot n_{(F)}\right) + \$V + \$P}{n_{(F)}}$$

Where \$S is the daily wage per staff member, FTE is a Full Time Equivalent weighting (quantifying staff time contribution to CNP activity), \$D is diving costs per diver, and $n_{(D)}$ is the number of divers conducting planting (Table 1). Also, \$c is the cost per Coralclip® attachment device, $n_{(F)}$ is number of coral outplants, \$V is the vessel cost for accessing sites, and \$P is the capital cost for outplanting equipment. Each factor in Equation 1 is treated as fixed (\$S, \$c, \$P) or variable (\$D, FTE, n_D, n_F, \$V), and subject to several assumptions (Table 2.1): To determine staff time contribution to CNP activity within routine activity ("Routine Planting Days" and "Propagation and Maintenance Days") we applied an FTE weighting, which was calculated as the number of sites reported in CNP trip reporting forms expressed as a proportion of the total number of sites visited per day trip (Table 2.1). Whereas, for "Training Days" and "Dedicated Planting Days", staff time for the entire day was dedicated to CNP activity and hence FTE was assumed as 1.0. Vessel costs (\$V) were assigned a value of \$0 where CNP activity was integrated within routine tourism trips, however for "Dedicated Planting Days", we assigned an 'at-cost' vessel charter value (\$US 2,700). Finally, diving gear costs (\$D) were typically absorbed by operators as cost-efficiencies (\$0) since gear was already being utilised for diving operations or vessel/mooring maintenance, however for this cost analysis we assigned a variable $D value based upon the number of sites reported. All costs were calculated in Australian dollars ($AUD) and subsequently converted to US dollars ($US) using the mean daily exchange rate between August 2018 and December 2021, where US$1.00 = AU$1.28 (MacroTrends, 2022, Table 2.1).

Table 2.1. Description of the value and rationale for the variables involved in calculating coral planting cost (PC, $US coral^{-1} trip^{-1}) values for Coral Nurture Program (CNP) propagation, planting, and maintenance activity (Equation 1). All values are per trip. All costs listed are in $US based on fixed $2018 values but were converted from $AUD using the mean daily exchange rate between August 2018 and December 2021 (US$1.00 = AUD$1.28). Variable factors are those that change with any given trip, whereas fixed factors are kept constant across all trips over time.

Factor	Value ($US)	Description	Rationale	Factor treatment
Staff Wages ($S)	$225.00	Compensated labour costs per tourism staff member (8-hour workday), which includes return travel time to reef sites, and between 60-180 minutes of total dive time.	Fixed value over 2018-2021 determined through prior consultation with operators and funders. For "Dedicated Planting Days" and "Training Days", costs were calculated with in-kind research student and staff labour accounted for (i.e., $225/day).	Fixed
Diving Gear Costs ($D)	$3.79 - $9.55	Per diver assuming 1 dive for conducting CNP activity per site with a dive equipment cost of $0.91 per day and $2.88 per SCUBA tank refill (1 tank per site) (Table S2.1).	Dive equipment cost based upon the daily cost of a full set of diving gear ($1,080) with a lifespan of 5 years and four annual services costing $144. Operators have access to air compressors for filling tanks for diving operations, and hence tank costs are lower than commercial refills from dive shops ($7-11 per tank).	Variable
$n_{(D)}$	#no. of divers	Number of staff conducting CNP activity on any given day/trip.	This number occasionally includes in-kind research staff or student (i.e., volunteers) whose time we costed at $225/day.	Variable
Full Time Equivalent (FTE)	0.33, 0.67, 1.0 (3 sites); 0.5, 1.0 (2 sites); 0.5 (1 site)	FTE calculated as the number of sites reported for CNP activity expressed as a proportion of the maximum number of sites visited per trip for each respective operation, and assuming 1 dive was conducted per site: maximum 3 sites (FTE = 0.33 for 1 site, 0.67 for 2 sites or 1.0 for 3	For "Routine Planting Days" and "Nursery, Propagation and Maintenance Days", FTE weights staff time contribution to CNP activity based upon the number of sites where activity is conducted.	Variable

		sites), maximum 2 sites (FTE = 0.5 for 1 site or 1.0 for 2 sites). For tourism operations which routinely visit only 1 site in a trip, we assumed CNP activity was conducted for one out of two possible dives (FTE = 0.5).	For Training Days and Dedicated Planting Days, staff time for the entire day is dedicated to CNP activity and hence FTE was assumed as 1.0.	
Cost per Coralclip® ($c)	$0.20	Unit cost per Coralclip® planting device (one used per each coral planted).	Inclusive of materials cost, and labour costs for assembly.	Fixed
$n_{(F)}$	No. coral planted	Number of reported coral fragments planted per trip.	Operator practice is to count a set Coralclip® number (e.g., 100) before entering the water, and counting the remaining number upon finishing the dive.	Variable
Vessel Costs ($V)	$0.00 or $2,700	Cost of a full-day (~8 hour) return trip to outer reef sites (approximately 30-50 km offshore). Operator vessels are >24 m dual-hull catamarans requiring minimum 3 crew to operate. The true running costs for large tourism vessels on the GBR can be upwards of $5,000 per day (including overheads) however Dedicated Planting Days were chartered at cost-price.	Vessel operations costs were absorbed where CNP activity was conducted within routine tourism trips – i.e., $0 for "Routine Planting Days", "Planting, Propagation and Maintenance Days" and "Training Days". An 'at cost' vessel charter value of $2,700 was applied to "Dedicated Planting Days", covering operational expenses (fuel, skipper, and wages for necessary crew).	Variable
Planting equipment capital cost ($P)*	$2.36	Planting equipment includes a hammer ($6.48), scrubbing brush ($3.60), wire mesh or plastic basket for holding fragments and Coralclip® units ($18.00), and a chisel or wire cutters for fragmenting corals ($14.40).	Value calculated based upon biennial replacement (equipment lifespan of 2 years) of 4 sets of equipment (US$84.96/year), assuming 33 trips for CNP activity are conducted per year (average number of trips across operators in 2021) (Table S2.1).	Fixed

* Note we have not considered the capital costs of nursery frames in this current exercise (approx. US$60 per frame, holding up to 250 corals at any one time (see Suggett et al., 2020)). This is because (a) CoO accounted for ~90% of outplants during this period (Howlett et al., 2022), (b) propagated colonies are regularly pruned to collect fragments rather than planting entire colonies, and (c) operators routinely plant a mix of nursery-propogated and CoO on any given trip; hence it is not possible to apply an accurate propagation figure per outplant.

To examine the relationship between coral outplanting output and cost (PC, Eqn. 1), we further calculated the number of corals outplanted per diver per site, Planting Output (PO, corals diver^{-1} site^{-1}) for each trip as (Equation 2):

$$(2) \qquad PO = \frac{\frac{n(F)}{n(D)}}{n(sites)}$$

2.4.3 Quantifying survivorship of outplanted corals.

Attachment and survival of coral fragments outplanted with Coralclip® have been evaluated at various CNP sites on an *ad-hoc* basis since 2019, using either (i) a rapid roving survey technique to capture Coralclip® effectiveness of haphazard planting across a large area and sample size (as per Suggett et al. 2020), or (ii) marked fate-tracked plots to evaluate species and site-specific survivorship (Howlett et al., 2022). These prior fate-tracking assessments yielded outplant survivorship ranging 58.0% to 95.8% (Table S2.2). A similar dual survivorship assessment was employed for our current study to quantify both broad-scale coral outplant survivorship across reefs, and site-specific outplant survivorship within defined time periods to enable more robust inter-comparability across locations as follows:

Fate-tracked plots were established in September 2021 at sites at Mackay Reef ("Angels") and Hastings Reef ("1770") (Fig. S2.1). Triplicate 40 m^2 (4x10 m) marked belt transects were each outplanted with 100 CoO of mixed *Acropora* and *Pocillopora* species using Coralclip®: using a hammer, a diver would embed the masonry nail of the Coralclip® into bare rock, firmly brush the area free of algae or other debris and position the coral fragment securely under the stainless-steel spring-loaded clip (Fig. 2.1) (Suggett et al., 2020). Transects were planted in areas of bare substrate, 5-20 m apart at depths of 1.5-5 m, photographed and tagged with cattle-tags for repeat surveying over time. Coralclip® attachment success and survivorship of fragments were assessed via SCUBA 2 weeks (14 days, "T14") post-outplanting via visual surveys, where observed coral outplants were counted and categorised as coral alive (fragment attached, <100% partial mortality), coral dead (fragment attached, 100% mortality), and coral missing (empty Coralclip® still in place) (as per Suggett et al., 2020). Survivorship surveys were repeated at Hastings Reef only in November 2021 (54 days post-planting, "T54"), at both sites in January and February 2022, 3-5 months post outplanting (Hastings: 115 days, "T115"; Mackay: 154 days, "T154")) and again at both sites in winter/spring 2022 (June - October), 9 - 12 months (Mackay: 261 days, "T261"; Hastings: 379 days, "T379") post-outplanting. Timepoints were not consistent across sites because of staff availability and site access constraints.

Replicate roving survivorship surveys were also conducted in summer 2022 (February) via SCUBA on existing planted areas at Opal Reef (4 sites “Rayban”, “Mojo”, “Blue Lagoon”, “Beautiful Mooring”), Hastings Reef (2 sites “1770”, “Stepping Stones”), Mackay Reef (1 site “Angels”), Upolu Reef (1 site “Wonderwall”) and Moore Reef (1 site “Moore Reef Pontoon”) (Table S2.3; Fig. S2.1). The roving method identifies every visible Coralclip® deployed on the reef within outplant areas (Fig. 2.1) to classify attachment and survivorship as above. However, Coralclip® is designed to rapidly become inconspicuous to retain site aesthetics (Suggett et al. 2020) and, once overgrown by coral tissue, it becomes difficult to distinguish between outplanted and wild colonies (Fig. 2.1). Therefore, to improve on past *ad hoc* surveys, where Coralclip® devices were overgrown by established outplants – and hence not visible – we used metal detector surveys to detect non-visible Coralclip® units to supplement visual methods (Fig. 2.1) (see Supplement 2.8.2 for full description of method development and implementation, Table S2.4-S26).

2.4.4 *Realised cost of coral outplanting.*

Planting costs for each trip (PC; Eqn 1) were finally adjusted to account for the mean survivorship of outplants, and so derive a ‘realised cost’ (cost per surviving coral outplanted) (PC_R = \$ coral^{-1} trip^{-1}). Specifically, the number of fragments outplanted for each trip ($n_{(F)}$ in the denominator of Equation 1) was multiplied by the mean value of % outplant survivorship observed for the corresponding Reef ($mS_{(Reef)}$) through roving surveys (Eqn 3):

$$\textbf{(3)} \qquad \boldsymbol{PC_R} = \frac{\left((\$\boldsymbol{S} \cdot \boldsymbol{FTE}) + \$\boldsymbol{D}) \cdot \boldsymbol{n_{(D)}}\right) + \left(\$\boldsymbol{c} \cdot \boldsymbol{n_{(F)}}\right) + \$\boldsymbol{V} + \$\boldsymbol{P}}{\boldsymbol{n_{(F)}} \cdot \mathbf{mS(Reef)})}$$

To further consider the time-dependent nature of coral outplant survival (e.g., Edwards et al., 2010; Morand et al., 2022), PC_R was also calculated using Equation 3 for the fate-tracked corals outplanted at site “Angels” (Mackay Reef) and “1770” (Hastings Reef) at respective survey timepoints between September 2021- October 2022. Here, rather than adjusting Equation 3 by the mean % coral survivorship via roving surveys, we instead used the time-specific ouplant survivorship determined in plots throughout the 9–12-month period.

2.4.5 *Data analysis.*

Statistical analysis and data visualisation were conducted using R statistical software (v4.0.0, R Core Team 2021). All variables were visualised and tested for normality and equal variance prior to undertaking statistical analysis. P-values <0.05 were considered significant for analyses of all data considered here. PO, PC and PC_R values were pooled across reefs, and summary statistics were computed for each under the four different CNP operational contexts. For "Routine Planting Days", both variables were further grouped by reef, $Log_{10}+1$ transformed to stabilise group variances and analysed for distributional differences using the non-parametric Kruskal-Wallis rank sum test and Dunn post-hoc test, applying a Bonferroni p-value adjustment for multiple comparisons. Outplant survivorship from roving surveys were determined whereby the total number of 'alive', 'dead' and 'missing' corals observed through metal detector and visual counts were pooled, resulting in an adjusted overall total for each survey. All three variables were expressed as proportions of the total (between 0 and 1), grouped by reef, arcsine-transformed to stabilise heterogeneous variances and mean proportions compared with separate one-way Analysis of Variance (ANOVA) tests with Tukey's post-hoc pairwise comparisons between reefs. To examine survivorship in fate tracked plots, counts from each replicate transect (n= 3) for corals in each category ('alive', 'dead' and 'empty') were expressed as a proportion of the total count, grouped by time point and arcsine transformed to stabilise heterogeneous variances between groups. For both sites, a series of separate one-way ANOVA tests with Tukey's post-hoc pairwise comparisons were performed to compare the mean proportion of each category (% alive, % dead, % empty) between timepoints. Time-dependent PC_R values from fate-tracked plots were visually compared using bar-graphs.

2.5.0 Results

2.5.1 Coral outplanting activity costs.

Mean (± SE) planting cost (PC) across operations was lowest for "Routine Planting Days" at US\$2.30 ± 0.19 coral^{-1} trip^{-1}, and ranging from \$0.81 - \$5.74 (5th, 95th percentile) (Table 2.2). However, single PC values as low and high as \$0.73 and \$14.84 were recorded (Fig. 2.2B). On these routine tourism days, mean (± SE) planting output (PO) was 67.04 ± 3.44 corals diver^{-1} site^{-1}, but varying widely from 17.95 - 132.40 corals diver^{-1} site^{-1} (5th and 95th

percentile, Table 2.2, Fig. 2.2A). Of note, both PC and PO differed across reefs on "Routine Planting Days", with mean values ranging from US\$1.55 ± 0.10 coral^{-1} trip^{-1} (Opal Reef) to US\$7.19 ± 1.17 coral^{-1} trip^{-1} (Moore Reef) and from 20.39 ± 2.92 corals diver^{-1} site^{-1} (Moore Reef) to 85.70 ± 8.48 corals diver^{-1} site^{-1} (Mackay Reef), respectively (Fig S2.3, Table S2.7). However, given the conflation of different tourism operations and environmental variables inherent to each reef (Howlett et al., 2022; 2023), we are unable to currently resolve these site-based differences here.

Table 2.2. Summary statistics for planting cost (PC, \$US per coral (coral^{-1}) planted per trip (trip^{-1})) and planting output (PO, number of corals planted per diver (diver^{-1}) per trip (trip^{-1})) values for trips as part of the Coral Nurture Program (CNP) between August 2018 – December 2021. Trips are classified under four different operational contexts: (a) Routine Planting Day – planting only within tourism trip (b) Propagation and Maintenance Day – mixed activity within tourism trip (c) Training Day – training on tourism trips (d) Dedicated Planting Day – non-tourism trip where vessel use is dedicated to Coral Nurture Program (CNP) activity (and is hence costed). Lower and Upper range values represent the 5th and 9th percentiles.

			PC (US\$.coral^{-1}.trip^{-1})			PO (corals.diver^{-1}.site^{-1})		
Activity (& Operational context)	**# Trips**	**#Outplants**	**Mean (± SE)**	**Lower Range**	**Upper Range**	**Mean (± SE)**	**Lower Range**	**Upper Range**
(a) Routine Planting Day	110	30,031	2.30 (0.20)	0.81	5.74	67.04 (3.44)	17.95	132.40
(b) Propagation & Maintenance Day	30	3298	5.79 (0.81)	1.82	13.69	32.64 (4.35)	8.73	72.50
(c) Training Day	6	848	16.22 (5.15)	3.37	33.07	16.30 (3.20)	7.12	24.97
(d) Dedicated Planting Day	8	8877	4.42 (0.47)	2.85	6.24	88.07 (14.85)	42.78	138.16

As expected, on "Propagation and Maintenance Days" where staff time was dedicated to activities other than coral planting, mean PO was ~50% lower than that of "Routine Planting Days" at 32.64 ± 4.35 corals diver^{-1} site^{-1} (Fig. 2.2A), and hence mean PC was increased by 152% to US\$5.79 ± 0.15 coral^{-1} trip^{-1} (Table 2.2, Fig. 2.2B). Similarly, mean PO for "Training Days" was ~25% that of "Routine Planting Days" at 16.30 ± 3.20 corals diver^{-1} site^{-1} (Fig. 2.2A), resulting in a higher mean PC of US\$16.22 ± 5.15 coral^{-1} trip^{-1} (Fig. 2.2B). Mean PO on "Dedicated Planting Days" (88.07 ± 14.85 corals diver^{-1} site^{-1}; Table 2.2, Fig. 2.2A) was higher than that for "Routine Planting Days", presumably from more intensive focus on coral outplanting; however, inclusion of vessel costs (which accounted, on average, for 60% of total "Dedicated" trip costs) resulted in a near doubling of mean PC to \$US 4.42 ± 0.47 coral^{-1} trip^{-1} compared to "Routine Planting Days".

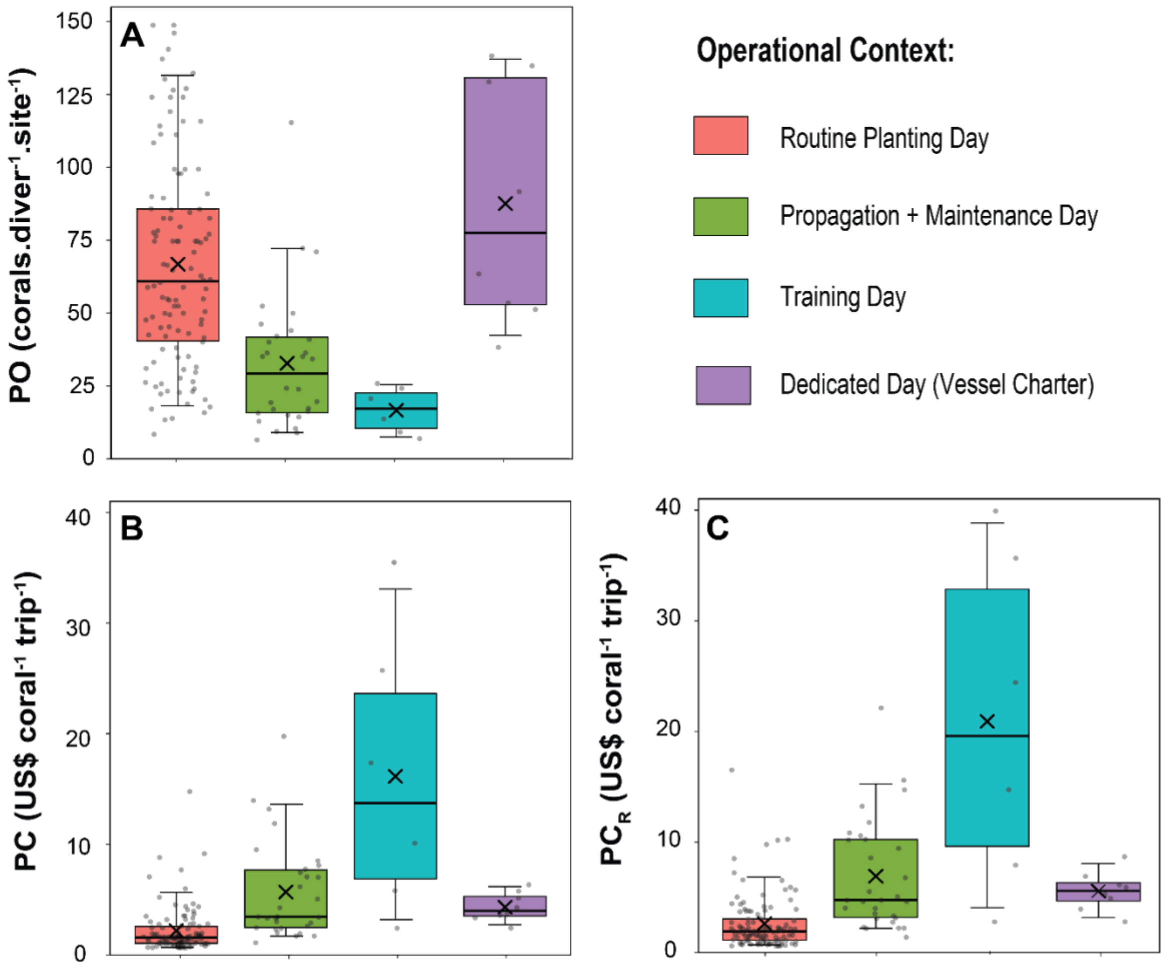

Figure 2.2.(A) Planting Output (PO, number of corals planted per diver (diver^{-1}) per site (site^{-1}), (B) Planting Cost (PC, $US per coral planted per trip) and (C) Realised Planting Cost (PCR, $US per surviving coral planted per trip) (see Table 2.4) values for planting trips as part of the Coral Nurture Program (CNP) under four different operational contexts: (i) Routine Planting Day (n = 110) – outplanting only within tourism trip (red) (ii) Propagation and Maintenance Day (n = 30) – mixed activity within tourism trip (green) (iii) Training Day (n = 6) – training on tourism trips (blue) (iv) Dedicated Planting Day (n = 8) – non-tourism trip dedicated to (CNP) activity where vessel charter costs are included (green). Boxplots show the median (centre line) and interquartile range (coloured box), with whiskers representing the 5th and 95th percentiles. Black crosses overlain on boxplots show mean values. Grey dots represent data points.

In extracting data to determine PC (and PO) from the routine operations logs, diverse logistical and environmental factors appeared to impact the workflow of CNP activities – and hence costs, which are summarised in Table 2.3. Factors included site access, coral material collection, propagation, outplanting, monitoring and maintenance. Such factors were identified in any number of combinations and presumably contribute to the dynamic range of PC (PO) reported here. We return to these factors in the discussion.

Table 2.3. Operational and environmental factors that regulate the workflow of coral propagation and out-planting activities and how these factors influence planting output (PO), planting cost (PC) and the realized cost of planting (PC_R). Activities are often not time or cost-tracked but interactions between factors contribute to the restoration cost-efficiency life cycle involved in boosting live coral cover. We therefore identify the core attributes required to resolve costs more accurately. CoO = Corals of Opportunity. CNP = Coral Nurture Program.

Activity	Factor influencing PC	Considerations for PC (and/or PO, PC_R)	Cost attributes
Site access	• Distance to reef site from port • Underlying site condition	• Island-based or fringing reef operations potentially require smaller vessels and enable greater time at site whilst removing (reducing) fuels costs • Near-shore sites may experience reduced water flow and/or greater nutrient/sediment loads, which can increase potential for fouling (greater nursery cleaning; poorer outplant survivorship). • Degraded reef sites (low live coral cover) are targeted for assisted site recovery but are less desirable for tourism visitation, and so may necessitate more costly "Dedicated Planting Days" – however efforts at degraded sites may deliver greater ecological benefit	• Fuel • Vessel cost • Vessel maintenance (and/or depreciation schedule) • Site- and time-specific coral survivorship • Ecosystem-scale metrics e,g., coral cover, population structure, restored area
Accessing coral out-plant material	• Source of coral material for planting • Distance between source and out-plant site	• Sites where CoO are readily available – often in areas with high existing cover of naturally fragmenting species may require less time for material collection. • CoO in more degraded areas may require more time pruning to ensure out-plant quality. • Pruning fragments from nursery-propagated colonies can reduce coral material collection time. • Material pruned from colonies propagated in nurseries requires transport to out-plant site (and ensure nurseries are well maintained). Transport needs – swimming or boat – increase over time as outplanting extent increases..	• Proportion of dive time spent propagating, harvesting, preparing and transporting (*in situ*) material *vs* planting • Vessel and fuel cost if *ex situ* transportation required • Dive labour and equipment costs
Nursery propagation and maintenance	• Nursery cleaning (coral health) • Time spent sourcing donor material for stocking nurseries	• Absence of beneficial fish communities that facilitate removal of biofouling algae and invertebrates on nursery structures will require greater time nursery cleaning.	• Proportion of dive time spent (i) sourcing stock material for nurseries, and (ii) cleaning and maintaining (including stock inventories and tracking) nurseries *vs* planting • Dive labour and equipment costs

		• Presence of corallivores may transmit disease and necessitate additional protective structures or continual stocking. • Maximising genetic, taxonomic and functional diversity (cultivation of rare /ecologically important species) requires time identifying and tracking.	
Out-planting	• Availability of bare substrate for planting • Planting experience level (secure attachment and speed).	• Decrease over time – and hence more time required to find – as planting footprint expands. • Likely to increase over time with experience. • Volunteers reduce costs but require more training time and may result in slower (and less effective) planting if sporadic, requiring more maintenance	• Dive labour and equipment costs • Proportion of dive time (labour) spent auditing out-plants (monitoring) *vs* planting
Monitoring, site maintenance	• Control of corallivores e.g., Crown-of-Thorns starfish (COTs), *Drupella spp.* etc. • Other coral "gardening" (overturning coral, wedging fragments, maintaining or replacing out-plants etc.) • Monitoring outplant areas, research trials, data capture.	• Necessitates time away from planting, however critical for improving outplant survival and overall site health. • Corals 're-planted' via wedging, overturning - or maintaining existing out-plant (e.g., remove failed or refill empty Coralclip®) are often part of the workflow but not reported. • Record-keeping, reporting and monitoring can divert time away from planting however are critical to knowledge generation that leads to adaptive practice, and potential cost-savings (e.g., PC_R).	• Proportion of dive time (labour) spent on "other" gardening and site maintenance activities *vs* planting • Time (labour) and materials spent on post-outplanting record keeping, research and data analysis

2.5.2 Survivorship of planted corals across all sites via roving surveys.

Visual-based roving surveys have been used to resolve the effectiveness of Coralclip® (see Suggett et al., 2020) but outplants are often difficult to locate after 12-18 months as Coralclip® devices become overgrown or corrode. Combining Coralclip® metal detector observations with visual-based roving surveys across reefs (n = 36) identified an additional 842 planted colonies that would not have been accounted for through visual surveys alone (Table S2.3). Overall, mean coral outplant survivorship (%, mean ± SE) was 76.6 ± 1.5%, ranging from 55.4 to 93.3% for any given survey (Fig. 2.3A). However, across all reefs % corals 'alive' was higher for Opal Reef and Moore Reef (81-88%) than for all other reefs (68-72%) (Fig. 2.3A; Tukey's post-hoc, $p_{Tukey} < 0.05$; Table S2.8 & S2.9). Such differences appear to be largely reflected by reef-specific differences in the proportion of corals missing ('empty Coralclip®'; Fig. 2.3B) rather than dead corals (Fig. 2.3C). Specifically, Opal Reef and Moore Reef exhibited lower % corals missing (9-11%) than for Mackay Reef and Upolu Reef (both 23%, Fig. 2.3B) (Tukey's post-hoc, $p_{Tukey} <0.05$; Tables S2.8 & S2.9). In contrast, % corals dead was the same (3-6%) for all reefs except Hastings Reef (10.5%; Fig. 2.3C) (Tukey's post hoc, $p_{Tukey} <0.05$, Table S2.8 & S2.9). Therefore, of all reef sites examined, only Hastings Reef exhibited overall lower survivorship driven by a higher contribution of dead corals (and lower contribution of missing corals). For Hastings and Opal Reef, where more than one site was surveyed, no within-reef differences were found in coral survivorship, detachment, or mortality (Tukey's post hoc, $p_{Tukey} > 0.05$, Tables S2.10-S2.12).

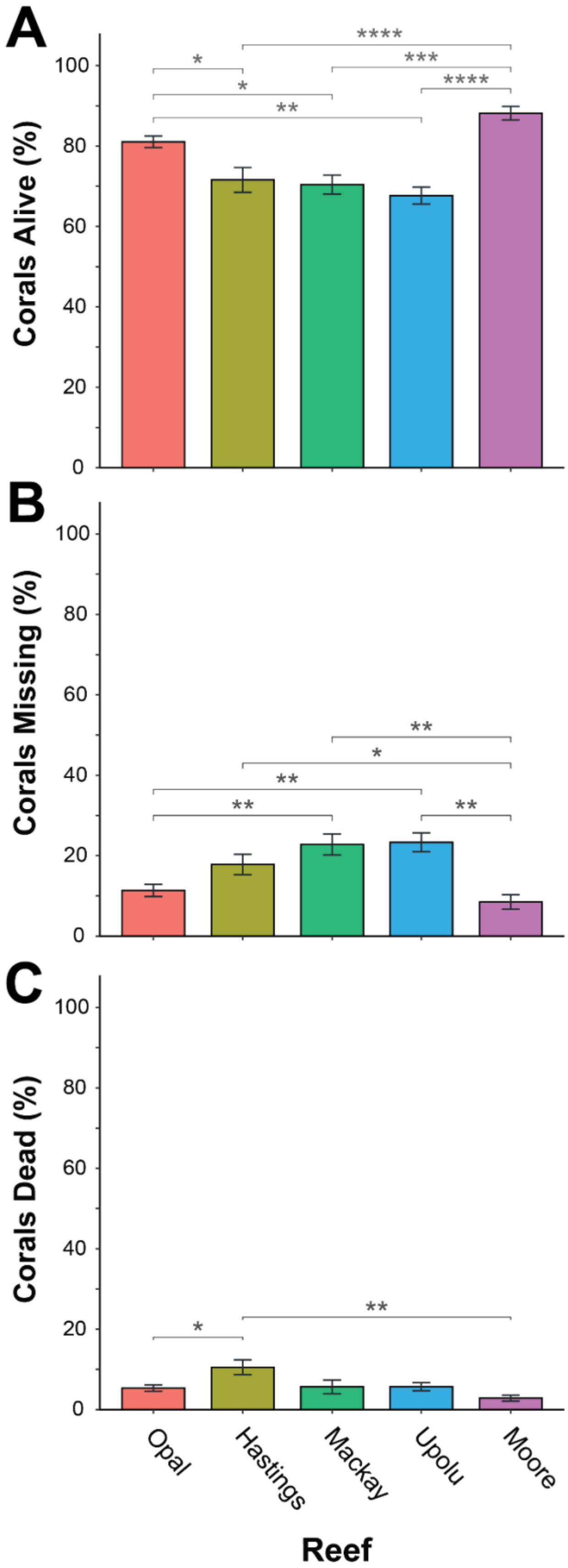

Figure 2.3. The mean (± SE) proportion of Coralip® observations with coral fragments alive (**Panel A**), missing (Coralip® empty, **Panel B**), and dead (dead coral still attached, **Panel C**) from replicate timed-swim surveys of outplanted reef areas at Opal Reef (n = 14), Hastings Reef (n = 8), Mackay Reef (n = 5), Upolu Reef (n = 5) and Moore Reef (n = 5), using combined visual and metal detector surveys. Bar graphs display untransformed proportion data. Horizontal bars and asterisks represent significant Tukey's post-hoc comparisons ($p_{adj} < 0.05$) between Reef groups after conducting separate one-way ANOVA tests on arc-sin transformed proportions for each survivorship category (alive, missing, dead) (Tables S2.9 & S2.10). P-values represented by asterisks as follows: * < 0.05, ** <0.01, *** < 0.001 and **** < 0.0001.

2.5.3 Survivorship of planted corals over time via fate-tracked plots

Whilst roving surveys enabled evaluation of outplant survivorship in haphazardly planted areas, fate-tracked plots enabled tracking of Coralclip® effectiveness across the early stages of outplant establishment. At Hastings Reef (site "1770") mean coral survivorship (%, mean ± SE, n = 3) was consistent during the early stages of post-outplant establishment (95.3 ± 0.7% at T14 days and 93.7 ± 0.7% at T54 days) (Fig. 2.4A). During this period, the proportion of outplanted corals that died or detached from Coralclip® remained low and constant (0.3% or 4-6%, respectively). All categories changed after four months (T151 days) and 12 months (T379 days) with declines in survivorship (87.1 ± 3.6 %, 70.2 ± 2.7%) and increases in both dead corals (3.3 ± 2.0%, 12.5 ± 4.8%) and corals missing (8.25 ± 0.6%, 17.3 ± 5.7%) after 151 and 379 days, respectively (Fig. 2.4B & 2.4C) (Tukey's post hoc, $p_{Tukey} < 0.05$; Table S2.13 & S2.14); however, of these changes, the increase in corals missing (i.e. "empty Coralclip® devices") was not significant (Fig. 2.4C) (Tukey's post-hoc, $p_{Tukey} > 0.05$, Tables S2.13 & S2.14).

A similar pattern was observed at Mackay Reef (site "Angels") as for Hastings Reef (site "1770"). Specifically, mean coral survivorship (%, mean ± SE, n = 3) decreased from 96.7 ± 0.9% (T14 days) to 72.6 ± 4.6 % and 58.5 ± 8.2% (T154 days/5 months and T261 days/9 months, respectively) (Fig. 2.4D) (Tukey's post-hoc, $p_{Tukey} < 0.05$, Table S2.15 & S2.16). These declines were driven by increases in the proportion of corals missing, from 2.8 ± 1.4% (T14) to 16.4 ± 5.0% and 21.3 ± 3.3% (T154, T261), and dead corals, from 0.4 ± 0.5% (T14) to 11.0 ± 2.0% and 20.2 ± 10% (T154, T261; Fig. 2.4E & 2.4F) (Tukey's post-hoc, $p_{Tukey} <$ 0.05, Tables S2.15 & S2.16). Thus, at both sites, changes in survivorship were evident from ~120-150 days (3-4 months) from initial outplanting.

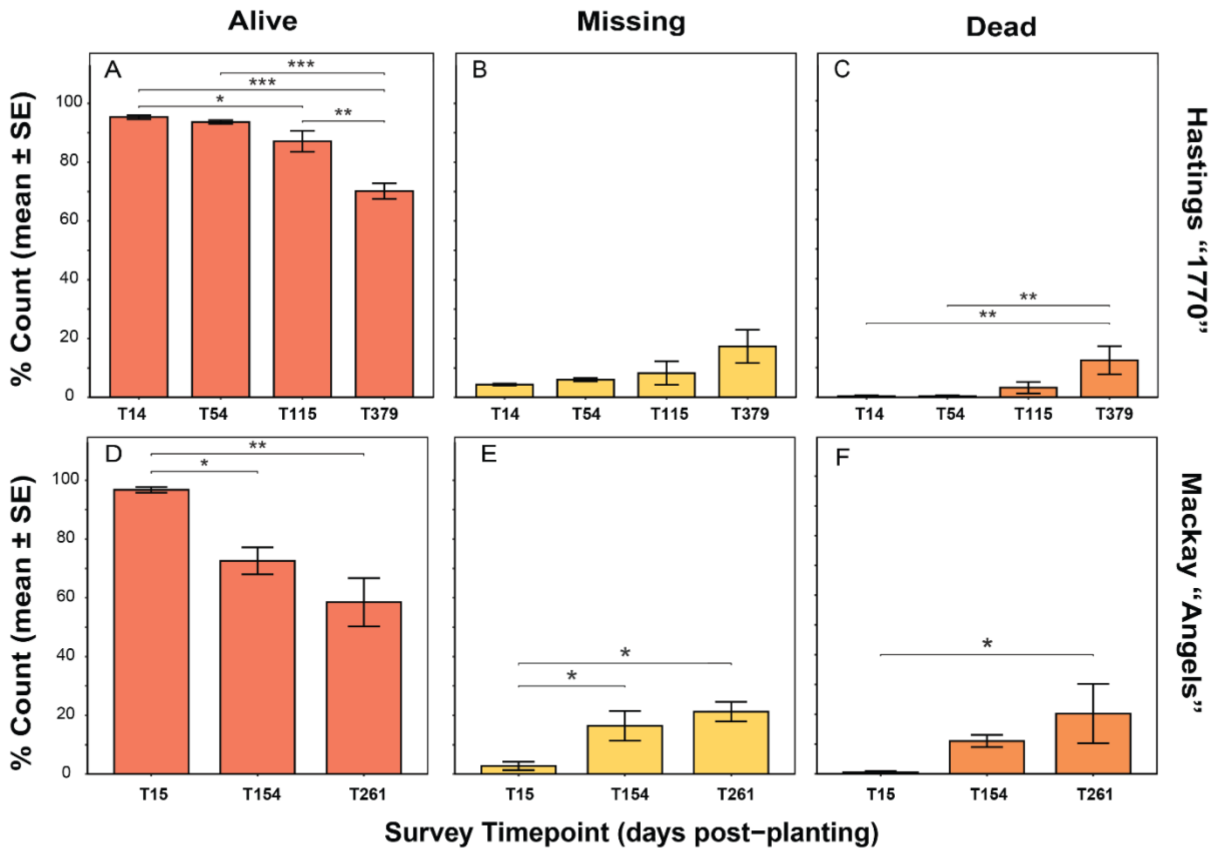

Figure 2.4. Mean (± SE) proportion of Coralclip® observations with coral fragments alive (left, panels A,D), missing (Coralip® empty, middle, panels B,E), and dead (dead coral still attached, right, panels C,F) from triplicate 4x10m^2 plots outplanted in September 2021 at "1770" Hastings Reef (16°31.3′S 146°0.45′E) (panels A-C) and "Angels" at Mackay Reef (16°2.8′S 145°38.8′E) (panels, D-F). Bar graphs display untransformed proportions (%) data. Horizontal bars and asterisks represent significant Tukey's post-hoc comparisons ($p_{adj} <$ 0.05) between timepoints after conducting separate one-way ANOVA tests on arc-sin transformed proportions for each survivorship category (alive, missing, dead) (top row: Table S2.14 & bottom row: Table S2.16). P-values represented by asterisks as follows: * < 0.05, ** <0.01, *** < 0.001 and **** < 0.0001.

2.5.4 Realised cost of coral planting.

Accounting for mean survivorship for each respective reef (Fig. 2.3A) resulted in realised planting costs (PC_R, \$ coral^{-1} trip^{-1}) that were higher by 25-35% compared to the original planting cost (PC, \$ coral^{-1} trip^{-1}; Table 2.2; Fig. 2.2B & 2.2C) across all operational contexts (Table 2.4). On "Routine Planting Days", mean PC_R was higher by \$0.64 compared to PC to \$2.94 ± 0.23 coral^{-1} trip^{-1}, ranging between US\$1.00-7.25 coral^{-1} trip^{-1} (5-95% percentile, Table 2.4). When trips were separated by reef, mean PC_R ranged from \$1.91 ± 0.13 to \$8.14 ± 1.33 coral^{-1} trip^{-1}, increasing PC values by 13 – 48% across reefs (Fig. S2.4) relative to reef-specific outplant survivorship (Fig. 2.3A). Mean PC_R across other operational contexts remained greater than those for "Routine Planting Days", with PC_R higher than PC by

US\$1.45 at US\$7.24 ± 0.17 coral^{-1} trip^{-1} on "Propagation and Maintenance Days"; by US\$5.01 on "Training Days" to US\$21.23 ± 6.13 coral^{-1} trip^{-1}; and by US\$1.48 on "Dedicated Planting Days" (with vessel costs) to US\$5.90 ± 0.64 coral^{-1} trip^{-1} (Table 2.4, Fig 2.2C).

Table 2.4. Summary statistics for "realised" costs of coral outplanting (PC_R, cost per estimated surviving coral per trip (\$US coral^{-1} trip^{-1})) for the different operational contexts of Coral Nurture Program outplanting trips between August 2018 and December 2021. PC_R accounts for the mean survivorship of outplanted coral material derived from roving surveys at respective Reefs (Fig. 2.3A). The final column is the difference between mean realised cost (PC_R) and mean planting cost (PC, Table 2.2). Lower and Upper range values represent the 5th and 9th percentiles.

	PC_R (US\$ coral^{-1} trip^{-1})			
Operational Context (n trips)	**Mean (± SE)**	**Lower Range**	**Upper Range**	**Difference in mean cost (PC_R – PC) (\$US, %)**
(a) Routine Planting Day (110)	2.94 (0.23)	1.00	7.25	0.64, 27.83%
(b) Propagation and Maintenance (30)	7.24 (0.17)	2.52	15.52	1.45, 25.04%
(c)Training Day (6)	21.23 (6.13)	4.39	39.18	5.01, 30.88%
(d) Dedicated Planting Day (Vessel cost included) (8)	5.90 (0.64)	3.52	8.38	1.48., 33.48%

PC_R was similarly calculated for the fate-tracked plots examining coral outplant survivorship over time at both site "1770" at Hastings Reef and "Angels" site at Mackay Reef (Fig. 2.5). Here planting costs (PC) were US\$2.53 and \$1.76 coral^{-1} trip^{-1}, respectively. Outplanting at Hastings Reef occurred over two days, and hence resulted in higher time investment (FTE, Table 2.1) and thus PC. As a result of the decline in survivorship over time at both sites (Fig. 2.4), PC_R for Hastings Reef increased to US\$3.60 coral^{-1} trip^{-1} after 379 days from outplanting (12 months establishment on the reef), representing an increase of US\$1.07 (42%) from the initial PC at T0 (Fig. 2.5). As declines in outplant survivorship were similarly documented at plots at Mackay Reef (Fig. 2.4D), PC_R at 261 days from outplanting was US\$3.01 coral^{-1} trip^{-1}, and hence increased by 71% (~\$1.25) from the initial PC (T0) after 9 months (Fig. 2.5). As such, accounting for the time-dependent nature of survivorship is clearly critical to more accurately resolving realised planting costs.

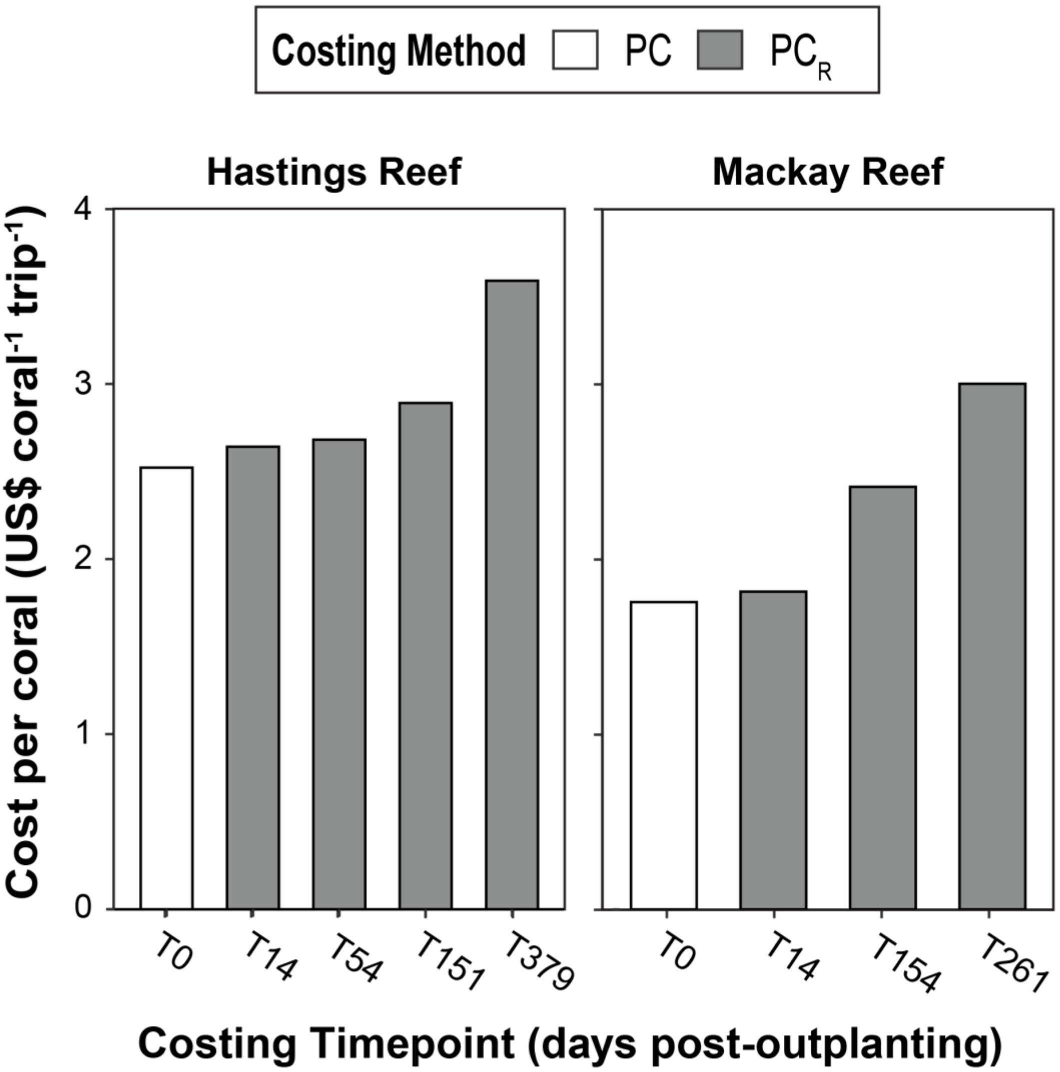

Figure 2.5. Cost ($US) over time for coral fragments (n = 300) outplanted in fate-tracked plots in September 2021 at Site "1770" at Hastings Reef and "Angels" at Mackay Reef. White bars represent planting cost for each respective deployment (PC ($ coral^{-1} trip^{-1}), Equation 1), where 100% of corals are alive at planting (T0). Grey bars show the realised cost (PC_R, ($ coral^{-1} trip^{-1}), Equation 3) of surviving corals in these plots at respective survey timepoints across 9-12 months at each site.

2.6.0 Discussion

Asexual-based coral propagation approaches have increasingly grown in technical and biological feasibility for reef restoration (Rinkevich, 2019; Boström-Einarsson et al., 2020. However, the costs of interventions and the factors underpinning these costs have been sparsely documented alongside outcomes, thereby limiting evaluation of their viability as cost-effective reef management aids for ongoing and future implementation (Bayraktarov et al., 2019). Here we discuss factors influencing costs of coral outplanting activity under the CNP tourism-led targeted site restoration approach and identify several core steps needed to better establish a life cycle costing framework for informing investment, management, and practitioner decisions in sustaining or initiating activity.

2.6.1 Planting cost considerations under a tourism-stewardship model.

CNP was originally conceived as a site stewardship and restoration model integrated into existing tourism day trips (Howlett et al., 2022). As expected, planting costs were therefore lowest on "Routine Planting Days" where CNP activity focused on outplanting, and PC was less than US$3 coral^{-1} for 80% of trips (representing 28,554 corals). This suggests that previous cost estimates for tourism-integrated outplanting activity at Opal Reef (Suggett et al., 2020; US$0.60 to $3.00 coral^{-1}) were generally representative of the costs we observed here for more diverse CNP reef systems and operations. We note that the costs for Opal Reef by Suggett et al. (2020) (n= 4,580; Aug 2018-May 2019) are integrated into – but only represent <20% of – our costs considered here (Aug 2018-Dec 2021, US$1.55 ± 0.1 coral^{-1} trip^{-1}, n = 22,445). Within our costs, CNP staff wages account for ~80% of overall trip costs, and as such, PC was predominantly moderated by staff costs and planting output. This reaffirms the need for effective operational models (e.g., absorption of expensive vessel running costs) and cost-effective and user-friendly coral attachment methods for scaling of coral restoration efforts (Suggett et al., 2020; Vardi et al., 2021).

Reports of coral restoration costs via propagation and outplanting to date are few but range from US$10,000 to $1.5 million/ha (Bayraktarov et al., 2019; 2020). Other programs have specifically reported costs that vary by an order of magnitude lower than (e.g., US$0.15-0.36 coral^{-1}, Philippines (de la Cruz et al., 2014)), higher than (e.g., US$33.40 coral^{-1}, Seychelles (Montoya-Maya et al., 2016, reported in Bayraktarov et al., 2019)) or similar to ($US5.30 coral^{-1} outplanted, Philippines (Villanueva et al., 2012)) the mean PC determined for "Routine Planting Days" for CNP here (US$2.30 coral^{-1}trip^{-1}). However, direct cost comparison between studies remains challenging and, in some cases, not appropriate where differences are governed by location-specific restoration contexts, and logistical and socioeconomic factors; for example, labour costs of $13-28 day^{-1} in the Philippines (de la Cruz et al., 2014; Baria-Rodriguez et al., 2019; Harrison et al., 2021) compared to >$200 day^{-}

[1] on the GBR in Australia. Importantly, cost differences also reflect use of alternate restoration methods (which again may be location or context specific), degree of volunteer involvement, scales, timeframes, and cost-accounting across projects (Bayraktarov et al., 2019). Costs for coral deployment, although often carrying the highest expense (ca. 30-50% of project costs; Edwards et al., 2010; Toh et al., 2014; Humanes et al., 2021) are rarely the only activities involved in reef restoration. As such, higher cost estimates may result from differences in cost-accounting across the 'whole life' of interventions (Spurgeon & Lindahl, 2000). Hence, we also considered outplanting cost estimates to account for other modes of operation essential to site stewardship under CNP activity.

"Propagation and Maintenance Days" and "Training Days" were less frequent than "Routine Planting Days", and unsurprisingly mean PC was increased by 2- to 7-fold since staff time costs did not always result in corals outplanted. Such an outcome is consistent with other coral propagation projects employing intermediate nursery propagation phases, owing to added capital costs for nursery materials and labour requirements for cleaning and maintenance (e.g., Shafir & Rinkevich, 2010; Montano et al., 2022). Whilst nursery propagation and maintenance move focus from outplanting, coral nurseries provide readily available (Böstrom-Einarsson et al., 2020; Howlett et al., 2022, 2023) and selected-for coral stock (Baums et al., 2019; Shaver et al., 2022), thereby reducing time required for coral material collection on outplanting days. Nurseries importantly serve as visually appealing demonstration sites for educating visitors on reef stewardship activity (Howlett et al., 2022) and hence are necessary for overall project life-cycle investment for the CNP operational approach.

"Training Days" were the costliest operational context, but also are conducted most infrequently. Capacity-building reef industry-stakeholders is foundational to the CNP "learn by doing" approach (*sensu* Quigley et al., 2022; also, Howlett et al., 2022), which in turn is the critical step to improving planting efficiency (Suggett et al., 2020), and hence PO that regulates PC. In other reef restoration programs, costs of capacity-building reef stakeholders through training are unclear, yet undoubtedly deliver immense benefit for reef-dependent communities via enhanced employment opportunities, income diversification and community education (e.g., projects in the Caribbean and Eastern Tropical Pacific, Israel, and the Seychelles (Bayraktarov et al., 2020; Vaughan, 2021)). Indeed, for several tourism operators in the CNP, such capacity provided industry resilience during tourism downturns where tourism operators received funding for site stewardship activities, including restoration, to retain industry assets (Howlett et al., 2022, Suggett et al., 2023). Furthermore, the near doubling of mean PC by including vessel charter costs on CNP "Dedicated Days" demonstrates how cost-effective outplanting on the GBR – as with other restoration programs globally (e.g., dela Cruz et al., 2014; Toh et al., 2017; Bayraktarov et al., 2020) – hinges upon stakeholder involvement. Capacity-building and stewardship are key success indicators of coral restoration (Hein et al. 2017), and integral to the longer-term sustainability of local restoration efforts (Hein et al., 2020; Quigley et al., 2022), and hence costs of training would appear logical to consider in life-cycle costings.

Resolving discrete time- and cost-tracking of the individual stages in site restoration (e.g., as per Edwards et al., 2010) was not possible here, but clearly remain an important means to guide improved operational cost-effectiveness in future. It is important to reiterate that the data captured through CNP reporting forms – and used to examine costs here – were largely opportunistic of the requirement to report core outplanting and nursery activities for permitting. Such opportunistic cost-tracking often precluded differentiation of staff time to non-outplanting activity for several trips, necessitating exclusion from this cost-analysis. Stakeholder-led restoration projects are often not set up initially to capture critical cost attributes, or incentivised to report them in scientific literature (Bayraktarov et al., 2015, 2020), as depth of data recording and reporting presents a time-cost trade-off to outplanting effort, and funding is often governed – or indeed program success measured – by simple activity metrics such as numbers of coral outplanted (Hein & Staub, 2021; Suggett et al., 2023). In the case of CNP operations, time-cost trade-offs between data reporting and activity are governed by tourism schedules (Howlett et al., 2022). Thus, resolving greater accuracy of PC estimates, and indeed the full life-cycle costs of the processes underpinning successful restoration, requires more rigorous documentation of staff time – the greatest cost under the CNP approach – across outplanting and non-outplanting activities.

2.6.2 Survivorship of outplanted corals and 'realised' costs.

Whilst coral propagule survivorship is broadly acknowledged as an insufficient metric to describe overall project feasibility and socioecological effectiveness (Bayraktarov et al., 2015; Hein et al., 2017; Boström-Einarsson et al. 2020), it provides a useful means to benchmark and compare restoration costs in terms of new biomass returned to the reef (Edwards et al., 2010). Overall, mean coral outplant survivorship observed across all reefs through roving surveys (76.6%) was higher than the mean value (~65%) reported previously from >30 coral outplanting projects (Boström-Einarsson et al., 2020). Resolving the factors contributing to variable survivorship across surveyed reefs (ranging 68 – 88%) was beyond the scope of our study; however, differences appeared to be largely driven by fragment dislodgement from Coralclip® (mean 15.6% observations), rather than post-attachment mortality (mean 6.2% observations). Such dislodgement-related mortality likely occurred in the first 3 months post-outplanting, as branching coral fragments have been shown to self-attach to substrates between 40-60 days depending upon species (Guest et al., 2009; Howlett et al., 2022). Coralclip® devices may lose coral before fragments can self-attach owing to insecure attachment or coral 'knock-out' from fish feeding (J. Edmondson, pers. Obs.; see also Frias-Torres et al., 2015; Horoszowski-Fridman et al., 2015), as well as potentially higher predation by corallivores (Seraphim et al., 2020). Empty devices can remain visible for several months before they begin to corrode or are overgrown or replaced through maintenance outplanting, and thus such roving visual surveys are more suited to capture early survivorship estimates (Suggett et al., 2020).

Corals that do attach to the substrate inevitably overgrow Coralclip® devices and do not always remain visible after 1-2 years of growth, thus confounding estimates of later-stage survivorship. However, we employed a novel metal detector survey approach that could detect the Coralclip® nail, and indeed demonstrated improved identification of coral outplants by 5-54% compared to visual-based surveys alone. Given that nails are often used for attaching coral outplants or settlement plugs (reviewed in Suggett et al., 2020), the use of the metal detector can support monitoring efforts where material is planted haphazardly and not well mapped for re-assessment over time.

Fate-tracked plots were further used to improve our survivorship estimates. At 9 months post-outplanting, mean survivorship of coral outplants at Mackay Reef declined to 58.5%, yet after 12 months was 70.2% at Hastings Reef. Other studies have similarly documented outplant mortality rates of ca. 30-40% in the first year (Schopmeyer et al., 2017; Morand et al., 2022), although survivorship as low as <5% has been recorded ((Baria-Rodriguez et al., 2019; Ware et al., 2020). Indeed, longer-term outplant survivorship is known to vary substantially (e.g.,

<20% in Japan (4 years); Okubo, 2023; 9% in the Virgin Islands (12 years); Garrison & Ward, 2012; versus 82% after 13 years in Belize; Carne & Trotz, 2021). In our study, increased mortality between the penultimate and final timepoints at both sites potentially reflected enhanced stress from an anomalous heating event in March 2022, where severe bleaching (>60% of colonies) was documented across the central-northern GBR (GBRMPA, AIMS & CSIRO, 2022), though little mortality was observed elsewhere across CNP reef sites (T.Agius, J. Edmondson, S.Edmondson, P.Coulthard, R.Hosp, personal observation, April 2022).

Mean 'realised' planting costs (PC_R) increased by 25-35% compared to PC across all operational contexts. Such an outcome is consistent with other studies, with realised costs increasing by several orders of magnitude (e.g., 13-fold; Baria-Rodriguez et al., 2019), where variable survivorship of coral propagules was accounted for. Fate-tracked coral outplants in plots resulted in time-dependent survivorship (PC_R increasing by 42-71% after 1 year) and thus, realised costs are inevitably time-bound to when 'effectiveness' is evaluated (Edwards et al., 2010; Baria-Rodriguez et al., 2019). For example, in a larval enhancement project, Harrison et al. (2021) documented a ca. 40% increase in realised costs over time owing to mortality, from US\$13.73 coral^{-1} at 10 months to US\$17.79 coral^{-1} at 34 months. Costings are further confounded in circumstances where propagules reach reproductive maturity and result in self-generation of further biomass to the reef (e.g., Harrison et al., 2021) or indeed mass mortality events that may occur after costs are reported (e.g., Fadli et al., 2012). Longer-term fate-tracking is therefore clearly warranted but inevitably entails higher monitoring costs, thus highlighting the need to resolve cost-benefit trade-offs that enable practitioners to optimise restoration approaches. For example, Humanes et al. (2021) and Baria-Rodriguez et al. (2019) determined that extending coral nursery rearing periods for sexual recruits resulted in enhanced survivorship over the long-term, thereby negating any additional costs associated with longer husbandry periods. In the context of the CNP, although typically more expensive, "Training days" and "Dedicated days" are not only critical to evaluating efforts, trialling new techniques, training staff, and improving practice, but also concentrating efforts at degraded sites where coral population recovery is most needed (Howlett et al., 2023; Roper et al., 2022).

Ultimately, the realised costs of restoration efforts are dynamic over space and time and are highly dependent upon how ‘effectiveness’ is defined and captured, and for how long. For example, employing ecological changes (e.g., live coral volume, Morand et al., 2021; or population structure, Roper et al., 2022), ecosystem service values (Abrina & Bennett, 2021) or socioeconomic benefits (Hein et al., 2017, 2019) in cost evaluations rather than outplant survivorship, would likely deliver vastly different, and arguably more informative (Suggett et al., 2023) assessments of cost-effectiveness. Furthermore, as climate-driven disturbances increasingly drive coral mortality, measures of ecological and social resilience will become essential to justify return-on-investment in cost-benefit analyses (Shaver et al., 2022). As enthusiasm grows to invest into coral restoration, it is increasingly time-sensitive to resolve a framework for transparent costings that can be adopted across stakeholder-led restoration programs. Our approach identifies how opportunistic reporting can be exploited to examine costs, and in turn identify factors (e.g., staff time reporting, longer monitoring periods) needed to improve cost data capture to further improve resource allocation within restoration practice. We have presented the first reports for costs associated with tourism-led restoration of high value GBR sites, and the inherently variable nature of cost-effectiveness across highly diverse operations and environments.

2.7 Acknowledgements

We wish to express immense thanks to the Great Barrier Reef Marine Park Authority (GBRMPA), whose support established the permit for the out-planting activity and coral nursery deployment (G19/42553.1), as well as owners and/or staff from Sailaway Port Douglas, Wavelength Reef Cruises, Passions of Paradise, Ocean Freedom and Great Adventures (notably Katrina Edmondson, Alan Wallish, Scott Garden, Perry Jones, Scott Daniels and Dougie Baird), who have been an essential part of the Coral Nurture Program operations, data collection and research support through immense in-kind contributions throughout. This research is supported by an Australian Government Research Training Program (RTP) Scholarship. All activity costed here was supported by essential funding from the Australian & Queensland Governments (“Solving the bottleneck of reef rehabilitation through boosting coral abundance: Miniaturising and mechanising coral out-planting” to D.J.S., E.F.C., and J.E.) (2018-2019), and the partnership between the Australian Government’s Reef Trust and Great Barrier Reef Foundation and by DIAGEO Reeftip Drinks Co. (to D.J.S., E.F.C. and J.E.) (from 2021 onwards). Funding activity from 2021 onwards also enabled opportunistic data collection on survivorship reported here by RS. E.F.C. was also supported by a University of Technology Sydney Chancellor's Postdoctoral Research Fellowship and an ARC Discovery Early Career Researcher Award (DE190100142). The authors declare no conflicts of interest or competing interests.

2.8.0 Supplementary Information

2.8.1 Methods Additional Information

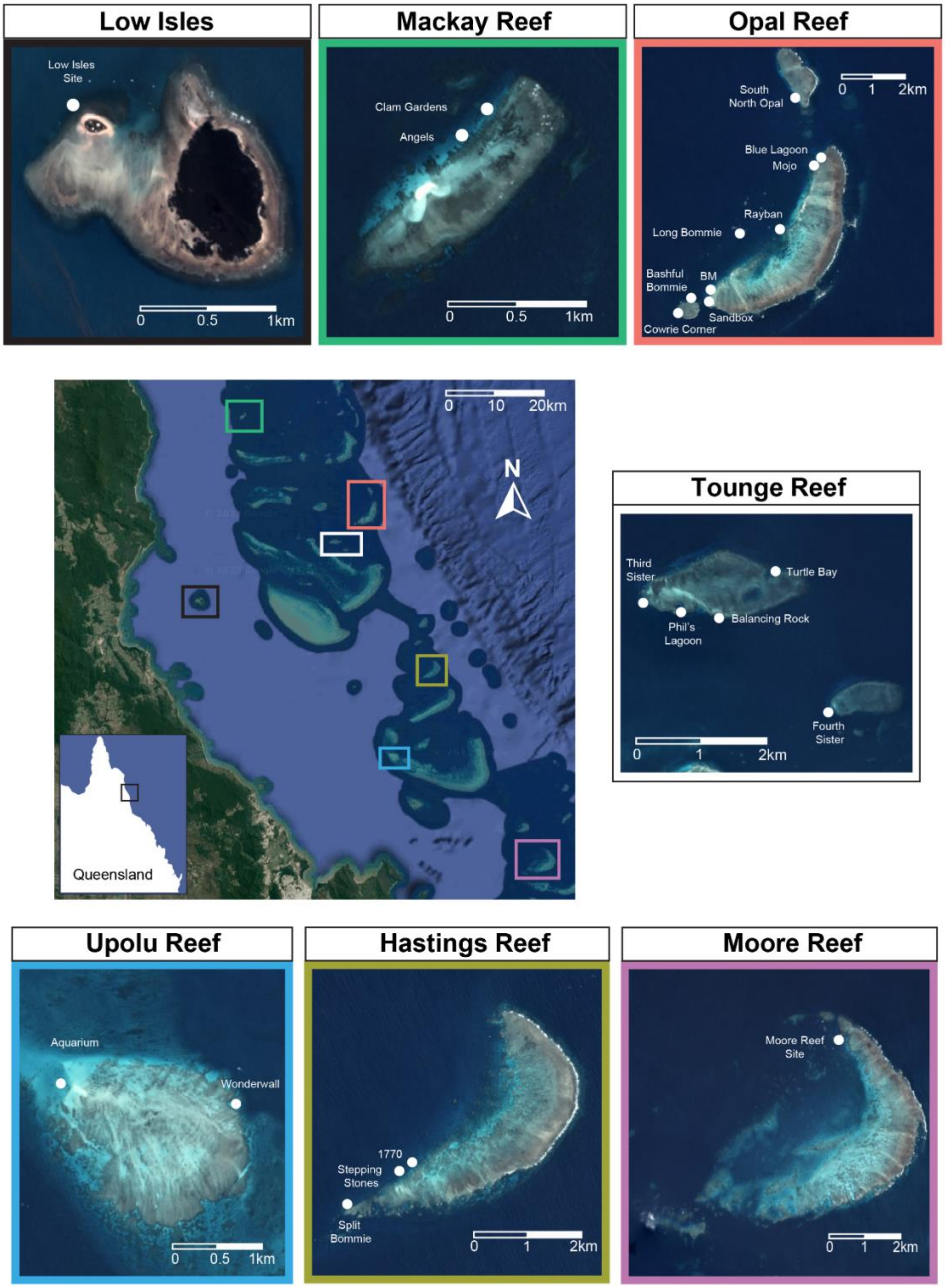

Figure S2.1 - Map showing the locations of all 23 Coral Nurture Program outplanting sites on 7 Reefs within the Cairns-Port Douglas region of the Northern Great Barrier Reef, Australia: Opal Reef (16°13′S 145°53.5′E, red square) “Blue Lagoon”, “Mojo”, “RayBan”, “Beautiful Mooring (BM)”, “Bashful Bommie”, “Long Bommie”, “Cowrie Corner”, “Sandbox”, “South-North Opal”; Tongue Reef (16°16’51.2”S 145°49’11.7”E, white square) “Turtle Bay”, “Phil’s Lagoon, “Third Sister”, “Fourth Sister, “Balancing Rock”; Hastings Reef (16°31.3′S 146°0.45′E, yellow square) “1770”, “Stepping Stones”, Split Bommie”; Mackay Reef (16°2.8′S 145°38.8′E, green square) “Angels”, “Clam Gardens”; Low Isles (16◦23.2′S 145◦33.8′E, black square “Low Isles Site”; Upolu Reef (16°40.6′S 145°56.3′E, blue square) “Wonderwall”, “Aquarium”; Moore Reef (16°52.5′S 146°14.0′E, purple square) “Moore Reef Site Pontoon” (see also Howlett et al, 2022). Satellite image sourced from Google Earth and allencoralatlas.org.

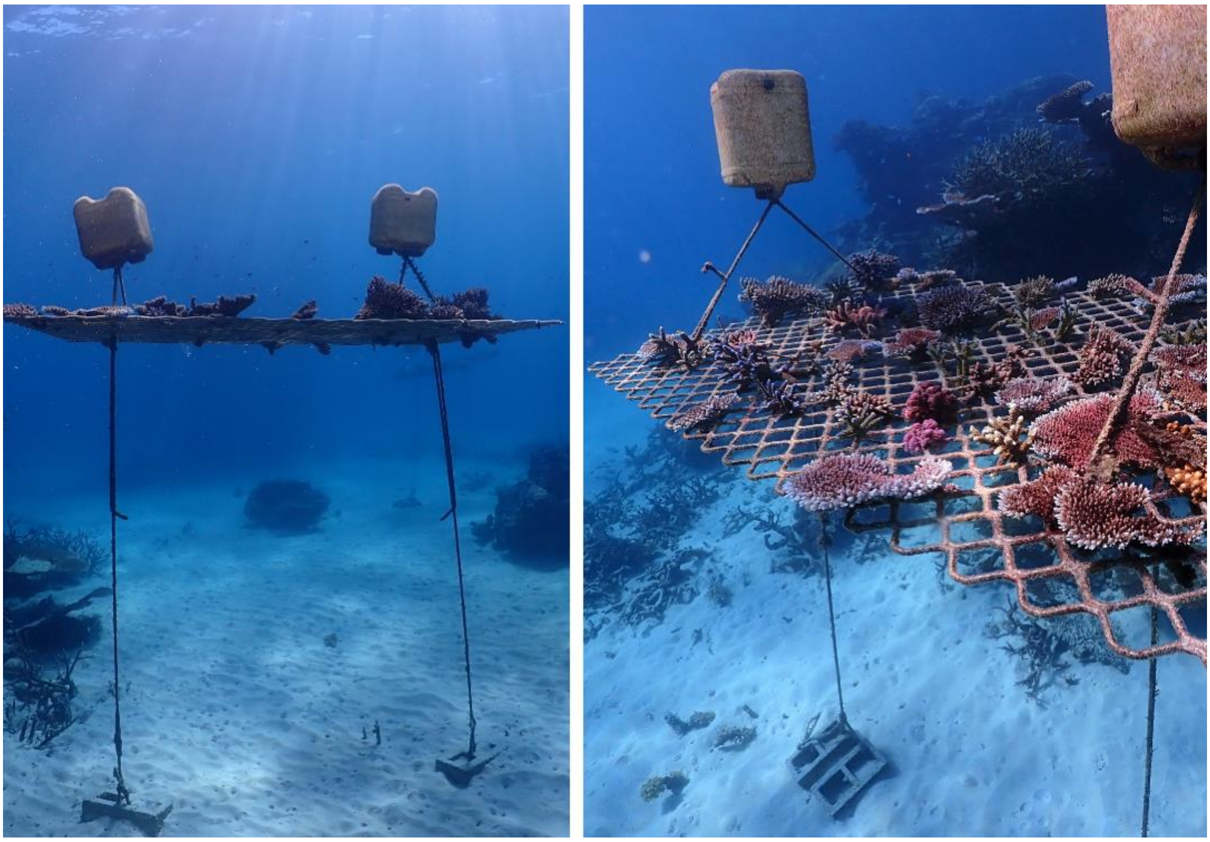

Figure S2.2. Low-cost coral nurseries used across Coral Nurture Program sites, which are typically located above sandy lagoon areas (at depths of 4-8m) adjacent to vessel moorings. Nursery frames consist of a 2.0 x 1.2m sheet of diamond aluminium mesh tethered 1-2m above sandy lagoons by rope to 2 x 9kg concrete besser blocks, and suspended mid-water by two empty 20L plastic drums. The materials cost for each frame is approximately US$60, with a lifespan of >10 years with maintenance of ropes and floats. Each frame takes two divers approximately 1 hour to install, and can hold ca. 150 coral propagules from which coral material can be continuously generated. Photos of nurseries at Opal Reef supplied by J.Edmondson.

Table S2.1. Calculations employed to calculate \$P (planting equipment capital cost) and \$D (diving gear costs) values in planting cost (PC, Eqn 1) and realised cost (PC_R, Eqn 3) equations (see Table 2.1).

(a) Planting Equipment (\$P)		
Item	**Cost (\$AUD)**	**Cost (\$US)**
Hammer	9.00	6.48
Scrubbing brush for preparing substrate	5.00	3.60
Stainless steel wire cutters for fragmenting corals	25.00	18.00
Wire mesh trays for carrying coral fragments and Coralclip®	20.00	14.40
Total	**59.00**	**42.48**
4 sets of gear	236.00	169.92
Cost year^{-1} (Assume 2yr lifespan)	118.00	84.96
Trips per year[a]	33	33
Cost trip^{-1} (\$P)	**3.58**	**2.57**
(b) Diving Equipment (\$D)		
Item	**Cost (\$AUD)**	**Cost (\$US)**
Full set of SCUBA gear (assuming lifespan of 5 years)	1500.00	1080.00
Full annual gear service fee	200.00	144.00
Total gear cost over 5 years (assuming 4x annual services)	2300.00	1656.00
Gear cost day^{-1}	1.26	0.91
Cost of SCUBA tank refill	4.00	2.88
Cost trip^{-1} (\$D)[b] = (daily gear cost + tank*n(sites))		
1 site	**5.26**	**3.79**
2 sites	**9.26**	**6.67**
3 sites	**13.26**	**9.55**

[a]mean number of trips across the 5 CNP operations in 2021.

[b]Per person - assuming 1 dive per site

Table S2.2. Summary of coral outplant survivorship information collected across the Coral Nurture Program to December 2021 through (1) Roving surveys - where survivorship of mixed species is assessed visually by binning Coralclip® observations into "coral alive", "coral dead", "coral missing", and (2) Fate-tracked plots evaluating survivorship for specific species.

Reef (Site)	**Date Survey**	**Species**	**Time after planting**	**% Survivorship Mean (SE), n**
		(i) Roving surveys		
Opal (BM)*	Mar 2019	Predominantly various *Acropora* sp., and *P. damicornis*,	3-7 days	84.8 (1.9), 359
Opal (BM)*	May 2019		4-5 months	97.4 (1.3), 445
Opal (Ray Ban)*	Mar 2019		3-7 days	95.4 (1.2), 903

Opal (Ray Ban)*	May 2019		4-5 months	93.3 (4.5), 435
Hastings (1770)**	Mar 2020		7 months	93.1 (5.22), 261
Hastings (Stepping Stones)**	Mar 2020		7 months	71.1 (12.3), 225
Hastings (1770)***	Aug 2021		6-9 months	63.1 (3.2), 109
Hasting (Stepping Stones) ***	Aug 2021		6-9 months	58.0 (4.5), 122
Mackay (Angels) ***	Aug 2021		6-9 months	72.5 (2.9), 201
Mackay (Clam Gardens) ***	Aug 2021		6-9 months	69.9 (3.3), 132
Moore Reef***	Aug 2021		6-9 months	95.8 (n/a), 144
(ii) Fate-tracked plots for targeted species evaluation				
Opal (Ray Ban)**	Jun 2019	*P. meandrina*	11 months	60.0 (n/a), 20
Opal (Ray Ban)**	Jun 2019	*A. gemnifera*	11 months	100.0 (n/a), 10
Opal (Ray Ban)**	Jun 2019	*A. spathulata*	11 months	70.0 (n/a), 10
Opal (Ray Ban)**	Jun 2019	*A. intermedia*	11 months	70.0 (n/a), 10

**Suggett et al. (2020) Coralclip®: a low-cost solution for rapid and targeted out-planting of coral at scale. Restoration Ecology 28: 289-296.*
*** Howlett et al 2022. Adoption of coral propagation and out-planting via the tourism industry to advance site stewardship of the northern Great Barrier Reef. Ocean & Coastal Management. 255: 106199*
****August 2021 site audits (Markus Mende, CNP subcontractor).*

Table S2.3: Summary of the roving survivorship surveys conducted at each Reef (Fig.S1) with the combined metal detector/visual survey method, describing the site, number of repeat surveys, the average depth (m), average survey time (min), average number of combined metal detector and visual Coralclip® observations (alive, dead, missing), and total number of observations recorded by the visual surveyor and metal detector surveyor. Survey length was determined by planting density until either combined counts of Coralclip® either reached >100, or a maximum survey time of 25 minutes. Depth values were read off a Sunto Zoop Novo Dive computer. Values in brackets are ± standard error.

Reef	Site ('n' replicate surveys)	Mean depth (m)	Mean time per survey (mins)	Mean number of observations per survey	Total observations (Visual)	Total observations (Metal Detector)
Opal Reef	Rayban (4)	4.3 (0.1)	8.3 (1.5)	137.7 (3.8)	363	50
	Mojo (3)	4.5 (0.4)	10.7 (2.2)	133 (17.7)	270	129
	Blue Lagoon (3)	5.6 (0.2)	17 (4.4)	109.3 (2.3)	312	16

	BM (4)	3.8 (0.3)	13.5 (1.9)	135.3 (7.0)	398	143
Hastings Reef	1770 (3)	7.6 (0.2)	16.7 (3.7)	140 (5.3)	378	42
	Stepping Stones (5)	5.0 (0.5)	16 (2.0)	110 (18.7)	457	93
Mackay Reef	Angels (5)	3.4 (0.3)	12.6 (1.4)	181.2 (8.1)	808	98
Upolu Reef	Wonderwall (5)	4.8 (0.1)	16.6 (2.3)	131 (9.7)	426	229
Moore Reef	Pontoon (1.2)	4.6 (2.6)	14.2 (2.8)	102.2 (9.1)	469	42

2.8.2 Roving Surveys & Metal Detector Testing

Methods

Operators have conducted coral planting at sites at various time points and to varying degrees of intensity since mid-2018 (Opal and Mackay reef) and mid-2019 (Moore, Upolu and Hastings reef), and therefore outplants in survey areas ranged from a few months up to >3 years old. As such, we evolved the previous visual-based survey method to include dual assessment via an underwater metal detector with a circular detection "coil" 20 cm in diameter (PulseDive SCUBA, Nokta Metal Detectors, Istanbul Turkey), which both emits a sound and vibrates upon detecting a Coralclip® (Fig. 2.1). At each site, replicate timed swim surveys (n = 3-7, 5-25 minutes each depending on planting density) were conducted in buddy pairs in outplant areas following a path parallel to the reef slope (Table S2.3). One diver conducted visual surveys (only counting clearly visible Coralclip® devices) whilst the second diver (metal detector operator) closely followed the same path with the metal detector, rotating it around all corals with no visible Coralclip® for up to 5 seconds, recording the overall number of corals tested, and the number of planted corals (dead or alive) detected. Metal detector settings were kept consistent across surveys. Initial trials with the metal detector on outplants of known age and size determined the approximate size-detection limit, and thus only colonies with a maximum length of ~40 cm or less were tested (Table S2.5). Larger colonies were either unlikely to be planted corals (due to maximum growth possible in 2-3 years) or were too dense or morphologically complex for Coralclip® to be detected. Additional tests were conducted to determine the false negative detection rate of the metal detector (i.e., failing to detect known outplants) based on colony size and morphology, as well as to ensure that there was no operator bias in sampling effort, given personnel availability during the field campaign precluded consistency in metal detector operator across surveys. These tests are described below.

Experiment 1: Testing false 'negative' detection rate of the metal detector.

All roving surveys were conducted with the same metal detector (set to Dive Mode, detection level 5, with sound and vibrate ON before entering the water). Dive mode effectively ‘locks’ the control buttons, preventing metal detector settings from being accidentally changed by water pressure and from detecting metal when held upside down. During surveys, it quickly became apparent that the large size and shape of the 20cm circular metal detector surface made it difficult to access the base of the coral colony where the Coralclip® was located (as was often required to “detect” a planted coral). This issue was particularly pronounced for large branching morphologies or densely packed tabular or corymbose colonies.

To therefore assess the rate of “failed” detection, the metal detector employed in the roving survivorship surveys (Metal Detector A) was compared with a second metal detector with a shorter 9cm pointer detection coil using the same settings and frequency (Metal Detector B - PulseDive SCUBA, Nokta Metal Detectors, Istanbul Turkey). The compact shape of Metal Detector B meant it could be inserted between the narrow gaps in branching corals or underneath tabular and corymbose colonies to access their base where the Coralclip® attached them to the substrate. Colonies (n = 67) of varying sizes and morphologies were first tested by one diver with Metal Detector B to confirm the presence of the Coralclip®. The second diver then tested the same colony with Metal Detector A for approximately ~5 seconds, and the colony’s size (maximum length by perpendicular width, measured in-situ with measuring tape) and morphology were recorded. The proportion of successful detection over the total number of colonies tested was calculated for both metal detectors, as well as the detection rate (% over total) for each detector type based on colony morphology (corymbose - any compact branching colony, branching - open, arborescent branching colonies, digitate, plating) and size (small = length or width <20cm, Medium = length or width > 20cm, but <30cm, Large = length or width >30cm, but <40cm, Extra Large = length or width >40cm).

Experiment 2: Testing Metal Detector Operator Bias in sampling effort and detection frequency

To test differences in metal detector sampling effort, roving survivorship surveys (visual + metal detector) were repeated at three sites: “Angels”, Mackay Reef (n = 5), “Rayban”, Opal Reef (n = 3) and “Long Bommie” (n = 3). Opal Reef) on the same planted area with different metal detector operators. The visual surveyor - who recorded planted colonies with visual Coralclip® units - was kept consistent across all surveys, and these observations showed high agreement between repeat surveys. Each metal detector operator recorded the number of planted colonies detected and the total number of colonies tested (i.e., detected and not detected). This was expressed as a proportion (number of colonies detected/overall total colonies tested), and the average proportion across replicates for each survey was compared between metal detector operators. Proportion data was arc-sin transformed, tested for normality (Shapiro-Wilk test) and equal variance (F-test) assumptions, and compared with an unpaired Welch's t-test.

Results

Based on repeated surveys of the same area at three sites with two different metal detector operators, we observed no significant differences in detection rate of planted corals relative to the sampling effort between metal detector operators (Welch's t-test, $p > 0.05$, Table S2.4). The false negative detection rate of the 20 cm circular metal detector utilised in roving surveys (referred to as Metal Detector A, shown in Fig. 2.1) was quantified by comparing detection rates of 67 known planted colonies with a second, shorter 9 cm pointed detection coil (Metal Detector B), which was easier to place near the base of planted colonies where the Coralclip® attaches the colony to the substrate. Overall, the detection rate of Metal Detector A was only 67% of planted colonies compared to 94% by Metal Detector B (Table S2.4 & S2.5); however, roving surveys using Metal Detector A had already been conducted at several sites and therefore we continued to use Metal Detector A for consistency. As such, counts for surviving planted corals derived in roving surveys may be underestimated (where some older and inconspicuous planted corals may not have been accounted for, even with the dual method) and our survivorship extents reported are conservative. On occasion operators conducted 'maintenance planting' where dead coral fragments, or empty, and still viable Coralclip® units were replaced with new FoO (see also Morand et al., 2021). Such activity was not always reported (and thus may potentially inflate survivorship estimates), but highly infrequent from subsequent discussions with operators and thus considered negligible for the purposes of our current costing exercise.

Table S2.4: Results of repeat surveys comparing Coralclip® detection frequency relative to sampling effort between different metal detector operators. Shown are untransformed mean (± SE) metal detector operator detection counts expressed as a proportion of sampling effort (overall number of colonies tested) across replicate surveys at "Rayban", "Long Bommie" (Opal Reef) and "Angels" (Mackay Reef) and the outcome of Welch's unpaired t-tests on arc-sin transformed proportion data between metal detector operators (n = 2).

Site (n replicates)	Metal Detector Operator	Mean proportion (SE) (colonies detected/colonies tested)	Welch's unpaired t-test
Rayban (n=3)	A	0.630 (0.105)	T = 1.684, df = 2.52, p = 0.208
	B	0.434 (0.042)	
Angels (n=5)	C	0.458 (0.089)	T = 0.347, df = 6.492, p = 0.740
	D	0.418 (0.053)	
Long Bommie (n=3)	B	0.450 (0.018)	T = 1.588, df = 2.386, p = 0.233
	E	0.360 (0.053)	

Table S2.5. Coralclip® detection success rates for 67 coral colonies of mixed morphologies and species (morphology denoted in brackets) for metal detector A (20cm circular detection coil, used in the roving survivorship surveys) and Metal Detector B (9cm pointer detection coil). Red cells denote unsuccessful detection of the Coralclip®, and green cells denote successful detection of the Coralclip®.

Metal Detector A	Metal Detector B	Colony genus/species (morphology)	Max length (cm)	Max perpendicular width (cm)	Area (length x width, cm^2)
x	x	*Pocillopora damicornis* (corymbose)	24	22	528
x		*Acropora* sp. *(corymbose)*	24	20	480
x		*Acropora* sp. *(corymbose)*	28	31	868
x		*Acropora* sp. *(corymbose)*	28	20	560
x		*Acropora* sp. *(branching)*	28	22	616
		Acropora sp. *(corymbose)*	33	19	627
		Acropora sp. *(corymbose)*	17	10	170
		Acropora sp. *(plating)*	16	11	176
x		*Acropora* sp. *(branching)*	33	18	594
		Acropora sp. *(branching)*	33	23	759
		Acropora sp. *(plating)*	17.5	14	245

		Acropora loripes (corymbose)	16	18	288
		Acropora gemmifera (digitate)	23	21	483
x		*Acropora loripes (corymbose)*	22	23	506
		Pocillopora damicornis (corymbose)	11	16	176
		Acropora millepora (corymbose)	13	11	143
		Acropora sp. *(plating)*	18	17	306
		Pocillopora verrucosa (corymbose)	23	21	483
		Acropora loripes (corymbose)	19	19	361
		Acropora sp. *(corymbose)*	11	21	231
		Acropora sp. *(corymbose)*	19	20	380
		Acropora millepora (corymbose)	29	19	551
		Pocillopora verrucosa (corymbose)	17	25	425
x		*Acropora* sp. *(plating)*	24	11	264
		Pocillopora verrucosa (corymbose)	17	15	255
	x	*Acropora millepora (corymbose)*	20	19	380
		Acropora millepora (corymbose)	39	43	1677
x		*Acropora millepora (corymbose)*	39	31.5	1228.5
x	x	*Acropora cf. gemmifera (digitate)*	43	37	1591
x		*Acropora* sp. *(plating)*	17	16	272
x		*Pocillopora meandrina (corymbose)*	15	10.5	157.5
x		*Acropora* sp. *(plating)*	11.5	8	92
x		*Acropora* sp. *(corymbose)*	17	13	221
x		*Acropora* sp. *(corymbose)*	13	10	130
		Acropora sp. *(corymbose)*	26	26	676
		Acropora intermedia (branching)	53	41	2173

		Acropora florida (branching)	24	15	360
x		*Acropora intermedia (branching)*	60	53	3180
x		*Acropora* sp. *(corymbose)*	28	23	644
		Acropora sp. *(corymbose)*	26	29	754
		Acropora sp. *(corymbose)*	27	29	783
x		*Acropora intermedia (branching)*	42	44	1848
		Acropora millepora (corymbose)	38	25	950
		Acropora sp. *(branching)*	34	25	850
		Pocillopora meandrina (corymbose)	14	13	182
		Acropora sp. *(corymbose)*	20	17	340
		Acropora sp. *(corymbose)*	29	26	754
		Acropora sp. *(corymbose)*	29	26	754
x	x	*Acropora* sp.*(branching)*	28	18	504
		Pocillopora damicornis	32	21	672
		Acropora (branching) (corymbose)	42	38	1596
		Acropora sp. *(branching)*	37	26	962
		Acropora sp. *(corymbose)*	30	23	690
		Pocillopora damicornis (corymbose)	28	23	644
		Acropora sp. *(branching)*	55	47	2585
		Acropora sp. *(branching)*	42	35	1470
		Acropora sp. *(corymbose)*	35	29	1015
		Pocillopora meandrina (corymbose)	15	15	225
		Acropora sp. *(corymbose)*	31	27	837
		Acropora sp. *(corymbose)*	23	23	529
		Acropora florida (branching)	33	29	957

x		*Acropora* sp. *(branching)*	63	69	4347
x		*Acropora* sp. *(plating)*	29	31	899
		Acropora sp. *(corymbose)*	28	29	812
		Acropora sp. *(corymbose)*	28	29	812
		Acropora sp. *(plating)*	19	20	380
x		*Acropora* sp. *(branching)*	55	43	2365
67		Total colonies tested			
45	63	Total planted colonies successfully detected			
67.2%	**94.0%**	Detection rate (% of total)			

Table S2.6. Coralclip® detection success rates for the two metal detectors (A - large, circular 20cm detection coil, B = small, pointed 9cm detection coil) for 67 coral colonies planted with Coralclip®, classified by size (binned according to maximum perpendicular length or width measurement) and morphology.

	Detection Rate (% total count)	
Morphology (n)	Metal Detector A (circular)	Metal Detector B (pointer)
Corymbose (38)	61%	94%
Branching (18)	76%	95%
Digitate (2)	50%	50%
Plating (9)	44%	100%
Size Class (cm, n)	Metal Detector A (circular)	Metal Detector B (pointer)
Small (<20cm, n =18)	67%	100%
Medium (>20cm, but <30cm, n = 15)	67%	80%
Large (>30cm, but <40cm, n = 25)	76%	100%
Extra Large (>40cm, n = 9)	45%	89%

2.6.3 Additional Results and Statistical Tables

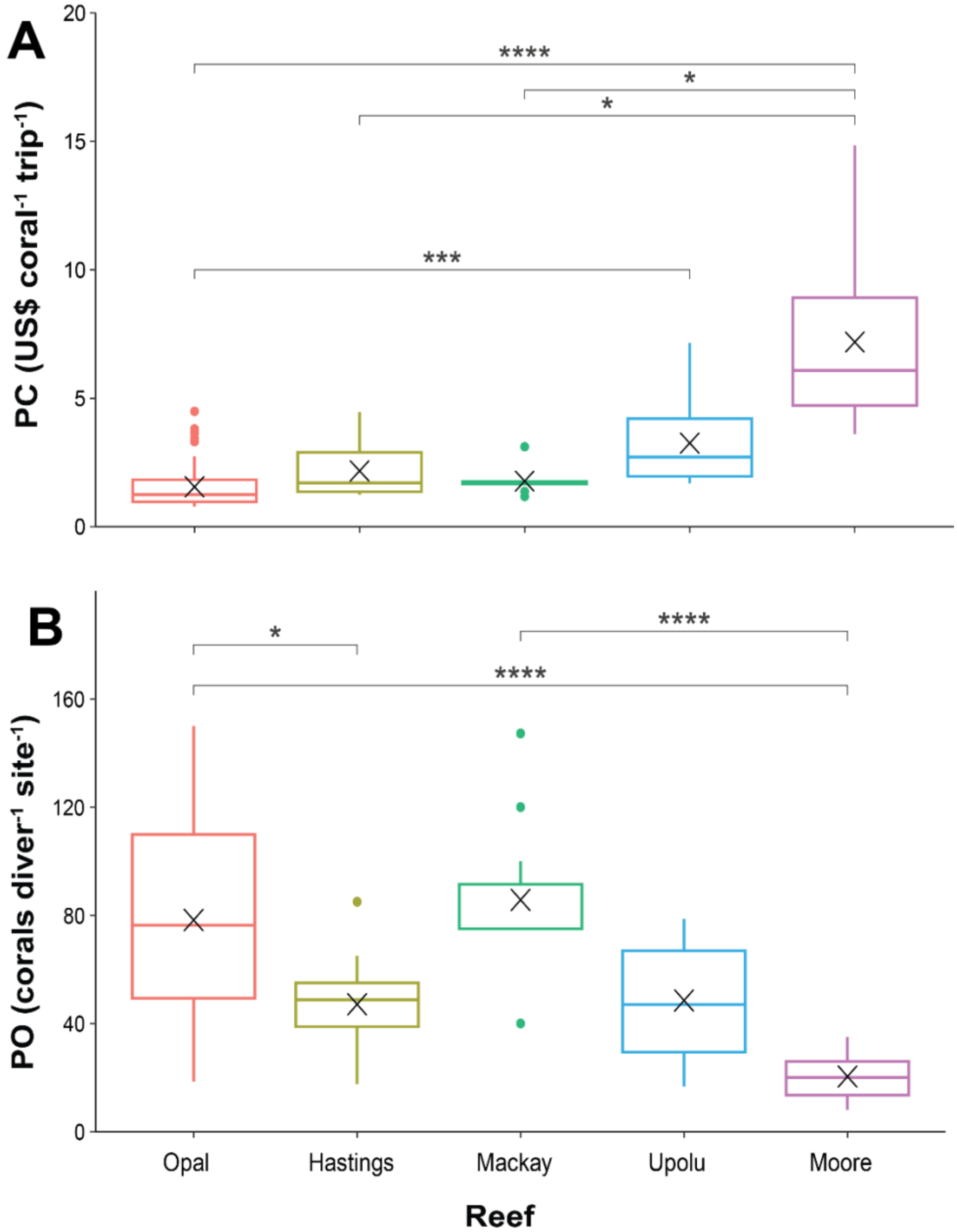

Fig S2.3. The distribution of planting cost (PC, **Panel A**) and Planting Output (PO, **Panel B**) values from 110 Coral Nurture Program "Routine Planting Days" across 5 reefs: Opal Reef (64 trips), Hastings Reef (14 trips), Mackay Reef (11 trips), Upolu Reef (12 trips), Moore Reef (9 trips). Shown in both figures is a box-and-whisker plot of untransformed PC (A) and PO (B) values per trip by Reef, showing the interquartile range (IQR), median value (centre line), range (whiskers, within 1.5x the IQR above and below the 75th and 25th percentile) and outlier values (points). Black crosses overlain on boxplots depict mean values. On both panels, horizontal bars and asterisks represent significant post-hoc comparisons ($p_{adj} < 0.05$) between Reefs following Kruskal-Wallis tests on log10 transformed data (Table S2.7). Significant p-values for both tests are represented by asterisks as follows: * < 0.05, ** <0.01, *** < 0.001 and **** < 0.000.

Table S2.7. Reef-based differences in PC and PE of coral planting activity on Coral Nurture Program "Routine Tourism Days. Shown are the results of (a) non-parametric Kruskal-Wallis ($p<0.05$) tests of stochastic distributional dominance between Reef groups performed on log-transformed PC and PE values across Reefs and (b) Dunn test post-hoc pairwise comparisons between Reefs of log-transformed PC and PE values, showing Bonferroni-adjusted p-values. (*denotes statistically significant $p < 0.05$).

(a) Kruskal Wallis test			
Variable	**KW chi-squared (H)**	**df**	**p-value**
PC	43.041	43.041	1.015×10^{-8}
PE	35.396	4	3.851×10^{-7}

(b) Dunn Test Post-Hoc Test				
Variable	**Group 1**	**Group 2**	**Statistic**	**P_{adjust}**
PC	Hastings	Mackay	-0.456	1.00
	Hastings	Moore	3.02	0.0254*
	Hastings	Opal	-2.26	0.237
	Hastings	Upolu	1.65	0.100
	Mackay	Moore	3.28	0.0104*
	Mackay	Opal	-1.48	1.00
	Mackay	Upolu	1.99	0.466
	Moore	Opal	-5.50	3.83×10^{-7}*
	Moore	Upolu	-1.46	1.00
	Opal	Upolu	4.18	2.92×10^{-4}*
PE	Opal	Hastings	-2.92	0.0354*
	Opal	Mackay	0.802	1.00
	Opal	Upolu	-2.57	0.101
	Opal	Moore	-4.89	1.01×10^{-5}*
	Hastings	Mackay	2.79	0.0535
	Hastings	Upolu	0.130	1.00

	Hastings	Moore	-2.06	0.394
	Mackay	Upolu	-2.57	0.103
	Mackay	Moore	-4.45	8.39×10^{-5}*
	Upolu	Moore	-2.11	0.347

Table S2.8. Results of separate one-way ANOVA tests of the mean proportion of Coralclip® observations with coral fragments 'alive', 'missing' and 'dead' dead between Reef groups from combined visual and metal detector roving surveys. Analysis conducted on arc-sin transformed proportional data (*denotes statistically significant $p < 0.05$).

Variable		df	Sum Sq	Mean Sq	F-value	p-value
% corals alive	Reef	4	0.2610	0.06525	13.21	2.16×10^{-6}*
	Residuals	31	0.1531	0.00494		
% corals missing	Reef	4	0.2201	0.05502	8.593	8.69×10^{-5}*
	Residuals	31	0.1985	0.00640		
% corals dead	Reef	4	0.08433	0.021082	4.514	0.00546*
	Residuals	31	0.14479	0.004671		

Table S2.9. Post-hoc comparisons of the mean proportion of (a) 'coral alive' (b) 'coral missing' and (c) 'coral dead' Coralclip® observations from combined visual and metal detector roving surveys between Reef groups from a Tukey's Honest Significant Difference test (HSD) following a one-way ANOVA for each variable (*denotes statistically significant $p < 0.05$).

	Contrast	Lower Bound	Upper Bound	Difference	p_{adjust}
(a) % corals alive	Hastings-Opal	-0.20	-0.020	-0.11	0.011*
	Mackay-Opal	-0.23	-0.020	-0.13	0.014*
	Upolu-Opal	-0.26	-0.050	-0.16	0.0016*
	Moore-Opal	-0.008	0.21	0.10	0.079
	Mackay-Hastings	-0.13	0.10	-0.016	1.00
	Upolu-Hastings	-0.16	0.10	-0.45	0.70
	Moore-Hastings	0.10	0.33	0.21	9.36×10^{-5}*

	Upolu-Mackay	-0.16	0.10	-0.030	0.96
	Moore-Mackay	0.098	0.355	0.226	1.50×10^{-4}*
	Moore-Upolu	0.127	0.385	0.256	2.25×10^{-5}*
(a) % corals missing	Hastings-Opal	-0.010	0.20	-0.94	0.094
	Mackay-Opal	0.037	0.28	0.16	0.0058*
	Upolu-Opal	0.044	0.29	0.17	0.0038*
	Moore-Opal	-0.17	0.072	-0.050	0.77
	Mackay-Hastings	-0.067	0.20	0.065	0.62
	Upolu-Hastings	-0.060	0.20	0.072	0.52
	Moore-Hastings	-0.28	-0.011	-0.14	0.028*
	Upolu-Mackay	-0.14	0.15	0.0069	1.00
	Moore-Mackay	-0.36	-0.062	-0.21	0.0023*
	Moore-Upolu	-0.36	-0.068	-0.22	0.0016*
(a) % corals dead	Hastings-Opal	0.01	0.18	0.94	0.034*
	Mackay-Opal	-0.10	-0.11	0.0026	1.00
	Upolu-Opal	-0.10	0.11	0.0089	1.00
	Moore-Opal	-0.17	0.041	-0.063	0.41
	Mackay-Hastings	-0.20	0.021	-0.091	0.16
	Upolu-Hastings	-0.20	0.028	0.085	0.21
	Moore-Hastings	-0.27	-0.044	-0.16	0.0028*
	Upolu-Mackay	-0.12	0.13	0.0063	1.00
	Moore-Mackay	-0.19	0.059	-0.066	0.55

	Moore-Upolu	-0.20	0.053	-0.072	0.47

Table S2.10. Summary of mean (±SE) proportion (%) of Coralclip® observations with coral fragments alive, missing or dead by Site from combined visual and metal detector timed-swim surveys. Superscript letters (a,b,c,d,e,f,g,h,i) next to mean % values denote significant Tukey's post hoc comparisons between survey sites ($p < 0.05$, Table S14), after significant one-way ANOVA tests for arc-sin transformed % corals Alive, Empty and Dead (ANOVA, all $p < 0.05$, Table S2.12)

Reef	Signif. Key	Site (n surveys)	Mean % fragments alive (SE)	Mean % fragments missing (SE)	Mean % fragments dead (SE)
Opal Reef	a	Rayban (3)	82.7 (2.1)^{h}	9.4 (1.5)$^{g, h}$	5.0 (1.1)
	b	Mojo (3)	86.2 (1.4)$^{g, f, h}$	7.2 (1.5)$^{g, h}$	4.7 (1.8)
	c	Blue Lagoon (3)	77.3 (3.9)	19.4 (3.5)	3.5 (0.6)
	d	BM (4)	78.8 (1.7)	10.0 (0.5)$^{g, h}$	7.7 (1.6)
		Combined (13)	81.1 (1.4)	11.4 (2.5)	5.4 (0.8)
Hastings Reef	e	1770 (3)	73.7 (2.0)^{e}	16.4 (3.2)	9.9 (3.7)
	f	Stepping Stones (5)	70.3 (4.9)$^{b, i}$	18.7 (3.8)^{i}	10.9 (2.3)^{i}
		Combined (8)	71.6 (2.3)	17.8 (2.5)	10.5 (1.9)
Mackay Reef	g	Angels (5)	70.4 (2.4)$^{b, g}$	22.8 (2.6)$^{d, b, i, a}$	5.7 (1.7)
Upolu Reef	h	Wonderwall (5)	67.7 (2.1)$^{b, i, a}$	23.3 (2.4)$^{d, b, i, a}$	5.7 (1.0)
Moore Reef	i	Pontoon (5)	88.2 (1.7)$^{e, g, f}$	8.5 (1.8)$^{g, f, h}$	2.9 (0.8)^{f}
Overall			**76.6 (1.5)**	**15.6 (1.3)**	**6.3 (0.7)**

Table S2.11. Results of separate one-way ANOVA tests performed on arc-sin transformed mean proportions of Coralclip® observations with coral fragments 'alive', coral fragments 'missing' and coral fragments 'dead' between reef Sites (n = 9) from combined visual and metal detector surveys. Analysis conducted on arc-sin transformed proportional data using the Rstatix package in R Studio. (*denotes statistically significant $p < 0.05$).

Variable		df	Sum Sq	Mean Sq	F	p-value
% corals alive	Site	8	0.29	0.036	7.73	2.38×10^{-5}*
	Residuals	27	0.13	0.005		

% empty	Site	8	0.28	0.035	6.74	7.49×10^{-5}*
	Residuals	27	0.14	0.005		
% dead	Site	8	0.10	0.013	2.69	0.0259*
	Residuals	27	0.128	0.005		

Table S2.12. Results of significant (p_{adjust}<0.05) Tukey's Honest Significant Difference test (HSD) post-hoc comparisons between reef Sites performed on the transformed mean proportions of (a) 'coral alive' (b) 'coral missing' (c) 'coral dead' Coralclip® observations from combined visual and metal detector surveys, following one-way ANOVA tests for each variable.

	Contrast	**Lower Bound**	**Upper Bound**	**Difference**	$\mathbf{P_{adjust}}$
(a) % corals alive	Moore Reef Pontoon-1770	0.022	0.36	0.19	0.018
	Mojo-Angels	0.027	0.36	0.20	0.014
	Moore Reef Pontoon-Angels	0.081	0.37	0.23	0.0005
	Stepping Stones-Mojo	-0.36	-0.024	-0.19	0.016
	Wonderwall-Mojo	-0.39	-0.057	-0.23	0.0031
	Stepping Stones-Moore Reef Pontoon	-0.37	0.078	-0.22	0.0006
	Wonderwall-Moore Reef Pontoon	-0.40	-0.11	-0.26	0.0001
	Wonderwall-Rayban	-0.34	-0.008	-0.18	0.034
(b) % corals missing	BM-Angels	-0.34	-0.012	-0.17	0.029
	Mojo-Angels	-0.40	-0.050	-0.34	0.0051
	Moore Reef Pontoon-Angels	0.36	-0.055	-0.21	0.0026
	Rayban-Angels	-0.36	-0.009	-0.19	0.034
	Wonderwall-BM	0.019	0.343	0.181	0.021
	Wonderwall-Mojo	0.057	0.410	0.234	0.0036
	Stepping Stones-Moore Reef Pontoon	1.36×10^{-4}	0.306	0.153	0.050

	Wonderwall-Moore Reef Pontoon	0.062	0.37	0.22	0.0018
	Wonderwall	0.16	0.37	0.19	0.025
(b) % corals dead	Stepping Stones-Moore Reef Pontoon	0.018	0.31	0.12	0.019

Table S2.13. Results of separate one-way ANOVA tests of the mean proportion of Coralclip® observations with coral fragments 'alive', 'missing', and 'dead' between survey timepoints (n = 4) at triplicate fate-tracked plots planted at "1770", Hastings Reef. (*denotes statistically significant p <0.05).

Variable		**df**	**Sum Sq**	**Mean Sq**	**F-value**	**p-value**
% corals alive	Timepoint	3	0.24	0.024	26.70	1.61×10^{-4}*
	Residuals	8	0.024	0.0030		
% corals missing	Timepoint	3	0.075	0.025	2.90	0.10
	Residuals	8	0.069	0.0086		
% corals dead	Timepoint	3	0.21	0.071	12.45	0.0022*
	Residuals	8	0.046	0.0057		

Table S2.14. Results of post-hoc comparisons between survey timepoints (n = 3) of the transformed mean proportion of (a) 'coral alive' (b) 'coral missing' (c) 'coral dead' Coralclip® observations at fate-tracked plots planted at "1770", Hastings Reef from Tukey's Honest Significant Difference tests (HSD) following a one-way ANOVA for each variable (*denotes statistically significant p <0.05).

	Contrast	**Lower Bound**	**Upper Bound**	**Difference**	p_{adjust}
(a) % corals alive	T54-T115	-0.25	0.035	0.12	0.15
	T14-T115	-0.29	-0.002	0.14	0.047*
	T379-T115	-0.36	-0.074	-0.22	0.0054*
	T14-T54	-0.18	0.11	0.037	0.84
	T379-T54	-0.47	-0.18	-0.32	3.86×10^{-4}*
	T379-T14	-0.50	-0.22	-0.36	1.79×10^{-4}*
(b) %	T54-T151	-0.018	0.38	-0.18	0.076

corals dead	T14-T151	-0.018	0.38	-0.18	0.076
	T379-T151	-0.06	0.34	0.14	0.15
	T14-T54	-0.20	0.20	-6.93×10^{-17}	1.0
	T379-T54	0.12	0.52	0.32	0.0039*
	T379-T14	0.12	0.2	0.32	0.0039*

Table S2.15. Results of separate one-way ANOVA tests on the mean proportion of Coralclip® observations with coral fragments 'alive', 'missing' and 'dead' from between survey timepoints (n=3) at fate-tracked plots planted at "Angels", Mackay Reef. (*denotes statistically significant p <0.05).

Variable		df	Sum Sq	Mean Sq	F-value	p-value
% corals alive	Timepoint	2	0.43	0.22	19.82	0.0023*
	Residuals	6	0.065	0.011		
% corals missing	Timepoint	2	0.19	0.097	8.762	0.017*
	Residuals	6	0.066	0.011		
% corals dead	Timepoint	2	0.26	0.13	7.066	0.027*
	Residuals	6	0.11	0.019		

Table S2.16. Results of post-hoc comparisons between survey timepoints (n = 3) of the arc-sin transformed mean proportion of (a) 'coral alive' (b) 'coral missing' (c) 'coral dead' Coralclip® observations at fate-tracked plots planted at "Angels" Mackay Reef from Tukey's Honest Significant Difference tests (HSD) following a one-way ANOVA for each variable (*denotes statistically significant p <0.05).

	Contrast	Lower Bound	Upper Bound	Difference	p-value
(a) % corals alive	T261-T154	-0.41	0.11	-0.15	0.26
	T14-T154	-0.63	-0.12	-0.37	0.011*
	T14-T261	-0.78	-0.26	-0.52	0.002*
(b) % corals missing	T261-T154	-0.20	0.33	0.068	0.72
	T14-T154	0.008	0.54	0.27	0.045*
	T14-T261	0.076	0.60	0.34	0.018*

(b) % corals dead	T261-T154	-0.23	0.45	0.11	0.62
	T14-T154	-0.045	0.64	0.30	0.083
	T14-T261	0.062	0.74	0.40	0.026*

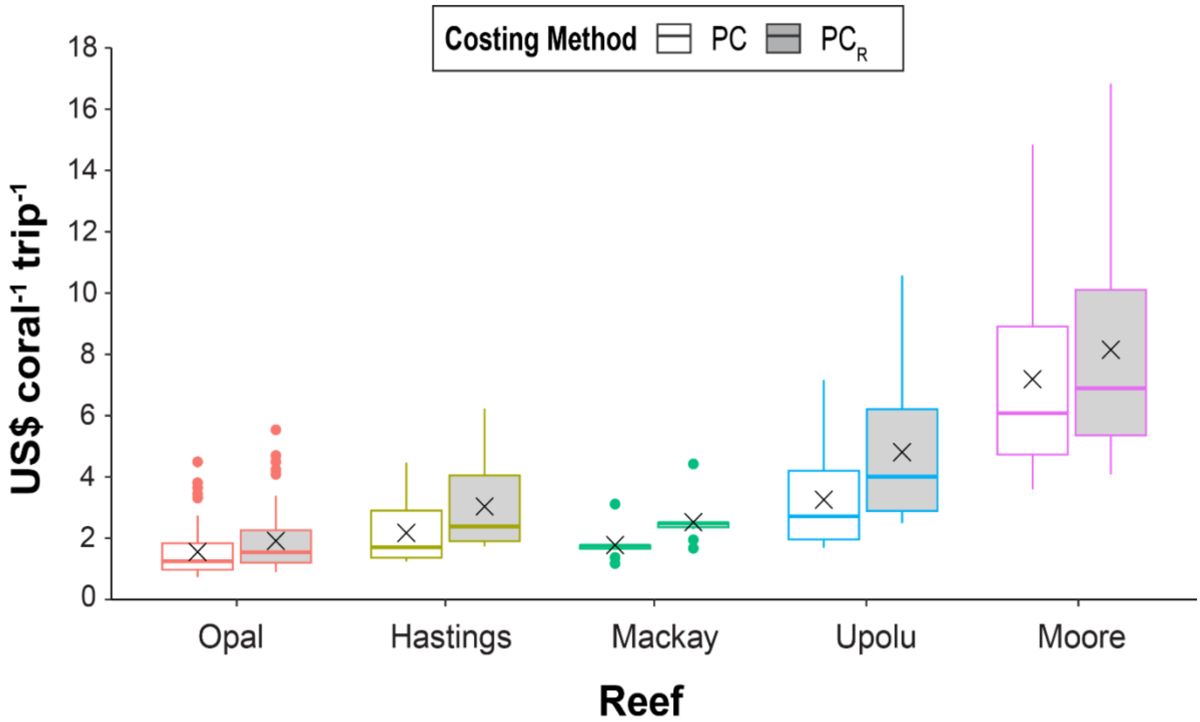

Figure S2.4. Comparison of planted cost (PC, Eqn 1, white boxes, left) and realised cost (PC_R, Eqn 3, grey boxes, right) values of 30,031 corals planted across 110 "Routine Planting Day" trips conducted across 5 reefs on the GBR: Opal Reef (64 trips), Hastings Reef (14 trips), Mackay Reef (11 trips), Upolu Reef (12 trips), Moore Reef (9 trips). PC (Fig. S2.3) assumes 100% survivorship of planted material, whereas PC_R accounts for the mean survivorship of planted coral material derived from roving surveys of the respective Reef (Fig 2.3A). Mean PC_R increased from mean PC by 23.3%, 39.7%, 42.1%. 47.7% and 13.4% respectively across Reefs (left – right). Shown are box-and-whisker plot of PC and PC_R values depicting the interquartile range, the median (centre horizontal line), range (whiskers, within 1.5x the IQR above and below the 75th and 25th percentile) and outliers (points). Black crosses overlain on boxplots depict mean values.

Chapter 3: Early-stage outcomes and cost-effectiveness of implementing tourism-led coral propagation and outplanting in the Whitsundays (Great Barrier Reef).

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3.1 Abstract

To support reef resilience and recovery, implementation of reef restoration practices within management strategies is accelerating. However, full costs underpinning restoration project feasibility – critical to informed decision making around restoration cost-benefits – have historically been under-reported. Such knowledge is especially lacking for Australia's Great Barrier Reef, where stakeholder-led coral restoration was initiated in 2017 (Cairns-Port Douglas) and continues to scale. Here we describe the early outcomes and costs of implementing tourism operator-led coral propagation and outplanting practices (Coral Nurture Program, CNP) in the Whitsundays. Specifically, we detail the local operational and environmental context (e.g., baseline benthic ecology and stakeholder engagement), describe the associated costs of implementation and continuation of restoration activities and evaluate survivorship of coral outplants across sites for the first nine months after project establishment (August 2022 to June 2023). Baseline benthic surveys revealed low hard coral (HC) cover at the three chosen restoration sites (ranging from 3.22-8.67%), which significantly differed in benthic composition from coral collection sites (HC cover: 16.67-38.06%), supporting high motivation by tourism operators to undertake restoration activities at the chosen restoration sites. Mean coral outplant survivorship in fate-tracked plots differed between restoration sites after 267 days (23.33 - 47.58 %) but stabilised at all sites four months. Low survivorship was driven by coral dislodgement rather than disease or mortality. Early-stage cost-effectiveness associated with implementation of restoration activity varied from US\$10.33-178.55 coral^{-1} (n = 4,425 outplants) depending on whether 'in-kind' costs, restoration activity (outplanting only *vs.* total costs from planning through monitoring), outplant survivorship, or a combination of these factors, were considered. Our results therefore highlight a time-critical need for consistent reporting and monitoring methods, and the consideration of costs often unquantified in restoration cost evaluations (e.g., in kind costs, program overheads), as restoration projects continue to become established globally. Furthermore, in documenting the implementation of CNP activity in the Whitsundays, we highlight the need to consider the influence of local social-environmental contexts and their associated cost-benefits in future application and socioeconomic evaluations of reef restoration. Finally, we highlight that long-term and locally tailored socio-economic and -ecological monitoring is needed to determine and improve cost-benefits of investment as activity continues.

3.2 Introduction

To boost resilience of coral reef ecosystems under persistent anthropogenic pressures, adoption of coral restoration approaches for targeted local-scale site intervention is accelerating globally (Boström-Einarsson et al., 2020; Shaver et al., 2022). Coral restoration, in parallel with mitigation of global climate change and local stressors, is now considered a central means to conserve the socio-ecological value of coral reefs (ICRI, 2021; Kleypas et al., 2021), including the Great Barrier Reef (GBR) (GBRMPA, 2017; Suggett et al. 2023). Implementation of coral reef restoration activity has extended to more than 50 countries over the last two decades (Boström-Einarsson et al., 2020), however, management of the GBR using coral restoration was not considered until the World Heritage Area was severely impacted by consecutive mass coral bleaching and mortality events in 2016 and 2017 (Anthony et al., 2017; McLeod et al., 2022). Widespread mortality of corals across the marine park network (Hughes et al., 2018, 2021) prompted rapid trialling of proactive interventions from 2017, notably via small-scale funding into various community-led restoration activities (McLeod et al., 2022) and parallel large-scale funding into restoration research and development (Anthony et al., 2020). Approaches in the past five years have spanned asexual coral propagation and outplanting (Cook, 2022; Cook et al., 2022; Howlett et al., 2022), substrate stabilisation (McLeod et al., 2022; Nuñez Lendo et al., In Review), macroalgae removal (e.g., Smith et al., 2022), and larval-based restoration approaches (e.g., Randall et al., 2021, 2023). Most of these approaches have been implemented at relatively small scales, except through networks of stakeholders undertaking parallel propagation activities (Howlett et al., 2022)

On the GBR, local reef stakeholders including tourism operators, Traditional Owners, and citizen scientists are central to monitoring and maintaining site condition across the marine park network (e.g., Marshall et al., 2012; Beeden et al., 2014; GBRMPA, 2015). Stakeholder “stewardship” programs have since evolved to play a key role in pioneering coral restoration efforts (Howlett et al., 2022; McCleod et al., 2022). The reef tourism industry has particularly demonstrated capacity for proactive stewardship practices to respond to reef disturbance beyond monitoring-focussed efforts (GBRMPA, 2020; Hein et al., 2020; Howlett et al., 2022; Bartelet et al., 2023). When co-designed and well planned, reef restoration-based stewardship can support local site recovery (Hein et al., 2020; Calle-Triviño et al., 2021; Howlett et al., 2023; Knoester et al., 2023) as well as provide positive feedback loops to reef stakeholders and Traditional Owners though socioeconomic and cultural benefits (Kittinger et al., 2016; Hein et al., 2019; Westoby et al., 2020; Suggett et al., 2023). Stewardship has been shown to promote shared responsibility amongst practitioners thereby solidifying sustained participation (e.g., Kittinger et al., 2016; Hein et al., 2019; Virdis et al., 2021), generating alternative livelihoods and revenue (Bayraktarov et al., 2020; Suggett et al., 2023), improving social licence through community education and awareness of reef threats (Palou Zúniga et al., 2023; Quigley et al., 2022) and importantly reducing costs through in-kind contributions of time, knowledge or resources (e.g., dela Cruz et al., 2014; Hein et al., 2018; Suggett et al., 2020, 2023). Partnerships and collaborations amongst stakeholders are therefore integral to support reef recovery at socio-economically and ecologically relevant temporal and spatial scales (Bayraktarov et al., 2015; Westoby et al., 2020; Suggett et al., 2023).

In 2018, reef researchers and tourism operators initiated "The Coral Nurture Program" (CNP) in the Cairns and Port Douglas region on the northern GBR, an industry-research partnership aimed at delivering cost-effective asexual coral propagation and outplanting (detailed in Howlett et al., 2022; Suggett et al., 2023). CNP was conceived with the dual aims to enhance local site recovery and the stewardship capacity of reef tourism operators through maintaining and improving hard coral cover at high-value reef tourism sites (Howlett et al., 2022). Staged implementation of activity by a pool of six 'high-standard' tourism operators over four years, has resulted in (as of mid-2023) over 100,000 corals outplanted and more than 120 coral nurseries established at 27 discrete reef sites. Detailed monitoring between 2018-2021 demonstrated average coral outplant survivorship of >75% across diverse reef sites **(**Scott et al., 2024; **Chapter 2)** and positive benefits of outplanting through enhanced recovery dynamics of key species (Roper et al., 2022), particularly at sites with lower initial hard coral cover (Howlett et al., 2023). Furthermore, the first corals propagated through the CNP reached reproductive maturity in 2021 (J. Edmondson & C. Roper, personal observation, October 2021), further boosting potential for accelerated ecological recovery. Whilst well-established, stakeholder-led coral restoration models have demonstrated capacity to scale in reef regions elsewhere (e.g., The Caribbean, see Carne & Trotz, 2021; Lirman & Schopmeyer, 2016; Bayraktarov et al., 2020; Blanco-Pimentel et al., 2022), how CNP-type restoration approaches initiated across Cairns and Port Douglas can be feasibly adopted elsewhere remains unknown. However, in August 2022, CNP coral propagation and outplanting was implemented at three inshore, fringing-reefs in the Whitsundays to determine if and how activity could be tailored to this region.

As with Cairns and Port Douglas, the Whitsundays represents a major GBR tourism gateway, where the reef tourism sector provides 28% of total employment (Tourism Research Australia, 2023) and generates upwards of US$900,000/km^2 in estimated annual tourism ecosystem service value (Spalding et al., 2016, 2017). However, the Whitsundays was devastated in March 2017 by tropical Cyclone Debbie (Category 4), which battered exposed reef sites for over 18 hours (Bureau of Meteorology, 2018), scouring coral communities and resulting in an average loss of 55% in coral cover in the region (Williamson et al., 2019). Whilst observations have documented early evidence of natural recovery via hard coral larval recruitment (McLeod et al., 2019; AIMS, 2022, 2023; Thompson et al., 2023), recovery of inshore coral assemblages has been challenged by chronic high nutrient and sediment loads from coastal runoff (Fig. S3.1A) (Waterhouse et al., 2021; Thompson et al., 2023). Given the slow rate and suppressed capacity of natural recovery of reef habitats in the Whitsundays (Fig S3.1B, Thompson et al., 2023), equipping tourism operators with new and additional site stewardship capacity may significantly contribute to to supporting the health and recovery of high-value tourism reef sites. Small-scale coral propagation activities have been conducted in the Whitsundays in the last five years (e.g., Cook, 2022; McLeod et al., 2022); however, the outcomes and implementation costs of these efforts are unresolved. For example, Cook (2022) reported 67-78% survivorship of propagated corals in nurseries after six months, yet the resulting survivorship of corals outplanted to the reef – a key factor underpinning cost-effectiveness – remains unknown.

Understanding the feasibility, and ultimately the sustainability of restoration interventions, rests on their financial viability or cost-effectiveness (Cook et al., 2017; Iacona et al., 2018; Suggett et al., 2023). However, in coral reef restoration practice, costs have been rarely and inconsistently reported (see **Chapter 1 & 2**). Few reports detail project life-cycle costs including implementation, training, maintenance, and monitoring (Spurgeon & Lindahl, 2000; Bayraktarov et al., 2019) and/or quantify the contribution of "in-kind" resources such as volunteer or researcher time (Edwards et al., 2010). Such a knowledge gap impedes collective understanding of the 'true costs' of restoration efforts (Hein & Staub, 2021), thereby limiting the ability of reef management, funding agencies and restoration practitioners to adequately budget for, invest in, and deliver effective and sustainable site intervention (Edwards et al., 2010; Bayraktarov et al., 2015, 2019; Suggett et al., 2023). Such data are especially lacking for the GBR, and our previous evaluation of outplanting cost-effectiveness of CNP activity in the Cairns-Port Douglas region (3.5 years, representing 154 coral outplanting 'trips or deployments) (Scott et al., 2024; Chapter 2) yielded mean 'realised' planting costs (adjusted for outplant survivorship) spanning US$2.94-21.23 coral^{-1} trip^{-1}. However, this cost-tracking exercise started mid-program, and thus failed to capture early implementation costs, which are inevitably prone to be higher. Initiation of CNP in the Whitsundays therefore provided an opportunity to track restoration costs more rigorously. To achieve this goal, we (i) detail the operational and environmental context for adoption of CNP activity in the Whitsundays (e.g., baseline benthic ecology and stakeholder engagement), (ii) describe the implementation and associated costs of restoration activity and (iii) evaluate early-stage survivorship of coral outplants across three sites during the first nine months of establishment (August 2022 to June 2023). Collectively, we use these data to examine the early-stage cost-effectiveness of implementing coral outplanting-based reef restoration activity in the Whitsundays based upon retaining new, surviving coral biomass at reef sites. We discuss the key achievements, challenges, and complexities of adapting the existing CNP reef stewardship approach from Cairns and Port Douglas to the Whitsundays reef system as a result of differing environmental conditions and tourism operational contexts.

3.3.0 Methods

3.3.1 Coral Nurture Program Whitsundays social-ecological context and implementation.

Coral Nurture Program Whitsundays (CNPW) is a partnership between researchers from the University of Technology Sydney (UTS) and three 'high-standard' Whitsundays tourism operators, with local coordination support from the natural resource management (NRM) group Reef Catchments (RC). The tourism operators were already involved in other reef stewardship activities through the Great Barrier Marine Park Authority's (GRBMPA) "Reef Protection Initiative" (RPI) and applied through an Expression of Interest to an open call to partner with CNPW. Establishment and operation of CNPW was financed through philanthropic funding in early 2022 (specifically "venture philanthropy"; Suggett et al. 2023) and supported through in-kind contributions via UTS, RC and tourism operators (herein referred to as 'operators'). Operator staff time and vessel use for several monitoring and maintenance trips were also funded via the RPI tourism stewardship program from January 2023. In April 2022, Prior to any CNPW activities, a variety of key stakeholders from the region were consulted and a detailed Public Information Package (PIP) was prepared and circulated through the CNP website, GBRMPA website, social media platforms and the local Whitsundays newspaper. Development of a PIP is part of the GBRMPA process to evaluate permitting suitability, seek Native Title consent from reef Traditional Owners, and feedback from the community. No comments were submitted, and the permit (G22/46543-1) was granted in August 2022.

Coral propagation and outplanting activity via the CNPW program was initiated at three fringing reef sites in the Whitsundays on Australia's Great Barrier Reef (GBR) in August 2022: "Blue Pearl Bay" (BPB) on Hayman Island, "Black Island" (BI) and "Luncheon Bay" (LB) on Hook Island (referred to a "Outplanting sites"; Fig. 3.1). Sites are located approximately 30 km offshore from Airlie Beach, on the north-eastern side of each respective island and hence were heavily exposed to high winds and storm surges generated by the south-western trajectory of Cyclone Debbie (2017). Whilst no historical data exists for the selected CNPW sites, declines in hard coral cover of 6-24% (2016-2020) were documented at nearby reef sites on Hook and Hayman Island, with most recent estimates of hard coral cover at ~15% (AIMS, 2022, 2023). Preliminary benthic video surveys at CNPW sites in 2020 and 2021 approximated hard coral cover at <7% and largely composed of 'massive' hard coral taxa (Table S3.1). Even so, sites remain heavily frequented by tourism operators and private charter boats via shared public moorings. Outplanting sites were chosen, in consultation with local tourism operators, for both their operational suitability (i.e., ease of access for routine monitoring and maintenance, alignment to tourism-led stewardship and community engagement activities; detailed local site knowledge), as well as habitat suitability for rehabilitation activities (e.g., flushing from offshore waters to mitigate sediment deposition (Ceccarelli et al., 2020) and ample availability of consolidated substrate for attaching corals with Coralclip® (Fig.3.2A, Table. S3.1)).

Each of the three operators self-nominated as 'lead practitioner' for one of the three sites based upon their regular visitation of nearby reef sites during routine tourism activity; in this way, operations aligned to those in CNP (Cairns-Port Douglas) where operators steward individual reef sites (Howlett et al. 2022). Specifically, for CNPW: Operator A, Blue Pearl

Bay (equipped for 3–5-day diving liveaboard trips for up to 10 passengers, 3 crew); Operator B; Black Island (equipped for snorkelling trips for up to 30 passengers, 3 crew); Operator C; Luncheon Bay (equipped for snorkelling trips for up to 25 passengers, 2 crew). However, in the Whitsundays region, vessel moorings are largely public, and thus, Operators A-C are not the only vessels that visit CNPW sites, and Operators A-C are not restricted to CNPW sites on their tourism days. Although each operator led stewardship of coral propagation and outplanting, site maintenance and monitoring at their respective site, activities were largely conducted collaboratively with all tourism operators, UTS researchers and the RC local coordinator. At project initiation, each operator agreed to the CNP code of operation, a set of key principles designed to align common goals, expectations, and trust across stakeholders (Howlett et al. 2022; see also coralnurtureprogram.org).

To establish the three CNPW 'outplanting' sites and provide operator training on the surveying and data reporting forms, a chartered 'Site Setup' trip was conducted over nine days in late August 2022. Baseline ecological surveys, nursery installation and stocking, coral colony collection, outplanting and training were conducted over three days at each site consecutively by CNP researchers and operators. Following this, operators were contacted monthly for project updates and problem mitigation. After two months of activity, all operators generally recognised the need for a coral identification workshop to facilitate improved coral selection and reporting, which was provided by UTS researchers in November 2022. A subsequent trip was made by a UTS researcher with Operator C in February 2022 to help re-stock coral nurseries at LB, and conduct training in donor colony collection. An additional nursery re-stocking trip was conducted at BI with Operator B and the RC local manager. In March 2023, UTS researchers returned to the sites over a three day 'Monitoring and Training' trip to reassess the sites, restock nurseries at BPB, and to provide ongoing training with the CNPW operator team. In June 2023, the end of the period considered for this current study, a UTS researcher accompanied each operator to conduct intensive outplanting and training at each site as part of a global coral stakeholder-led restoration awareness initiative, "Coralpalooza™".

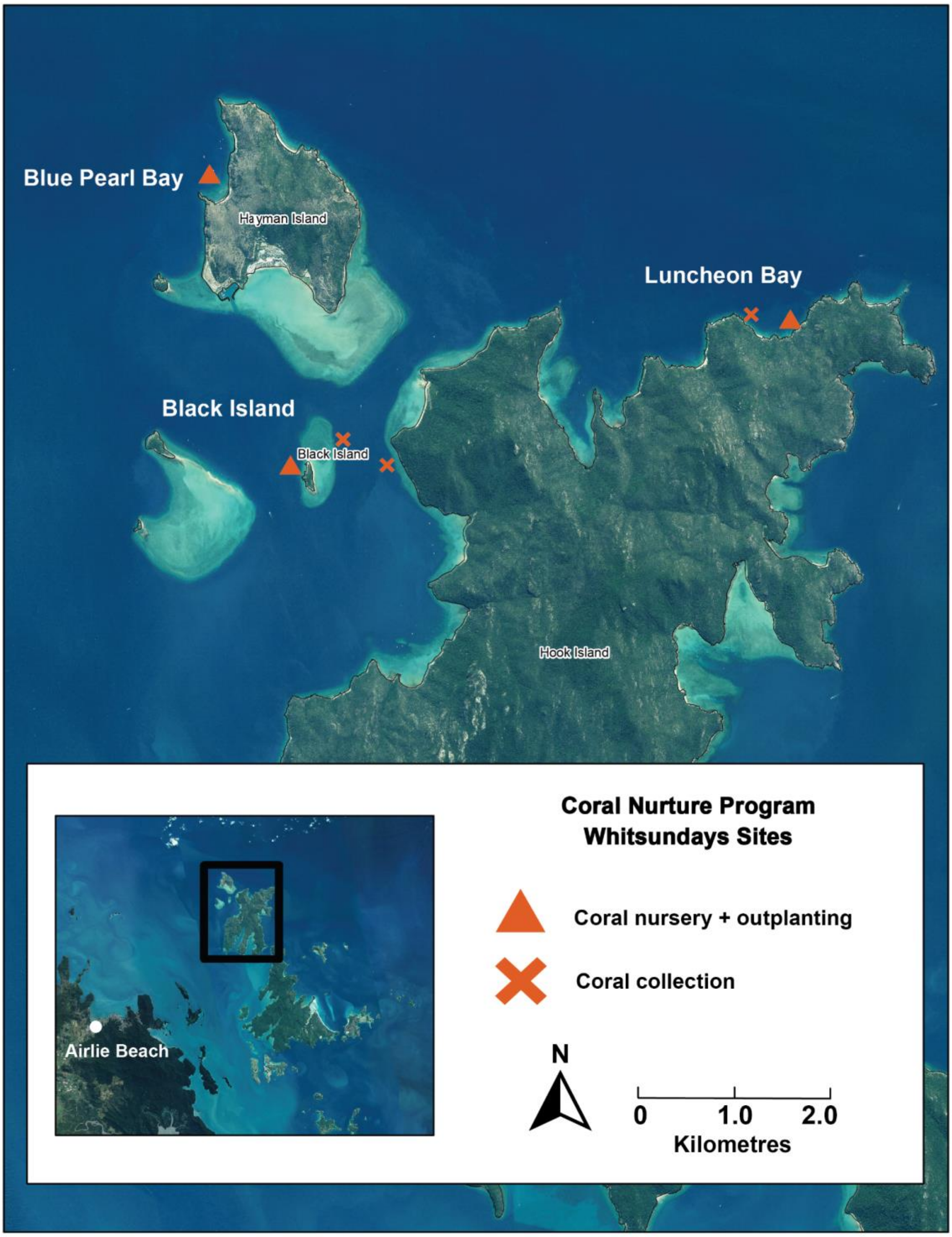

Figure 3.1. Map showing the locations of the three Coral Nurture Program Whitsundays (CNPW) 'outplanting' sites: Blue Pearl Bay (BPB) (20°2′48.91" S 148°52'5.76" E), Black Island (BI) (20°4′57.98" S 148°53'25.97" E) and Luncheon Bay (LB) (20°3′52.58" S 148°57'4.69" E) (triangles). 'Coral collection' sites are where Corals of Opportunity (CoO) and occasional donor colonies (within permit requirements) were collected to stock nurseries or outplant directly to CNPW sites: "Cockatoo Point" (CP) (20°4′57.42" S 148°53'41.82" E) on the western-side of Hook Island - adjacent to Black Island; "Wonderwall" (WW) (20°4′57.55" S 148°54'8.63" E) on the eastern-

side of Black Island; and “Luncheon Bay Donor” (LBD) (20°3′56.27”S 148°56’36.88”E) on the northern-side of Hook Island. All three CNPW ‘outplanting’ sites are located in Marine National Park Zones (no-take areas).

3.3.2 CNPW coral collection, nursery propagation and outplanting.

Establishing and trialling coral nurseries at ‘outplanting’ sites was considered critical from the project outset to overcome the low local coral cover (Table S3.1) and associated low availability of naturally detached coral fragments (Corals of opportunity (CoO)). Although installation and monitoring of nurseries can introduce considerable additional restoration costs (e.g., Edwards et al., 2010; Scott et al., 2024; **Chapter 2**), they simultaneously reduce donor-site impacts and time-costs associated with collecting coral material at neighbouring sites. In August 2022, table nurseries (n=3) were installed at each of the three restoration sites on sandy areas at a depth of 3-4 m (on low tide) (Fig. 3.2C). Each nursery frame consists of a sheet of diamond aluminium mesh, secured to two parallel 85 x 85 cm stainless steel frames with stainless steel wire. Frames were anchored to the sand with steel rebar stakes and sit 50 cm above the substrate to minimise sedimentation exposure and facilitate water flow. Nursery tables were initially stocked with coral material from nearby donor reef sites (“collection sites”) located within 10 km of CNPW ‘outplanting’ sites and with hard coral cover ranging from 17-38% (*see below, section iii;* Fig. 3.2C, Fig. 3.3A): “Cockatoo Point” (CP), “Wonderwall” (WW), and “Luncheon Bay Donor” (LBD) (Fig. 3.1). Collected material was largely CoO but was occasionally supplemented with in-situ fragmentation of donor colonies using a hammer and chisel or wire cutters to enhance diversity of propagated species (within permit conditions: <10% of parent colony, fragments >15 cm in size).

Collected corals were immersed in seawater and kept shaded for transportation by boat to the ‘outplanting’ sites (<1 hr transit), where they were immediately transferred back into the water. Further fragmentation of colonies was avoided for nursery stocking to prevent additional stress, and to enable ‘wedging’ of larger fragments into the spaces of the diamond mesh nursery frame (Fig. 3.2C). Once nurseries were stocked, colonies were photographed to conduct species identification. Stocked nursery corals were initially not tied down to prevent use of plastic cable ties (Boström-Einarsson et al., 2020; GBRMPA, 2020); however, subsequent coral loss due to suspected wave action and Bumphead parrotfish (*Bolbometopon muricatum*) predation knocking the fragments lose (C. Hayward, personal observation, December 2022), meant cable ties were used to secure coral fragments upon nursery re-stocking.

At each CNPW 'outplanting' site, reef areas were identified for outplanting, controls (no outplanting) or for fate tracked plot (see below, *section iv*). Outplanting was initiated during 'Site Setup' in late August 2022, and for this study was tracked until June 2023 (267 days) but remains ongoing. Operators could outplant at their own will during this period, but instead opted for more coordinated outplanting efforts. Outplanting was therefore primarily conducted collectively by tourism operator personnel and researchers across three 'outplanting blitz' events: at 'Site Setup' in late August 2022, after six months during a 'Monitoring and Training' trip in March 2023, and during the 'Coralpalooza™' event in June 2023 (nine months post establishment) where other CNPW volunteers were also involved.

During all events, coral material was outplanted using Coralclip® (Suggett et al., 2020) on areas of bare, consolidated substrate nearby nursery sites. A pre-outplanting demonstration was provided to all tourism operators, and initial outplanting efforts evaluated visually by researchers to ensure proper and consistent deployment. Specifically, Coralclip® units are hammered into bare areas of rock and checked for secure integration into the substrate (or otherwise removed and re-hammered). Surface sediment or turfing algae is then firmly brushed so that a coral fragment can then be securely positioned under the spring-loaded clip on bare substrate (Suggett et al., 2020; Howlett et al., 2022; Roper et al., 2023). Where possible, fragments were kept ⋝10cm, oriented upwards, with exposed skeleton positioned flush with the substrate to encourage self-attachment (Lewis et al., 2022) and to avoid smothering by sediment (Fig. 3.2D). Coral material was photographed prior to outplanting for later identification and the number and taxonomy (identified to species where possible, otherwise genus and morphology) of outplants was reported to central CNP management via standardised reporting forms (Howlett et al., 2022; Scott et al., 2024; **Chapter 2**).

Following 'Site Setup', coral outplant survivorship in fate-tracked plots (see below, *section iv*) and coral nurseries were monitored on six occasions between September 2022 and June 2023. During these visits, nursery frames were photographed and any dead, or diseased coral fragments photographed and removed. Monitoring was led by two tourism operator staff members and the RC local coordinator, with an accompanying CNP researcher to facilitate training and data collection.

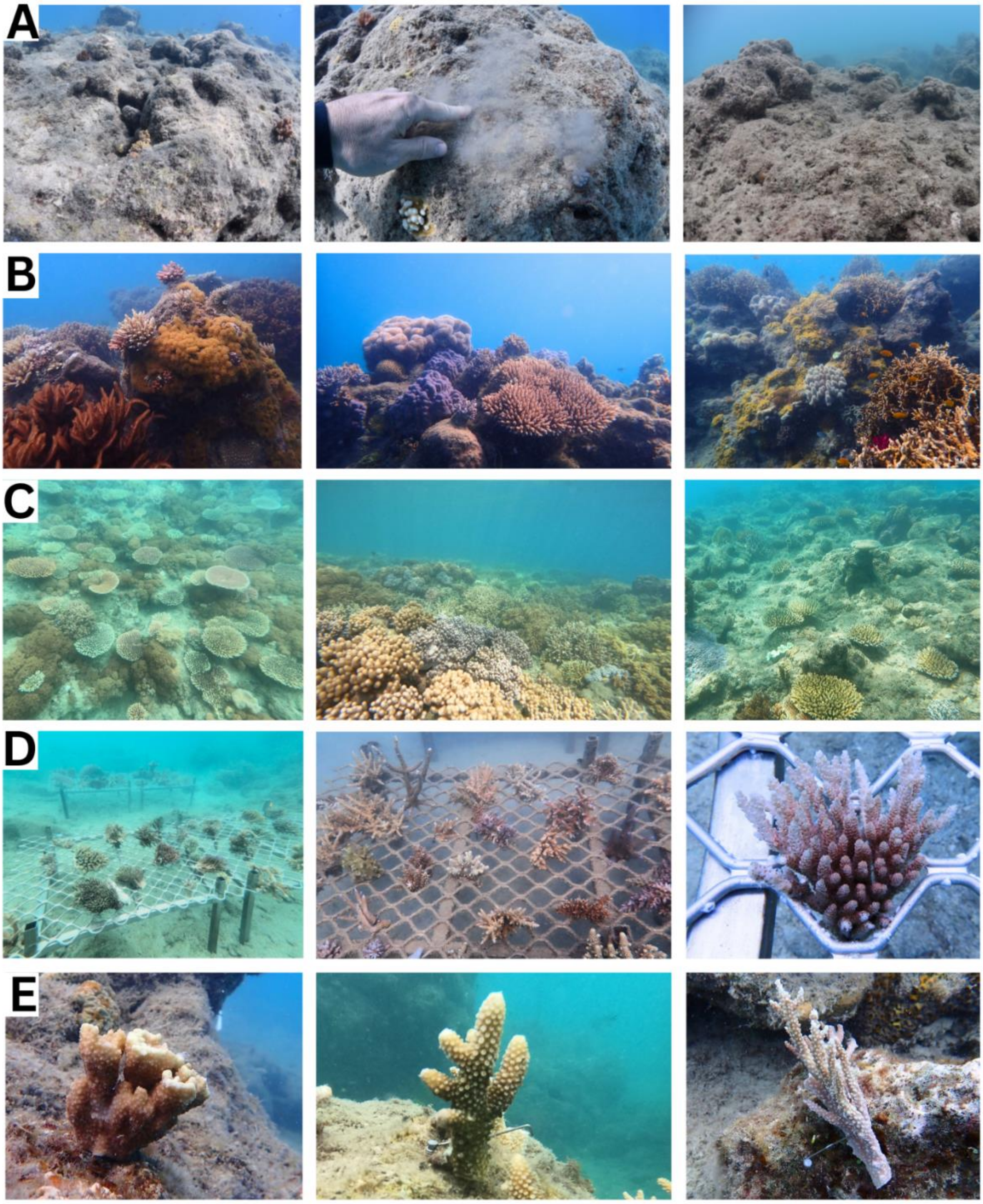

Figure 3.2. A-B: Images depicting variable site condition at the three CNPW outplanting sites featuring areas of consolidated substrate with turfing algae and sediment cover and low hard coral cover (photos: J.Gaskell, C.Hayward) (panel A); areas with greater hard and soft coral cover, some coral recruits and other benthic invertebrates (panel B); **C:** site condition at CNPW coral collection sites with higher cover soft and coral cover. **D:** Table nurseries installed at sites with larger coral fragments; a coral fragment wedged into the diamond-mesh of the nursery frame (fragments were subsequently secured with cable ties following predation); **E:** Coral fragment outplanted with Coralclip® deliberately oriented upright to prevent smothering by sediment.

3.3.3 Characterising baseline benthic composition.

During 'Site Setup', continuous line-intercept video transects (n = 3 x 30 m per site) were conducted (Howlett et al. 2022, 2023; Roper et al. 2023) in representative outplanting areas at CNPW outplanting sites and at coral collection sites to quantify baseline benthic composition (Fig. 1). Transect tapes were laid consecutively along the reef slope (5 – 15 m apart), perpendicular to the shoreline at 3-5 m depth. Using a GoPro HERO 9®, a diver filmed ~10-20 cm above the transect tape, capturing the substrate directly beneath it. During analysis, substrate directly under the transect line was recorded to the nearest 5 cm and categorised as: hard corals (identified to genus where possible), soft corals, macroalgae (including upright calcifying and fleshy macroalgae), consolidated substrate (rock), unconsolidated substrate (dead coral, coral rubble and sand), or other invertebrates (e.g., zooanthids, fire coral (*Millepora* sp.)). Notably, all abiotic hard surfaces at sites were covered in epilithic algae ranging in depth of cover of approximately 5-25mm (Fig. 3.2A). Fish community diversity was also captured alongside benthic surveys via parallel roving video surveys. Whilst these roving fish surveys recorded relatively similar family and trophic feeding groups across sites, and with greater total abundance observed at corresponding donor sites (Fig. S3.2), we note that more extensive fish community assays are needed to better assess site-specific dynamics and do not consider these data here further.

3.3.4 Evaluating coral outplant survivorship.

Documenting outplant survivorship has been a central means for tracking cost-effectiveness of coral restoration practices (e.g., Edwards 2010, Humanes et al., 2021; Mostrales et al., 2022; Scott et al., 2024; **Chapter 2**). Therefore, to benchmark initial coral outplant performance across the 'outplanting' sites, fate-tracked plots were established during initiation of the project (late August 2022), separate from areas designated for "routine " outplanting activity by operators. At each site, triplicate 5-7 m^2 control and treatment plots (n=6 total) were each marked with ~10 cm stainless steel rebar stakes and masonry nails for resurvey. Plots at BI and BPB were located at depths of 2-4 m, whereas plots at LB were at depths of 5-7 m, owing to suitable outplanting substrate availability. Each treatment plot was outplanted using Coralclip® with 60-80 coral fragments comprising different branching species: BPB (65 outplants ± 2 (mean ± SE outplants)), BI (70 outplants ± 1), LB (76 outplants ± 3). Lack of consistent coral material in sufficient quantity at 'collection' sites precluded full factorial replication by species, and hence 15-20 fragments of species of similar genera and/or growth morphologies were outplanted across sites: *Acropora millepora/Acropora spathulata, Acropora cerealis*, *Acropora intermedia/Acropora muricata, Pocillopora damicornis/Pocillopora verrucosa.* Coralclip® attachment success and survivorship of coral outplants was visually assessed 1-month post-outplanting (T32-days, September 2022) via visual SCUBA-based surveys, where observed coral fragments were counted by two divers and categorised as coral alive (fragment attached, <100% mortality), coral dead (fragment attached and covered in turfing algae, 100% mortality), and coral missing (empty Coralclip® still in place, but fragment missing) (as per Suggett et al., 2020; **Chapter 2**). Surveys were repeated five times over the next nine months at T67-days (November 2022), T109-days (December 2022), T191-days (March 2023), T232-days (April 2023), T267-days (June 2023).

3.3.5 Data analysis.

Statistical analysis and data visualisation were conducted in R (v4.0.0) (R Core Team 2021). All variables were visualised (qqplot and boxplots) and tested for normality (Shapiro-Wilk) and equal variance (Levene's test) prior to undertaking analysis. P-values and p_{adj}-values (Tukey's and Bonferroni) less than alpha ($\alpha = 0.05$) were considered significant for all tests. For benthic composition at sites, the cover of each benthic category (in cm) was expressed as a proportion of the total (3000cm) per replicate transect, and visually compared using a principal components analysis (PCA). To compare benthic composition profiles, separate one-way ANOVA tests were conducted on the extracted ordination axes for principal component 1 (PC1) and principal component 2 (PC2) between sites and site type (collection and outplanting). To specifically identify any differences in mean hard coral abundance (in cm) between sites, a subset of the dataset consisting of only the hard coral abundance data was tested with an additional one-way ANOVA, where significant differences between sites were identified with Tukey's HSD pairwise comparisons. To visualise mean proportional cover of hard coral genera at sites, stacked barplots were plotted, as were data on fish community characteristics (mean abundance by family and mean relative abundance by functional feeding group to visualise trophic composition).

For each fate-tracked plot at each site, duplicate counts of 'alive' outplants were averaged for each timepoint and survivorship was expressed as a proportion (between 0 and 1) in two ways (i) the number of corals observed alive at each timepoint relative to the original number outplanted (Strudwick et al., 2023) and (ii) relative to the total count of Coralclip® observations (alive, missing, dead) at corresponding survey timepoints (**Chapter 1**, Suggett et al., 2020). Both survivorship determinations are included as a number of original outplants could not be accounted for across sites during survey timepoints. For the former measure, (i) survivorship comparison between sites was conducted using a pairwise log-rank test of survival probability functions, derived from Kaplan-Meier survivorship curves, with counts of alive outplants observed in each plot at each timepoint as censored observations. P-values were adjusted by applying a Bonferroni correction. To determine differences in outplant survivorship between timepoints, the proportional change in survivorship at each timepoint was calculated (Δ alive) and arcsin-square-root transformed to meet parametric assumptions of normality, equal variances, and sphericity. As some negative values were present, a constant (1) was added to all values prior to square-root transformation. Transformed Δ alive values were then grouped by site and compared between timepoints with separate one-way repeated measures ANOVA tests. Following a significant interaction result, estimated marginal means were subsequently computed with the *emmeans* package (Lenth, 2023), on which paired t-tests with a Tukey p-value adjustment were performed to compare timepoint groups for each site. For the latter measure (ii) differences in Δ alive values (as proportion of total count per survey) were compared between timepoints using the same analysis described above. To compare survivorship between sites at the final timepoint (T-267 days), a final one-way ANOVA was conducted on arcsin-square-root transformed proportions, followed by Tukey's pairwise comparisons between sites.

3.3.6 Quantifying implementation costs.

Program costs were quantified from the beginning of grant funding in January 2022 through to the end of this current study (June 2023) and were grouped according to activity (following Edwards et al. 2010, and modifying Scott et al., 2024; **Chapter 2**): 'Coral material collection', 'Nursery installation', 'Nursery stocking and maintenance', 'Outplanting', 'Monitoring'. Additional categories were also included: 'Project planning and administration', 'Research', '*Ex-situ* training' and 'Travel and accommodation'. Costs incurred during each CNP 'event' (or trip) were partitioned by activity and categorised as (a) labour (b) vessel costs (c) consumables (d) capital equipment or (e) overheads (Edwards et al., 2010; Iacona et al., 2018). Labour costs were differentiated based upon the salary-level of the personnel conducting activity. Labour costs for the local coordinator and principal investigators were spread across the entire project and were thus included in 'overheads'. On trips where multiple activities were conducted during a single day by different divers, labour and vessel costs were partitioned proportionally based upon trip dive logs. Given program activity is ongoing, capital costs for equipment with repeat uses were costed once in their entirety at first use, rather than pro-rata use over time. All costs were calculated in 2022 Australian dollars ($AU) exclusive of GST (Goods and Services Tax). GST (10%) was subsequently added to final costs (except staff salaries in "Overheads"), which were converted to US dollars ($US) using the mean daily exchange rate between 1 January 2022 and mid-June 2023, where US$1.00 = AU$1.45 (MacroTrends, 2023).

Costs were first calculated with in-kind contributions included (e.g., labour costs for volunteers, research students, local coordinator, and principal investigator), which is likely a closer reflection of the 'true cost' of the intervention (Iacona et al., 2018; Hein & Staub, 2021). However, to examine the costs associated with in-kind time contributions and researcher involvement, costs were again calculated without these (e.g., 'Actual costs'). Finally, total costs were divided by the total number of corals outplanted during the study period, to first derive an overall planting cost (PC, $US. coral^{-1}; Scott et al., 2024; **Chapter 2**). The 'realised' cost of activity (PC_R, $US surviving coral^{-1}) was then estimated, whereby total outplant number was multiplied by the mean proportion of surviving outplants in fate-tracked plots across all sites, as well as by site-based survivorship at the final monitoring timepoint (T267-days). Full details of the assumptions of analysis, and cost calculations are provided in Supplement 3.2.

3.4.0 Results

3.4.1 Baseline benthic composition at CNPW outplanting and collection sites.

Benthic composition did not differ significantly between CNPW outplanting sites (Tukey's post-hoc, $p_{Tukey} > 0.05$; Table S3.2b). However, PCA visualisation of benthic composition showed discrete clustering between CNPW outplanting and collection sites with some overlap (Fig. 3.3C) suggesting differences in benthic communities. Principle component 1 (PC1) and 2 (PC2) accounted for 67.1% and 22.1% of the total variance in benthic cover respectively. ANOVA on these extracted ordination axes confirmed that collection sites significantly differed from outplanting sites along PC1 (ANOVA, $F_{1,16} = 42.73$, $p < 0.001$), but not PC2 (ANOVA, $F_{1,16} = 0.675$, $p = 0.42$; Table S3.3). The greatest loadings contributing to differences along PC1 were consolidated substrate, soft coral, and hard coral cover (Table S3.4). Mean hard coral cover was significantly higher at Cockatoo Point (CP) at 38.06 ± 4.91% (± SE) of total cover (Fig. 3.3A) compared to collection site Luncheon Bay Donor (LBD) and the three outplanting sites (Fig. S3.3; ANOVA, $F_{5,12} = 9.047$, $p_{Tukey} < 0.001$; Table S3.5). Mean hard coral cover was 17-22% at the other two collection sites (Wonderwall (WW) and LBD). Whereas at the outplanting sites, hard coral cover was 3.22 ± 0.87 %, 7.56 ± 3.42% and 8.67 ± 4.67% at Luncheon Bay (LB), Blue Pearl Bay (BPB) and Black Island (BI) respectively (Fig. 3.3A), and did not significantly differ between these sites (Fig. S3.3; Tukey's post hoc, $p_{Tukey} > 0.05$; Table S3.5). Hard coral cover at outplanting sites was largely composed of genera with massive, submassive and encrusting morphologies with low structural complexity (Fig. 3.3B). Total benthic cover of *Acropora* genera corals at outplanting sites was typically <0.6%, whereas at collection sites, *Acropora* genera comprised 8.24 ± 1.87% (mean ± SE) of total benthic cover. Turfing algae cover was present on consolidated rock at all sites but was lowest at collection site LBD (R. Scott, personal observation, August 2022). Macroalgae cover was only observed at outplanting sites BPB and LB, but not BI (Fig. 3.3A).

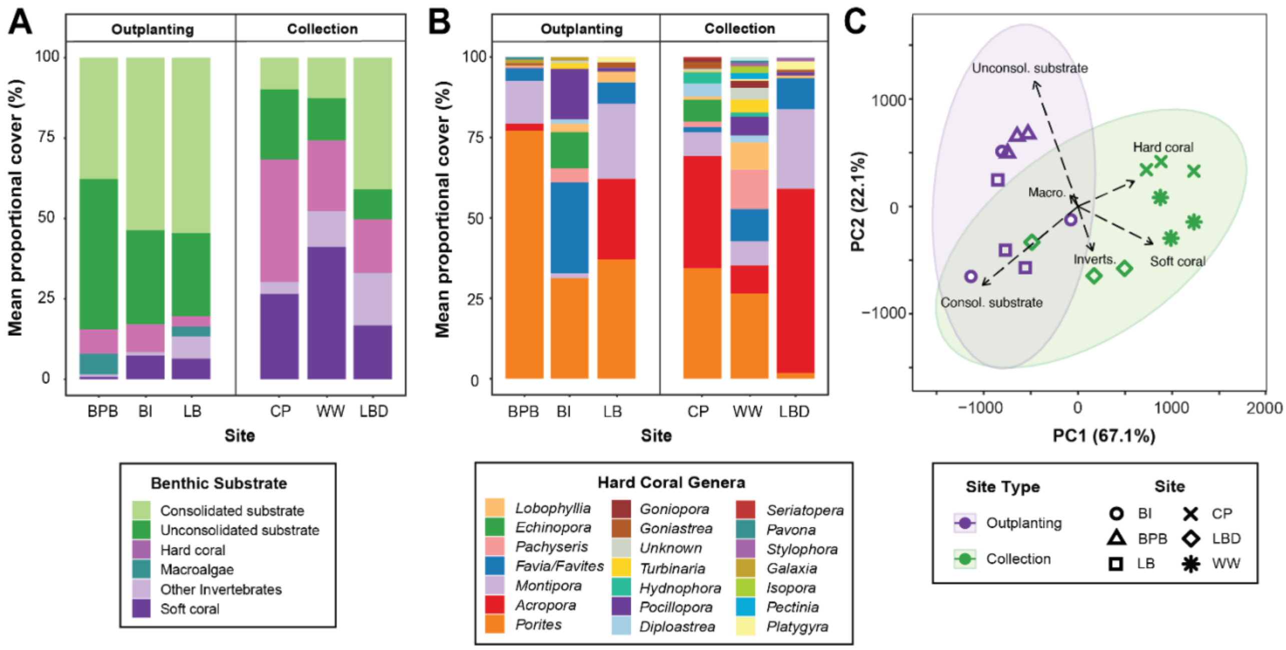

Figure 3.3. A) Mean proportional coverage of benthic substrates at Coral Nurture Program Whitsundays (CNPW) outplanting (BPB: Blue Pearl Bay, BI: Black Island, LB: Luncheon Bay) and collection (CP: Cockatoo Point, WW: Wonderwall, LBD: Luncheon Bay Donor) from triplicate 30m benthic video transects in September 2022. **B)** Mean cover of hard coral genera* as a proportion of hard coral coverage at

CNPW outplanting and collection sites. *Based upon capacity to identify corals from videos, *Favia* and *Favites* are conservatively grouped together **C)** Principal components analysis (PCA) of benthic categories grouped by site type (outplanting and collection). Ellipses show 95% CI. PCA loadings of benthic categories (shown as dashed arrow vectors) were scaled to PCA eigenvalues, with vector length indicating the strength of this contribution. Vector direction shows the contribution of each variable to the principal components (PCs): Inverts: other invertebrates, Macro: fleshy and upright calcifying macroalgae, Consol. susbtrate: consolidated hard coral rock covered in turfing algae of varying depths, Unconsol. substrate: unconsolidated sand, coral rubble and dead coral.

3.4.2 CNPW nursery propagation and outplanting activity.

During the study period, a total of 4,425 coral fragments were collectively outplanted by CNPW tourism operators, CNP researchers and volunteers at the three CNPW sites (Table 3.1). In total, 15 staff members across Operator A-C, and nine additional volunteers from other Whitsundays' tourism operators were trained in outplanting with Coralclip®. Between 25-30 different coral species were outplanted across the three sites, of which 68-87% were *Acropora* spp. and 10-18% were *Pocillopora* spp. (detailed in Table 3.2). Other branching coral genera including *Echinopora, Porites, Stylophora* and some genera with encrusting and massive morphologies were also outplanted.

Nursery frames across sites were stocked with 15-21 different species of coral, of which approximately 65% were from the genus *Acropora* with other genera including *Echinopora, Isopora, Montipora, Pocillopora, Porites,* and *Turbinaria* (detailed in Table S3.6). After 1-month some coral fragments were observed to self-attach to the nursery frame, but several colonies were lost (potentially due to initially being unsecure and/or predated upon). Consequently, frames were restocked with 64, 37 and 32 new colonies (secured to frames with cable-ties) in March 2023 at BI, BPB and LB respectively. To allow coral colonies on nurseries time to establish, no nursery corals were removed from frames for outplanting. Furthermore, 1-month post-establishment, sponges, ascidians, turfing and filamentous macroalgae had colonised nursery frames which smothered some smaller fragments, and sediment was observed accumulating on plating/foliose colonies (e.g., *Turbinaria*). As per permitting requirements, any diseased and dead colonies were removed from frames during the six monitoring trips conducted between September-June 2023 and filamentous algae was removed from any affected colonies. However, time-limits on these trips precluded intensive cleaning of nursery frames. Frames were intensively scrubbed of fouling organisms at each site during nursery restocking trips in February/March 2023 and again in May/June 2023.

Table 3.1. Number of coral fragments at each site collectively planted by UTS researchers, the three tourism operators and volunteers during three outplanting deployments from September 2022 - June 2023. Corals outplanted during "Site setup" include outplants in fate-tracked plots.

Deployment	Blue Pearl Bay	Black Island	Luncheon Bay	Total
"Site Setup" (August 2022)	673	523	422	1,618
"Monitoring and Training" (March 2023)	351	275	462	1,088
"Coralpalooza™" (June 2023)	460	644	615	1,719
Total	**1,484**	**1,442**	**1,499**	**4,425**

Table 3.2. The number of different coral species and relative abundance (%) of total outplants by coral genera outplanted at the three CNPW sites from August 2022 – June 2023. 'Sp./Spp.' denotes where coral species could not be identified. "Other" denotes coral genera contributing to <1% of total outplant number.

Site	Number of species outplanted	Coral species outlanted	Relative abundance (%) of outplants by coral genera
Blue Pearl Bay	27	*Acropora abrolhosensis, A. aculeus, A. cerealis, A.digitifera, A. elseyi, A. horrida, A. intermedia, A. latistella, A. millepora, A. muricata, A. nasuta, A. pectinata, A. selago, A. subulata, A. tenuis, Acropora spp., Echinopora horrida, Hydnophora rigida, Pavona cactus, Pavona sp., Pocillopora acuta, P. damicornis, P. verrucosa, Porites cylindrica, P. negrecians, P. rus, Stylophora pistillata.*	*Acropora* (67.82%), *Pocillopora* (18.17%), *Echinopora* (7.56%), *Porites* (3.12%), *Pavona* (1.73%), Other (2%)
Black Island	30	*Acropora abrolhosensis, A. carduus, A. cerealis, A. elseyi, A. florida, A. horrida, A. intermedia, A. latistella, A. longicyathus, A. loripes, A. micropthalma, A. millepora, A. muricata, A. spathulata, A. verweyi, Acropora spp., Echinopora horrida, Favia sp., Favites sp., Hydnophora rigida, Lobophyllia sp., Montipora sp., Pachyseris sp., Pectinia sp., Pocillopora acuta, P. damicornis, P. meandrina, P. verrucosa, Porites cylindrica, Stylophora pistillata*	*Acropora* (71.56%), *Pocillopora* (15.77%), *Porites* (4.51%), *Echinopora* (4.38%), *Stylophora* (1.95%), Other (2%)
Luncheon Bay	25	*Acropora abrolhosensis, A. abrotanoides, A. cerealis, A. elseyi, A. florida, A. gemmifera, A. humilis, A. hyacinthus, A. intermedia, A. micropthalma, A. millepora, A. muricata, A. pectinata, A. selago, A. spathulata, A. tenuis, A. valida, A. yongei, A. spp., Echinopora horrida, Montipora sp., Pocillopora damicornis, P. meandrina, Porites cyclindrica, Stylophora pistillata.*	*Acropora* (86.59%), *Pocillopora* (10.27%), *Echinopora* (1.80%), Other (1%)

3.4.3 Coral outplant survivorship in fate-tracked plots.

When assessed relative to the number of corals originally planted, mean survival of outplants after nine months (T267-days post-planting) was higher at BI (47.58 ± 3.56%) than LB and BPB, which was 25.70 ± 2.31% and 23.33 ± 2.96%, respectively (Fig. 3.4, Table 3.3). Throughout this study, mean outplant survival was between 9.46-40.48% higher at BI (across timepoints) than the other two sites. Kaplan–Meier survival curves were significantly different between BI and BPB (Pairwise log-rank, $p_{Bonferroni} < 0.001$), BI and LB (Pairwise log-rank, $p_{Bonferroni} < 0.001$), and LB and BPB (Pairwise log-rank test, $p_{Bonferroni} = 0.035$ (Fig. S3.4; Table S3.7-S3.9), though survivorship probabilities between BPB and LB did not significantly differ from T109-days onwards (Fig. S3.4; Pairwise log-rank test, $p_{Bonferroni} > 0.05$; Table S3.9). This was confirmed for all sites, where no significant differences were observed in the latter timepoints for proportional change in survivorship values (Δ alive) (Tukeys post-hoc $p_{Tukey} > 0.05$; Table S3.10 – Table S3.12), suggesting outplant mortality stabilised after four months (T109 days) in the period of study.

The same trend was observed when outplant survivorship was assessed relative to the total count of Coralclip® observations (Fig.S3.5; Table S3.13), although in this instance, mean survivorship estimates were substantially higher (i.e., 45-71% at T267-days) (Table 3.3; Fig.S3.5), and mortality stabilised at LB after T191 days (Δ alive; Tukey's post hoc, p >0.05; Table S3.16).

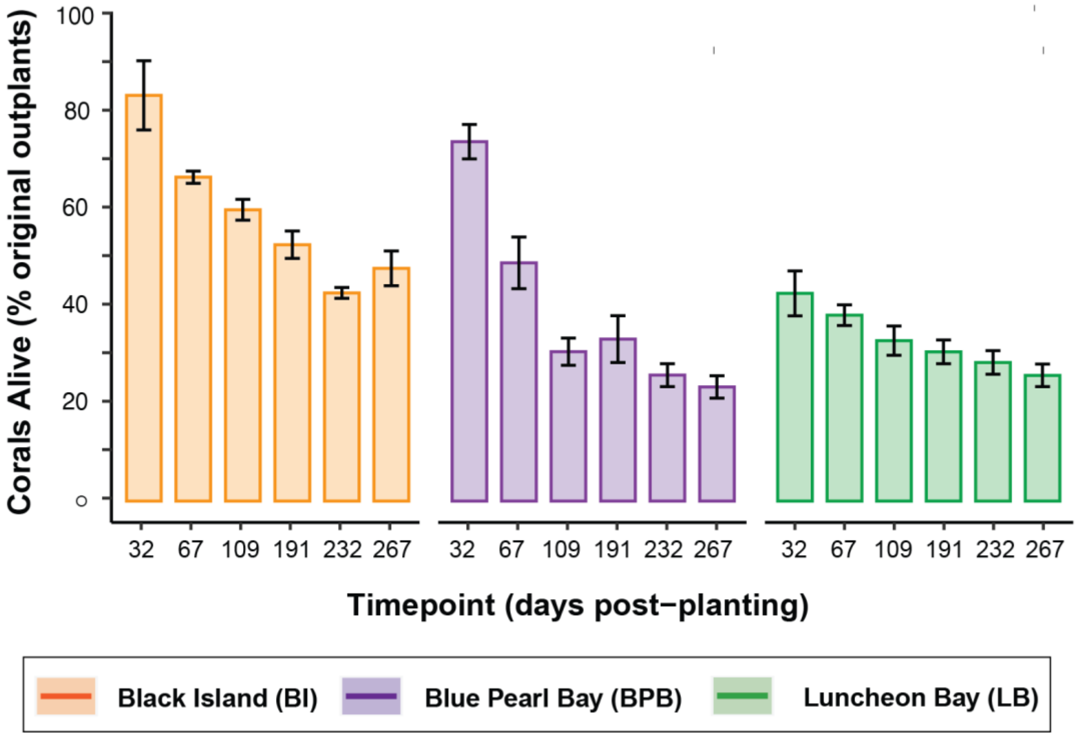

Figure 3.4. Mean (± standard error) proportion (%) of coral outplants 'alive' at each timepoint relative to the original number outplanted in triplicate fate-tracked plots at each site. Apparent increases in survivorship at BI at T-267 and BPB at T-191 were the result of observer error, but changes in survivorship were not significantly different from preceding and later timepoints (Table S3.11 & S3.12).

Table 3.3. Outcomes of coral fragments outplanted with Coralclip® in triplicate fate-tracked plots at the three CNPW sites after nine months (T267-days). Shown is the mean proportion (± standard error) of corals 'alive', 'dead', and 'Coralclip® empty' relative to (a) the number of original corals planted in plots and (b) the total count of observations in the final survey. 'Corals unaccounted' is the proportion of original outplants that could not be accounted for.

(a) Outplant Survivorship at 9 months (as proportion of original outplants)			
	Black Island (BI)	Blue Pearl Bay (BPB)	Luncheon Bay (LB)
% corals alive	47.58 ± 3.56	23.33 ± 2.29	25.69 ± 2.31
% Coralclip® empty	17.21 ± 2.39	29.52 ± 5.16	22.80 ± 5.42
% corals dead	2.40 ± 0.50	0.00	4.23 ± 2.22
% corals unaccounted	32.81 ± 4.12	47.15 ± 5.89	47.28 ± 6.17
(b) Outplant survivorship at 9 months (as proportion of total count)			
	Black Island (BI)	Blue Pearl Bay (BPB)	Luncheon Bay (LB)
% corals alive	70.87 ± 3.88	44.82 ± 4.48	49.12 ± 2.03
% Coralclip® empty	25.57 ± 3.15	55.18 ± 4.48	42.06 ± 5.32
% corals dead	3.56 ± 1.26	0.00	8.82 ± 7.67

Difference in survivorship estimates between determinations – i.e., as a proportion of original outplant number *versus* a proportion of total count at the final time point – was primarily the result of the proportion of 'corals unaccounted' (Table 3.3), with 33-47% of original Coralclip® units unaccounted for across sites after 9 months. Whilst the cause of this high proportion of 'unaccounted' Coralclip® units is unclear at this time, it can reflect an underestimation of 'alive' corals (where outplants had overgrown Coralclip®; Scott et al., 2024; **Chapter 2**), of 'empty' Coralclip® units because of smothering by turfing algae and sediment, and/or complete detachment from the substrate (Fig.3.5A-B). 'Empty' Coralclip® observations steadily increased over time at all sites (Fig.S3.6), indicating that declining survivorship at sites was likely driven by coral detachment rather than mortality of attached outplants (Table 3.3). After nine months, branching (open and closed morphologies) *Acropora* species showed higher survivorship at BI than the other two sites, whereas *Pocillopora* outplants showed higher survivorship at LB (Figure S3.7).

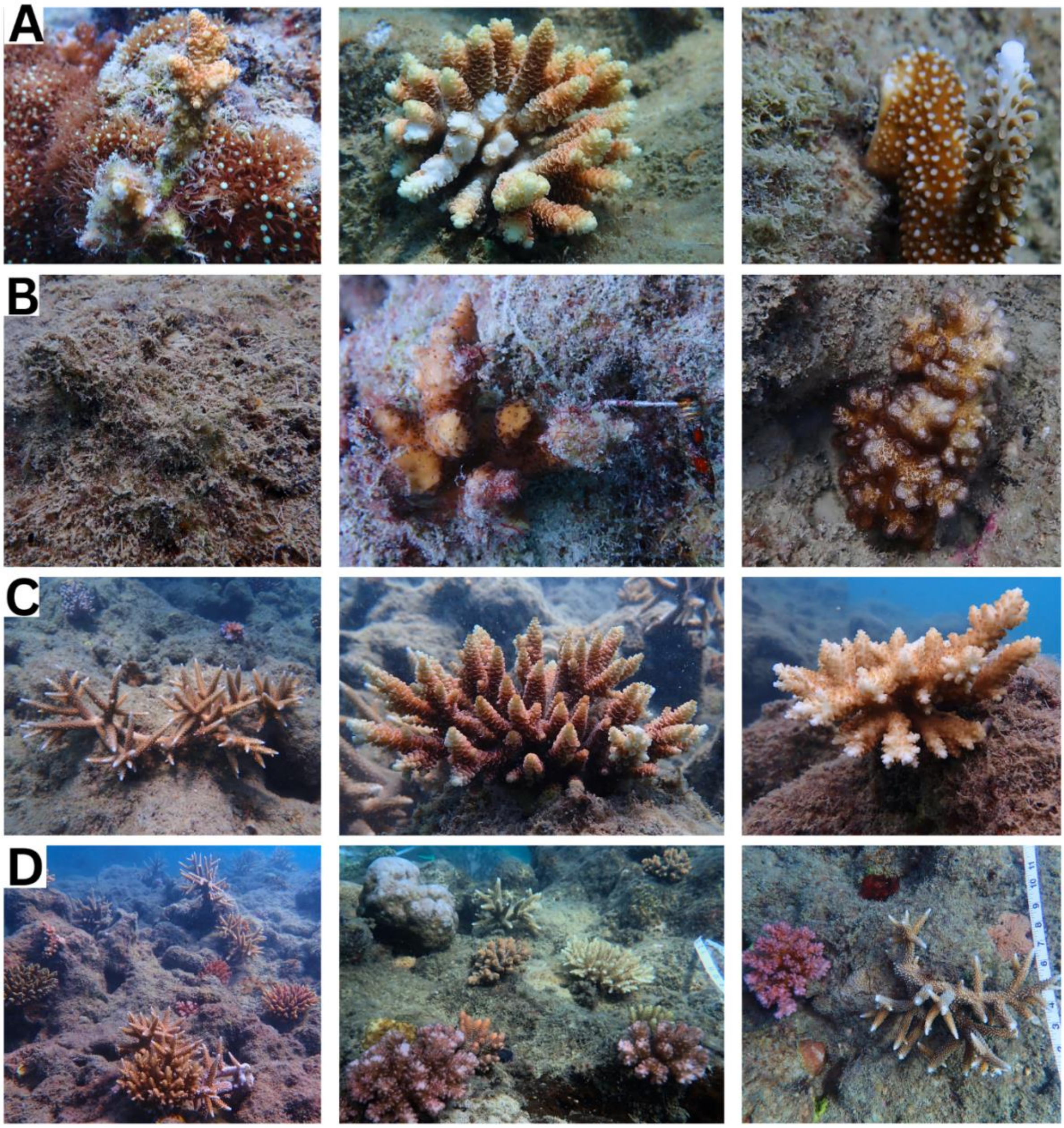

Figure 3.5. Images depicting fragments outplanted in triplicate fate-tracked plots across the three CNPW sites. **A:** depicts challenges experienced by coral outplants including competition with other benthic invertebrates (e.g., zooanthids, left), predation by corallivorous fish (middle) and difficulty self-attaching to algal-turf covered substrates (right). **B:** depicts challenges in assessing outplant survivorship including smothering of empty Coralclip® units by turfing algae (left), and detachment of outplants from substrate (middle, right). **C-D:** depict surviving outplants after nine months in June 2023 (shown left to right).

3.4.4 Implementation costs and realised costs.

Total initial implementation and maintenance costs for the first nine months of all coral propagation and outplanting activity at the three Whitsundays sites was US$253,800.38 ('true cost'; in-kind costs included). Based on the number of corals outplanted during this timeframe, this capital and operational expenditure yields an effective planting cost (PC) of \$57.36 coral^{-1} (n = 4,425) (Table 3.4). However, if only 'outplanting' costs were considered (e.g., as per Scott et al., 2024; **Chapter 2**), PC during this timeframe was \$10.63 coral^{-1} (Table S3.18 and Supplement S3.1). Overall, 'vessel use' and 'overheads' were the cost categories which accounted for the greatest contributions to total 'True costs' (30% and 48% of costs, respectively, Table 3.4). As such, when in-kind costs associated with 'overheads' (researcher and local coordinator time) as well as volunteer labour were *not* included (i.e., 'Actual costs'), total costs were 44% less (\$143,549.05), yielding a PC of US\$32.44 coral^{-1}. The costliest activities contributing to the large discrepancy between 'true' PC and 'actual' PC were 'Project Planning, Management and Administration' activities (49% of the total) due to high overhead costs. This was followed by 'Outplanting', 'Coral material collection' and 'Monitoring' activities which accounted for 19%, 10% and 7% of costs respectively (Table 3.4), as these activities required the most labour and the greatest proportion of vessel time. Travel and accommodation costs for UTS researchers to travel to Airlie Beach for activities accounted for 6% of total costs.

Further adjusting 'True costs' by the mean 9-month (at T-267 days) survivorship of outplants in fate-tracked plots (Table 3.3a) resulted in a realised cost (PC_R) of \$178.12. coral^{-1} overall but ranging \$120.55 - \$245.85. coral^{-1} depending on site-based survivorship (Table 3.5). Again, when only 'Outplanting' costs were considered (e.g., as per Scott et al., 2024; **Chapter 2**), PC_R was \$33.04 coral^{-1} (Table S3.18 and Supplement S3.1). When in-kind costs were excluded from total costs ('Actual cost'), PC_R was US\$100.75 (\$68.18 - \$139.05) (Table 3.5). When PC_R estimates were calculated using survivorship expressed relative to total count during surveys (rather than relative to original outplants (Table 3.3b), PC_R was substantially lower (\$80.93 - \$127.97 for 'True costs', 45.77 - \$72.38 for 'Actual costs') (Table S3.17). highlighting the value of robustly fate tracking all initial outplants. Collectively this approach demonstrates how the methods with which costs are calculated and survivorship is assessed substantially impacts PC and PC_R.

Table 3.4 Costs ($US) of Coral Nurture Program Whitsundays (CNPW) implementation over nine months by cost category (top row) and activity (first column). Proportional contribution of each category and activity to the overall total project cost ("True cost") is also presented. Cells with '-' indicate where no cost was incurred. Planting Cost (PC, $US coral^{-1}) is the total costs relative to the 4,425 coral outplants deployed September 2022 – June 2023. Full costings are presented in Supplement 2.2.

Activity	Labour	Capital	Vessel	Consumables	Overheads	Overall Total ("True" Cost)	Total "Actual" cost (less in-kind)	% Overall Total
Project Planning/ Management/ Administration	$379.50	-	$2,277.00	-	$121,833.96	$124,490.46	$24,165.72	49.05%
Coral Material Collection	$4,520.79	$502.71	$16,212.24	$3,347.19	-	$24,582.93	$23,083.91	9.69%
Nursery Installation	$474.38	$9,191.57	$1,912.68	$7.42	-	$11,586.05	$11,111.68	4.57%
Nursery Stocking & Maintenance	$1,691.15	$15.18	$5,897.43	$1,260.61	-	$8,864.37	$8,584.48	3.49%
Outplanting	$11,460.99	$1,366.82	$31,786.92	$2,459.16	-	$47,073.90	$41,815.35	18.55%
Monitoring	$3,356.30	$67.44	$13,635.44	-	-	$17,059.18	$15,676.28	6.72%
Research	$1,091.06	$5.13	$4,144.14	$101.14	-	$5,341.47	$4,667.86	2.10%
Training	$358.25	-	-	$22.77	-	$381.02	$22.77	0.15%
Researcher Travel/ Accommodation	-	-	-	$14,421.00	-	$14,421.00	$14,421.00	5.68%
% Overall Total	9.19%	4.39%	29.89%	8.52%	48.00%			
Grand total						**$253,800.38**	**$143,549.05**	
$US coral^{-1} (PC)						**$57.36**	**$32.44**	

Table 3.5. Realised costs (PC_R, $US surviving coral^{-1} of CNPW implementation relative to outplant survivorship (as a proportion of original outplants, Table 3a) in fate-tracked plots after nine months (T6, 267 days).

Mean Survivorship	PC_R (True Cost)	PC_R (Actual Cost – less in-kind)
Overall (32.20%)	$178.15	$100.77
BI (47.58%)	$120.56	$68.20
BPB (23.33%)	$245.88	$139.09
LB (25.70%)	$223.21	$126.26

3.5.0 Discussion

Further investment and scaled application of active restoration interventions, including on Australia's Great Barrier Reef (GBR), hinges upon addressing uncertainties around the feasibility of approaches for different reef environments (McLeod et al., 2022). Arguably, the central factor underpinning reef restoration feasibility is comprehensive understanding of the associated costs – including those potentially obscured as in-kind contributions or project overheads (Edwards et al., 2010; Iacona et al., 2018; Bayraktarov et al., 2019) – and the likelihood of 'success' in terms of delivering on program goals (Bayraktarov et al., 2019). However, the novelty of restoration-based management approaches on the GBR means that reports on restoration outcomes (e.g., Howlett et al., 2021, 2022, 2023; Roper et al., 2022; Smith et al., 2022; Randall et al., 2023) and costs (Suggett et al. 2020; 2023; Scott et al., 2024 **(Chapter 2)**) are only just beginning to emerge. Such knowledge is lacking for the high-value, fringing reefs in the Whitsundays region, where coral restoration activity was recently implemented via the Coral Nurture Program (CNP) tourism site stewardship approach previously established in the Cairns-Port Douglas region (Howlett et al., 2022; Scott et al., 2024; **Chapter 2**). We therefore tracked restoration costs from program outset to conduct a more robust cost evaluation exercise for the Coral Nurture Program Whitsundays (CNPW). In turn, through evaluating initial outplant survivorship, we examine the early-stage cost-effectiveness of CNP implementation in the Whitsundays. Returned costs were higher than those previously assessed for CNP Cairns-Port Douglas but were within reported values in reef restoration projects globally. We discuss the costs and cost-effectiveness of CNPW relative to the local operational and environmental context and discuss the importance of comprehensive cost-tracking to support decision-making processes in reef restoration.

3.5.1 Comprehensive cost-tracking of all restoration activities is essential to capture "true costs".

Through comprehensive cost-tracking of CNPW from the planning phase, we show that the costs of outplanting represent only a proportion of total project costs (albeit significant: ~20% of costs here and ca. 30-50% of project costs elsewhere, e.g., Edwards et al., 2010; Toh et al., 2017; Humanes et al., 2021) (Table 3.4, Table S3.18). This is consistent with findings for CNP Cairns-Port Douglas where we previously determined that outplanting costs (~ca. US$2.30 coral^{-1} trip^{-1} from >30,000 outplants) increased 2-to-6-fold where time allocation to additional and essential nursery propagation, site maintenance and practitioner training was accounted for in addition to outplanting cost (Scott et al., 2024; **Chapter 2)**. Whilst outplanting-only costs for CNPW in this early stage (\$10.63 coral^{-1}) still remained higher than those for more established activity via CNP (Table S3.18), the total “true” restoration costs for early-stage activity quantified here (US\$253,800.83, \$57.36 coral^{-1}) are similar to the median cost previously determined in a global review by Bayraktarov et al. (2019) for 20 coral propagation and outplanting projects (2010 US\$218,305 ha^{-1} yr^{-1}), although we have not derived a per-hectare cost in this current exercise. Costs for CNPW are further comparable to programs of a similar working size (i,e., number of reef sites, nurseries and outplants) in Latin America (e.g., Corales de Paz, Colombia, Sociedad Ambiente Marine, Puerto Rico; Baykratarov et al., 2020), and ‘realised’ costs for corals outplanted on a seawall in Singapore (US\$122.80 coral^{-1}; Toh et al., 2017) under similar ecological and socioeconomic conditions (e.g., inshore location with high sedimentation, ‘high income’ country (Bayraktarov et al., 2019)).

Our findings further demonstrate how significant the costs associated with essential program management, administration, and planning are to restoration projects and highlight the gravity of in-kind contributions of volunteer, stakeholder, and researcher time (representing almost 50% of “true costs”). For large-scale coral propagation and outplanting programs in the Maldives (Montoya-Maya et al., 2016, reported in Baykratartov et al., 2019) and the Florida Keys (Coral Restoration Foundation, 2023), costs involved in project overheads, fundraising, research, and development similarly accounted for a significant proportion of total expenses (approximately 20-40%). Elsewhere on the GBR, restoration activity on Fitzroy Island has been enabled through in-kind contributions of time, labour, and support totalling AU\$150,000/year (McCleod et al., 2022). In the instance of CNPW, collaboration with a research institution and a local natural resource management organisation meant that significant overhead expenses were not a true cost to the Program, and hence “actual costs” were 44% less. This highlights how essential in-kind contributions are to cost-effective restoration, which have been shown to effectively halve project costs in reef regions globally (Bayraktarov et al., 2020; dela Cruz et al., 2014; Suggett et al., 2020; Toh et al., 2017). However, such critical costs are often ‘invisible’ in restoration project costings (where costs are reported) (Iftekhar et al., 2017), potentially owing to an absence of comprehensive cost-tracking capability in stakeholder programs (Iacona et al., 2018; Ferse et al., 2021; Scott et al., 2024; **Chapter 2**) or publishing bias towards successful or low-cost interventions to access competitive grant funding (Edwards et al., 2010; Bayraktarov et al., 2015; Boström-Einarsson et al., 2020). Such under-reporting ultimately disadvantages collective restoration practice, through inhibiting adequate investment and effective budget forecasting for sustained restoration (Suggett et al., 2023) or project initiation elsewhere (Edwards et al., 2010).

3.5.2 Survivorship-based 'success' varies by site.

When reported, coral restoration costs are typically weighted relative to outplant survivorship to yield a cost per surviving coral (referred to here as "realised cost", **Chapter 2;** see also Edwards et al., 2010; Toh et al., 2017; Bayraktarov et al., 2019; Harrison et al., 2021; Humanes et al., 2021). Although outplant survivorship is not a holistic indicator of ecological "success" (Hein et al., 2017, 2021; Ladd et al., 2019; Boström-Einarsson et al., 2020; Goergen et al., 2020), this metric provides a good indication of technique feasibility in the early-stages of program establishment (i.e., within the first year) and can inform adaptive practice. Our data revealed substantial differences in cost-effectiveness estimates (PC_R) depending on the method used to quantify survivorship (%), underscoring the importance of transparency and consistency in monitoring (Goergen et al., 2020) and reporting of restoration outcomes and cost-effectiveness across projects (Bayraktarov et al., 2019; Eger et al., 2022)

Survivorship of fate-tracked outplants at CNPW sites after nine months (23-48%), was lower than we have observed previously for similar fate-tracked outplants for CNP Cairns-Port Douglas (which range 32-93% in the first year) (Howlett et al., 2022; Strudwick et al., 2023; Scott et al., 2024 (**Chapter 2)**). As such, CNPW realised costs (PC_R) increased substantially to >$US100 coral^{-1} (ranging $46 - $180 depending upon site, survivorship assessment approach and inclusion of in-kind costs) based on full costs (Table 3.4, 3.5, S3.17) or US$33.04 coral^{-1} based on outplant costs only (Table S3.18). This contrasts with outplanting PC_R for CNP (Cairns-Port Douglas) of $2.94 ± 0.23 coral^{-1} trip^{-1} (Table S3.18). Such higher 'realised' costs for CNPW are perhaps unsurprising given that poor water quality and proximity to land has been associated with lower outplant survival in reef regions globally (Foo & Asner, 2021). In the Whitsundays region water quality has challenged coral growth and recovery (Thompson et al., 2023) and a previous outplanting effort in the region similarly noted the challenges of turfing algae competition and sedimentation on successful outplant self-attachment and growth (Cook 2022). In other reef regions, outplanting studies at sites impacted by poor water quality have reported variable outplant survivorship estimates of between 40-80%, often depending upon species (e.g., Ferse et al., 2013 in Indonesia; Horoszowski-Fridman et al., 2015 in Eilat, Egypt, Bayraktarov et al., 2019 in Costa Rica; Toh et al., 2017 in Singapore). In our study, survivorship was not consistently higher for a particular species/morphology across sites, which underscores the importance of ensuring diversity (both genotypic and taxonomic) in outplanted coral assemblages to maximise survivorship and resilience of corals adapted to different conditions (Baums et al., 2019; Quigley et al., 2022; Shaver et al., 2022; Madin et al., 2023). Species that may perform better under high sediment and nutrient loads in the Whitsundays (e.g., encrusting. massive and foliose morphologies; (Anthony & Fabricius, 2000; Toh et al., 2017; Morgan et al., 2020; Thompson et al., 2023), are not currently preferred for Coralclip® use, highlighting the need to further tailor solutions for outplanting in this region.

Outplant survivorship displayed distinct site-differences, with nine-month outplant survivorship at BI significantly greater than LB and BPB. Such variability is consistent with previous site-based evaluations of coral outplants via CNP Cairns-Port Douglas (Howlett et al., 2022; Strudwick et al., 2023; **Scott et al., 2024; Chapter 2**). Furthermore, in the Whitsundays, Cook (2022) reported site-differences in six-month survivorship of corals propagated in *in-situ* nurseries (65% at CNPW site, BPB and 72% at Manta Ray Bay, adjacent to CNPW site, LB). Variable survivorship and attachment of corals in restoration can be influenced by many factors including prevailing abiotic conditions (e.g., light availability, currents and temperature; Ware et al., 2020; Strudwick et al., 2023), fish predation and dislodgement (Frias-Torres & van de Geer, 2015; Horoszowski-Fridman et al., 2015; Seraphim et al., 2020), corallivore predation (Cabaitan et al., 2015; Knoester et al., 2023), substrate quality (Ferse, 2010) and coral genotype (Ladd et al., 2017). In this study, regardless of site, coral 'mortality' was primarily explained by coral loss from Coralclip® and a large proportion of original outplants that were not accounted for during surveys. Surviving fragments were occasionally found nearby outplant areas, often still self-attached to the Coralclip®, highlighting how careful selection of consolidated substrate and rigorous removal of turfing algae is essential to the success of this method. Similar rates of detachment have been observed at certain CNP sites in Cairns-Port Douglas **(Scott et al., 2024; Chapter 2)**, and elsewhere during early-stage outplant establishment (e.g., 30%, in Horoszowski-

Fridman et al., 2015). Notably, abiotic and biotic conditions that regulate survivorship can occur at very fine spatial scales (e.g., 10s of metres, Randall et al., 2023), and thus whilst not examined here, further investigations are needed to understand the drivers of mortality and detachment at CNPW sites (e.g., seasonal algae growth, Brodie et al., 2012). Such knowledge will enable enhanced outplant site and species selection within and between sites of the Whitsundays to improve survivorship through iterative learning. Hence, ongoing monitoring will be essential to improving restoration cost-effectiveness in this region.

3.5.3 Operational-environmental context influences cost-effectiveness of coral restoration.

Costs and cost-effectiveness of coral restoration are highly specific to local context, including location, scale, and restoration method (Bayraktarov et al., 2015, 2019), project goals (Hein et al., 2021) and outcomes (e.g., time-specific survivorship; Harrison et al., 2021). For the CNPW, it was apparent that the CNP site stewardship model, originally conceived for seamless integration of coral propagation and outplanting activity into routine tourism operations (Howlett et al., 2022), required adaptation to the novel tourism operational and environmental context in the Whitsundays. For example, low and patchy coral cover at sites targeted for CNPW outplanting required coral collections from more abundant and diverse adjacent sites. These ecological conditions, combined with operational factors such as smaller CNPW operator vessels requiring fewer crew (as compared to Cairns and Port Douglas operations, Scott et al., 2024; **Chapter 2)**, meant that CNPW activity during routine tourism trips was largely limited to visual monitoring of restoration sites and nursery structures, precluding regular, *ad-hoc* outplanting and nursery maintenance. Furthermore, the shared, public nature of Whitsundays' vessel moorings meant that Operator A-C visitation to CNPW outplanting sites during "routine" tourism operations was less frequent than in Cairns and Port Douglas (Howlett et al., 2022) where moorings are largely private. This ability to spatially diversify tourism operations was likely critical to tourism resilience and adaptation in the Whitsundays post-Cyclone Debbie (Bartelet et al., 2023), and provides an important economic rationale for assisted recovery of CNPW sites to enable further spatial operational diversification in future. However, such factors collectively impacted coral outplanting output and had substantial cost implications. For example, 30% of expenditure went toward 'dedicated' vessel use, and costs for coral collection at donor sites accounted for ca. 10% of total costs. These represent common and significant costs in coral restoration (e.g., Edwards et al., 2010) that by comparison are not typical for CNP operations in Cairns-Port Douglas

(Howlett et al., 2022, Scott et al., 2024; **Chapter 2).** As such, noting differences in costing methodology (related to planting and diving gear calculations, currency year etc., see Supplement S2), there was almost a five-fold difference in outplanting-only PC estimates across the two programs (Scott et al., 2024; **Chapter 2**) which was largely the result of CNPW vessel cost requirements. Despite contextual differences, several interacting factors previously identified in Cairns and Port Douglas operations that regulate PC and $PC_{R,}$ such as underlying site condition and source of coral material (Scott et al., 2024; **Chapter 2**), were also at play in the Whitsundays.

The current study provides cost-analysis at the early-stage of CNPW, and further cost-tracking will be needed to assess whether PC and PC_R increase or decrease with increasing scale of operations (although evidence for 'economies of scale' in coral restoration is not yet apparent; Bayraktarov et al., 2015; Hughes et al., 2023), disturbance severity, operator experience and adaptive practice leading to improved survivorship (Iacona et al., 2018). The upfront costs of program establishment (e.g., nursery installation, coral collection, planning, researcher travel for training and monitoring) were significant. PC will likely decline as the program transitions from 'launch' phase to sustained operations, with reduced need for researcher involvement and training, and establishment of coral nursery colonies that provide a self-sustaining source of coral material. While costs and cost-effectiveness may be perceived as high, it is important to reiterate that the low underlying coral cover at high-value CNPW tourism sites may justify intensive efforts to improve site conditions. For example, if considered relative to the sites' estimated million dollar/km^2 tourism ecosystem service value (Spalding et al., 2016; De Valck & Rolfe, 2018; Suggett et al., 2023), costs incurred may deliver positive cost-benefit in retaining such value ((Naidoo et al., 2006). Further work is needed to confirm this notion via detailed cost-benefit analyses and longer-term, goal-based ecological monitoring that can capture ecological changes underpinning ecosystem function, resilience, and associated ecosystem service value (e.g., Hein et al., 2017; Ladd et al., 2019; Goergen et al., 2020). Future longer-term cost assessments may also consider transitioning from a per unit (e.g., 'realised cost' based on \$US $coral^{-1}$) to a per area (e.g., 'cm' or % change in coral cover) as a complimentary, and likely more representative measure of cost-effectiveness (Suggett et al., 2023; Bayraktarov et al, 2020). Although long-term monitoring and cost-tracking necessitates ongoing costs (Edwards et al., 2010; Hein & Staub, 2021), they are essential to goal-based evaluations of cost-benefit.

Although not captured in this study, wider socioeconomic benefits were evident in the adaptation of the CNP model to the Whitsundays and are an important aspect that future costing exercises should consider (e.g., through a social-ecological system (SES) framework; Suggett et al., 2023). For example, to overcome logistical challenges, operators opted for a coordinated, collective approach to outplanting, which on one occasion included involvement with other non-CNPW tour operator volunteers in the region. This in-effect, resulted in what has previously been described on the GBR as a "stewardship alliance" where tour operators collaborate to achieve mutually beneficial strategic objectives (Liburd & Becken, 2017). Whilst this approach necessitated higher financial costs (except where time was volunteered), cohesion amongst operators enabled standardised training, and likely resulted in benefits that extend beyond the CNPW operators alone, such as stewardship capacity-building for other reef tourism operators. Such cohesion is contrary to prior suggestions that tourism operators (actors that are fundamentally economic competitors) do not necessarily wish to see others benefit from restoration investments at shared reef sites (i.e., "the commons") (Gibbs & Newlands, 2022). This 'rallying together' of the reef tourism industry was previously documented in the region following tropical Cyclone Debbie (Prideaux et al., 2018) and will likely be critical to the industry resilience in the face of future disturbance. Quantifying such benefits through social science and economic methodologies (e.g,. Hein et al. 2019, 2020, 2021; Palou Zúniga et al., 2023) is thus an obvious and important priority avenue for future research to justify investment (Suggett et al., 2023).

3.6 Conclusions

Early assessments of coral restoration operational models, techniques and cost-effectiveness are essential to inform ongoing implementation and adaptive practice, and build public, stakeholder and management trust (McLeod et al., 2022; Quigley et al., 2022; Suggett et al., 2023). Here, we have described the adaptation of the existing CNP tourism-led assisted reef recovery approach to three inshore fringing-reef sites and different tourism operations in the Whitsundays. For the first time on the GBR, we have reported a detailed account of early-stage implementation and associated costs of community-led coral restoration. We show that activities often unquantified in the delivery of restoration programs (e.g., overheads, planning, in-kind contributions) contribute significant costs, and should thus be included in future cost-tracking efforts for transparent and effective budgeting. Furthermore, we show that accounting for outplant survivorship to assess cost-effectiveness elevates cost-estimates significantly, particularly in inshore environments where survivorship is challenged by poor water quality. We therefore highlight that long-term and locally tailored socio-economic and -ecological monitoring is needed to determine and improve cost-benefits of investment as activity continues.

3.7 Acknowledgements

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3.8 Supplementary Material

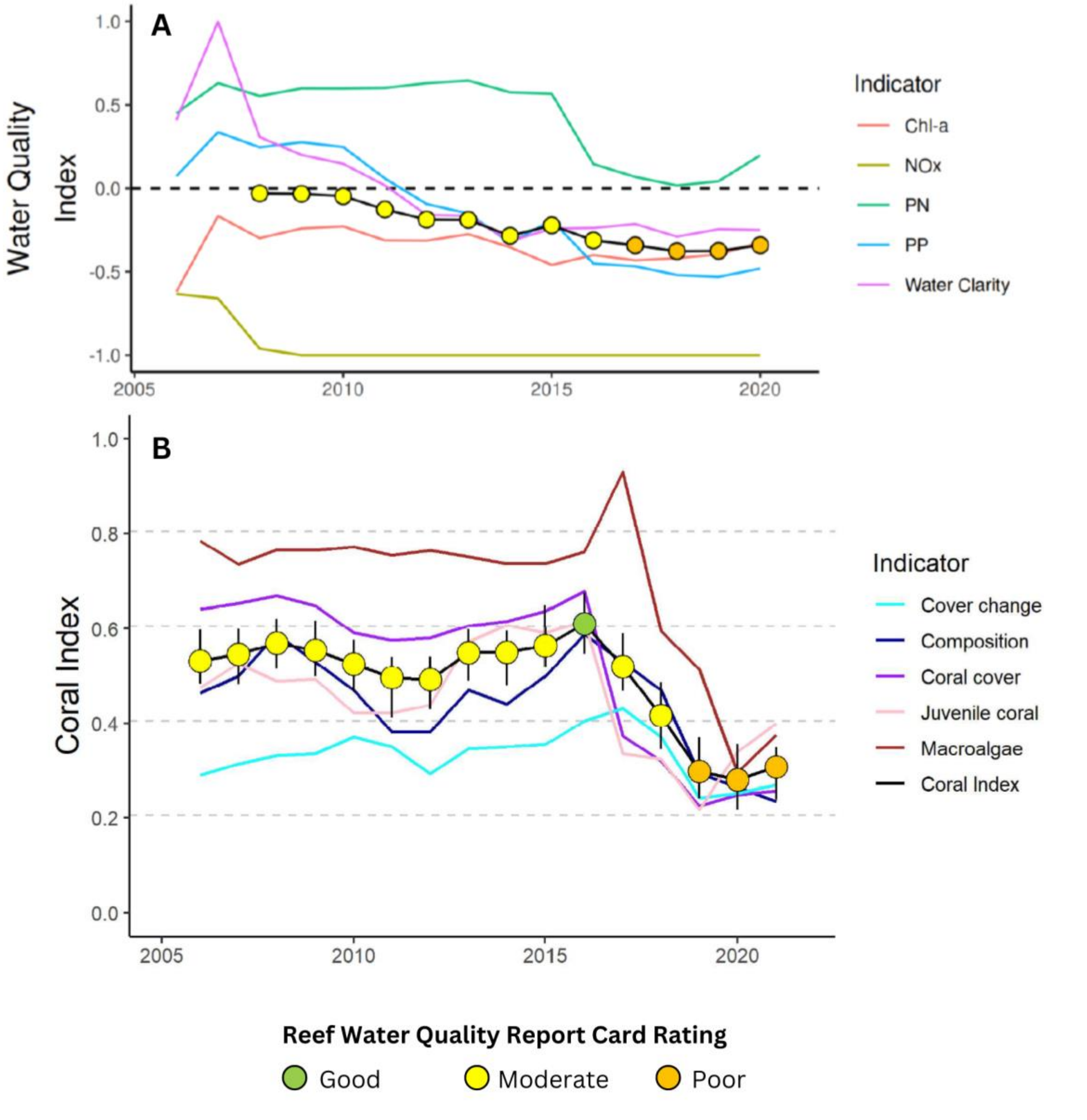

Figure S3.1. Trends in (A) Water Quality Index and *(*B) Coral Index for the Mackay-Whitsundays region demonstrating persistent poor water quality and slow coral community recovery. Figures are adapted from the long-term Great Barrier Reef In-shore Marine Monitoring Program (Waterhouse et al. 2021; Thomson et al. 2022). Index scores are calculated from aggregate coral community condition and water quality sub-indicators shown as coloured lines in each plot: A) The long-term water quality index has shown an overall decline as ‘moderate’ or ‘poor’ since 2008 with underlying parameters of turbidity, total suspended solids, secchi disk depth, chlorophyll a (chl-a), nitrate/nitrite (NOx) and particulate phosphorus (PP) all below guideline values (represented by the black dashed line) All indicators of coral community condition exhibited dramatic declines in 2017 following Cyclone Debbie after a period of relative stability. In 2021, the Coral Index remained ‘poor’ but showed slight improvement due to increases in observed density of juvenile corals indicating slow recovery.

Table S3.1. Results of preliminary scoping in-water video surveys conducted at the proposed CNPW site from July - November 2021 to approximate benthic cover and presence of coral recruits prior to restoration initiation.

	Blue Pearl Bay	**Black Island**	**Luncheon Bay**
Substrate Category	**% cover**		
Hard coral	4	7	5
Soft coral	6	13	11
Coral Rubble	32	27	52
Bare Substrate	58	53	32
Coral recruits	Present	Present	Present

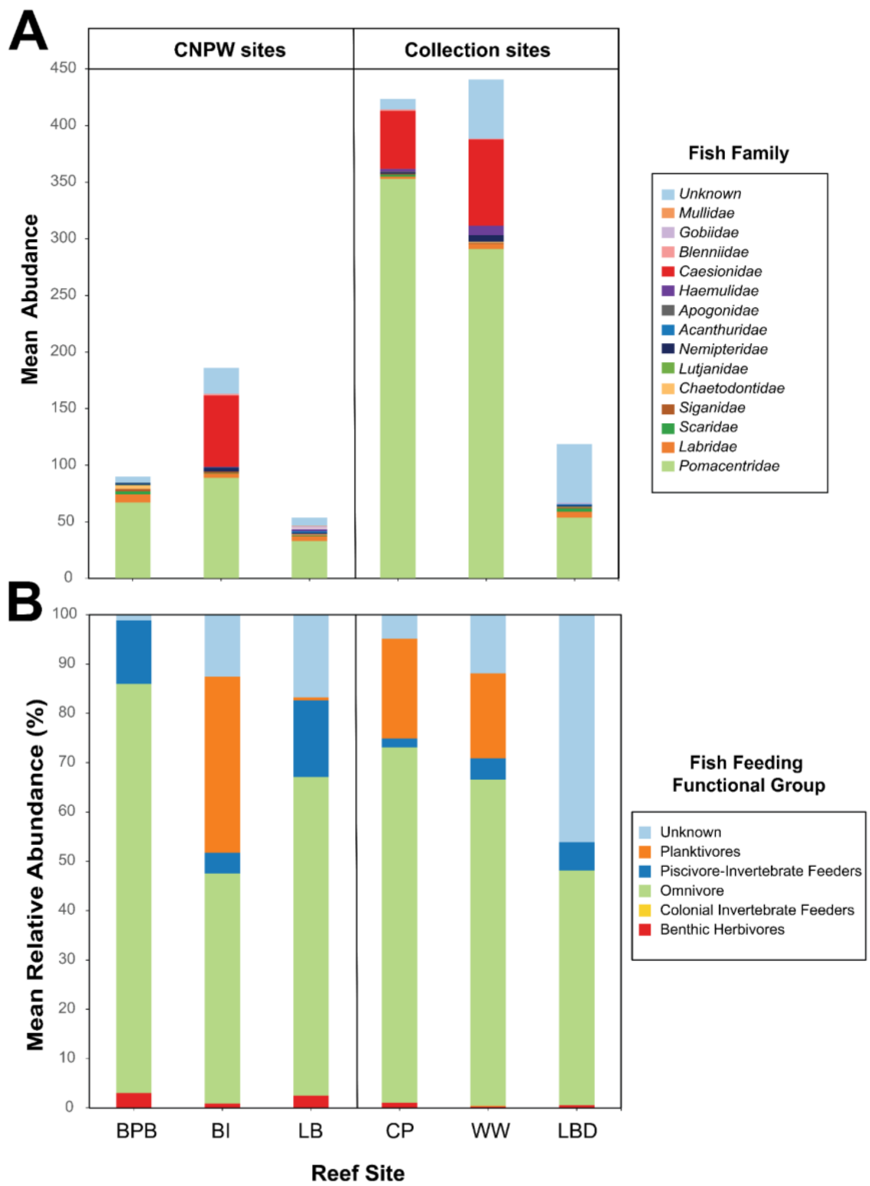

Figure S3.2. A snapshot of fish assemblage characteristics at the three Coral Nurture Program Whitsundays (CNPW) Outplanting (BPB: Blue Pearl Bay, BI: Black Island, LB: Luncheon Bay) and their corresponding collection sites (CP: Cockatoo Point, WW: Wonderwall, LBD: Luncheon Bay Donor). D. **A:** Mean abundance of fish species grouped by Family (n = 3, SE not shown). **B:** Mean relative abundance (%) of fish species grouped by functional feeding group (n = 3, SE not shown). Data was captured at "Site Setup" from triplicate 30m video transects in late August 2022 between 0900h and 1600h. Divers swam out of site for approximately five minutes to allow fish to return prior to slowly filming 1m either side of the 30 m transect line using a GoPro HERO 9®. The total abundance of fish per species was quantified along transects, whereby all individuals within visible range 0.5 m either side of the transect tape were counted. Identified fish species were assigned to a functional feeding group using a classification scheme (as per Mora et al. 2011) derived from dietary information recorded in FishBase (Froese & Pauly, 2011) (see also Villanueva et al. 2012).

Table S3.2. Results of a (a) one-way ANOVA between sites on principal component (PC) 1 and 2 scores derived from benthic composition for CNPW outplanting (BPB: Blue Pearl Bay, BI: Black Island, LB: Luncheon Bay) and collection (CP: Cockatoo Point, WW: Wonderwall, LBD: Luncheon Bay Donor) sites (b) Tukey's HSD pairwise-comparisons between sites. Asterix indicate a significant difference (P-value < 0.05). (*). † Indicates non-significant results between outplanting sites. df = degrees of freedom.

(a) One-Way ANOVA - Site						
		df	**Sum Sq**	**Mean Sq**	**F-value**	**p-value**
PC1	Site	5	10049536	2009907	17.75	3.57×10^{-5}*
	Residuals	12	1359030	113253		
PC2	Site	5	2549910	509982	5.087	0.0098*
	Residuals	12	1203126	100261		

(b) Tukey's HSD post-hoc test - Site					
	Contrast	**Difference**	**Lower Bound**	**Upper Bound**	**p-value**
PC1	BPB-BI	32.00	-890.95	954.95	0.99†
	CP-BI	1617.75	694.80	2540.70	8.07×10^{-4}*
	LB- BI	-54.14	-977.09	868.81	0.99†
	LBD-BI	731.37	-191.58	1654.32	0.16
	WW-BI	1703.22	780.27	2626.17	5.07×10^{-4}*
	CP-BPB	1585.75	662.80	2508.70	9.63×10^{-4}*
	LBD-BPB	699.37	-223.58	1622.32	0.19
	LB-BPB	-86.14	-1009.09	836.81	0.99†
	WW-BPB	1671.22	748.27	2594.17	6.03×10^{-4}*
	LBD-CP	-886.38	-1809.33	36.57	0.062
	LB-CP	-1671.909	-2594.84	-748.94	6.00×10^{-4}*
	WW-CP	85.47	-837.38	1008.42	0.99
	LBD-LB	785.51	-137.44	1708.46	0.11
	WW-LB	1757.36	834.41	2680.31	3.80×10^{-4}*
	WW-LBD	971.85	48.90	1894.80	0.037*
PC2	BPB-BI	695.36	-173.04	1563.76	0.15†
	CP-BI	450.37	-418.03	1318.76	0.53
	LB- BI	-155.29	-1023.69	713.11	0.99†
	LBD-BI	-430.35	-1298.75	438.05	0.58
	WW-BI	-30.98	-899.38	837.42	0.99
	CP-BPB	-244.99	-1113.39	623.41	0.93
	LB-BPB	-850.65	-1719.05	17.75	0.056†
	LBD-BPB	-1125.71	-1994.11	-257.31	0.0094 *

	WW-BPB	-726.34	-1594.74	142.06	0.12
	LB-CP	-605.65	-1474.05	262.74	0.25
	LBD-CP	-880.72	-1749.11	-12.32	0.046*
	WW-CP	-481.35	-1349.75	387.05	0.47
	LBD-LB	-275.06	-1143.46	593.34	0.89
	WW-LB	124.31	-744.09	992.71	0.99
	WW-LBD	399.37	-469.03	1267.77	0.65

Table S3.3. Results of a one-way ANOVA on principal component (PC) 1 and 2 scores derived from benthic composition for Site Type (Outplanting, Collection). Asterix ("*") indicates a significant difference (p-value < 0.05). df = degrees of freedom.

One-Way ANOVA - Site Type						
		df	**Sum Sq**	**Mean Sq**	**F-value**	**p-value**
PC1	Site Type	1	8300703	8300703	42.73	6.84×10^{-6}*
	Residuals	16	3107863	194241		
PC2	Site Type	1	151820	151820	0.675	0.42
	Residuals	16	3601216	225076		

Table S3.4: Loadings of benthic substrate categories in the first (PC1) and second (PC2) principal components of the Principal Components Analysis (PCA) of benthic composition at CNPW outplanting and collection sites.

Benthic category	**PC1**	**PC2**
Macroalgae	-0.50	0.06
Other invertebrates	0.10	-0.27
Consolidated substrate	-0.67	-0.49
Unconsolidated substrate	-0.31	0.78
Hard coral	0.40	0.16
Soft coral	0.53	-0.23

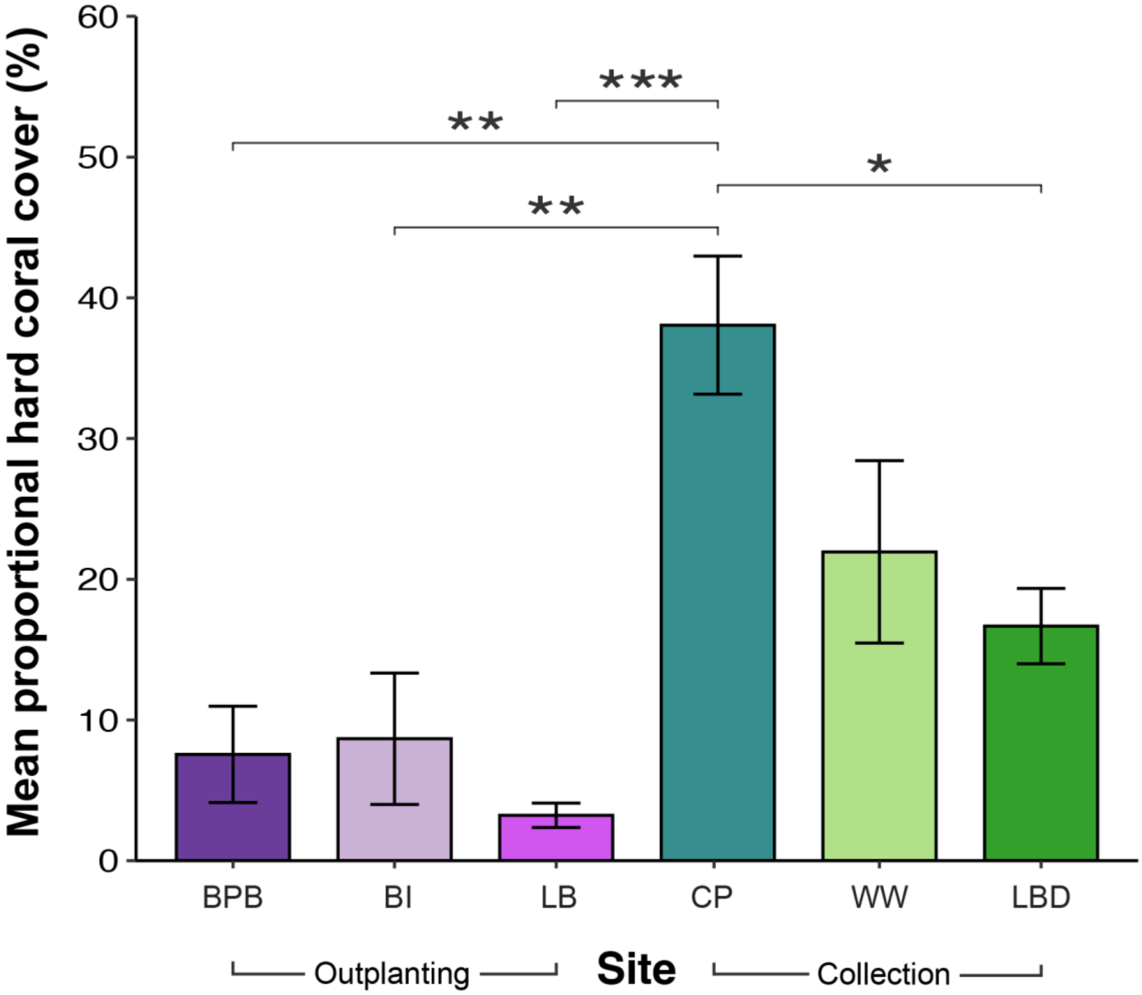

Figure S3.3. Mean (± standard error, n=3) hard coral cover at Coral Nurture Program Whitsundays (CNPW) outplanting (BPB: Blue Pearl Bay, BI: Black Island, LB: Luncheon Bay) and collection sites (CP: Cockatoo Point, WW: Wonderwall, LBD: Luncheon Bay Donor) from triplicate 30m benthic video transects in August 2022. Horizontal bars and asterisks represent significant Tukey's post-hoc comparisons ($p_{adj} < 0.05$) between Sites following a one-way ANOVA test (Tables S3.5) on hard coral cover (in cm). P-values represented by asterisks as follows: * < 0.05, ** <0.01, *** <0.001.

Table S3.5. Site-based differences in % hard coral cover at Coral Nurture Program Whitsundays outplanting (BPB: Blue Pearl Bay, BI: Black Island, LB: Luncheon Bay) and collection (CP: Cockatoo Point, WW: Wonderwall, LBD: Luncheon Bay Donor) sites from triplicate 30m benthic video surveys in August 2022 Shown are the results of (a) one-way ANOVA ($p<0.05$) of mean proportional coral coverage (in cm) between sites and (b) Tukey's HSD post-hoc pairwise comparisons between sites. Results which returned a significant p-value indicated with an asterix (*).† indicates non-significant results between outplanting sites. df = degrees of freedom.

(a) One-Way ANOVA - % Hard coral cover					
	df	**Sum Sq**	**Mean Sq**	**F-value**	**p-value**
Site	5	2188578	437716	9.047	9.26×10^{-4}*
Residuals	12	580567	48381		

(b) Tukey's post-hoc test				
Contrast	**Difference**	**Lower Bound**	**Upper Bound**	**p-value**
BPB-BI	-33.33	-636.57	569.91	1.00†
CP-BI	881.67	278.43	1484.91	0.0038*
LB-BI	-163.33	-766.57	439.9`	0.94†
LBD-BI	240.00	-363.24	843.24	0.76
WW-BI	398.33	-204.91	1001.57	0.30
CP-BPB	915.00	311.76	1518.24	0.0028*
LB-BPB	-130.00	-733.24	473.24	0.98†
LBD-BPB	273.33	-329.91	876.57	0.66
WW-BPB	431.67	-171.57	1034.91	0.29
LB-CP	-1045.00	-1648.24	-441.76	8.96×10^{-4}*
LBD-CP	-641.67	-1244.91	-38.43	0.035*
WW-CP	-483.33	-1086.57	119.91	0.15
LBD-LB	403.33	-199.91	1006.57	0.29
WW-LB	561.67	-41.57	1164.91	0.074
WW-LBD	158.33	-444.91	761.57	0.94

Table S3.6. Coral species stocked on table nursery frames at the three CNPW sites from August 2022 – June 2023. *Note: Acropora spp.' denotes where coral species could not be identified from photographs, but taxa in question comprised branching, corymbose and plating morphologies.

Site	Number of species on nursery frames	Coral species stocked*
Blue Pearl Bay	21	*Acropora cerealis, A. elseyi, A. florida, A. gemmifera, A. humilis, A. intermedia, A. latistella, A. millepora, A. muricata, A. nasuta, A. rosaria, A. sarmentosa, A. spathulata A. spp., Echinopora horrida, Echinopora lamellosa, Montipora aequituberculata, Montipora sp., P. verrucosa, Porites cylindrica, Stylophora pistillata*
Black Island	22	*Acropora abrolhosensis, A. cerealis, A. elseyi, A. humilis, A. intermedia, A. latistella, A. loripes, A. muricata, A. nasuta, A. spathulata, A. verweyi, A. yongei, Acropora spp., Echinopora horrida, Echinopora lamellosa, Leptoseries explanata, Pectinia Paeonia, Pocillopora acuta, Pocillopora. damicornis, Porites cylindrica, Stylophora pistillata, Turbinaria reniformis,*
Luncheon Bay	15	*Acropora cerealis, A. florida, A. gemmifera, A. humilis, A. hyacinthus, A. millepora, A. muricata, A. spathulata, A. tenuis, A. spp., Isopora palifera, Montipora sp., Pocillopora damicornis, P. meandrina, Stylophora pistillata.*

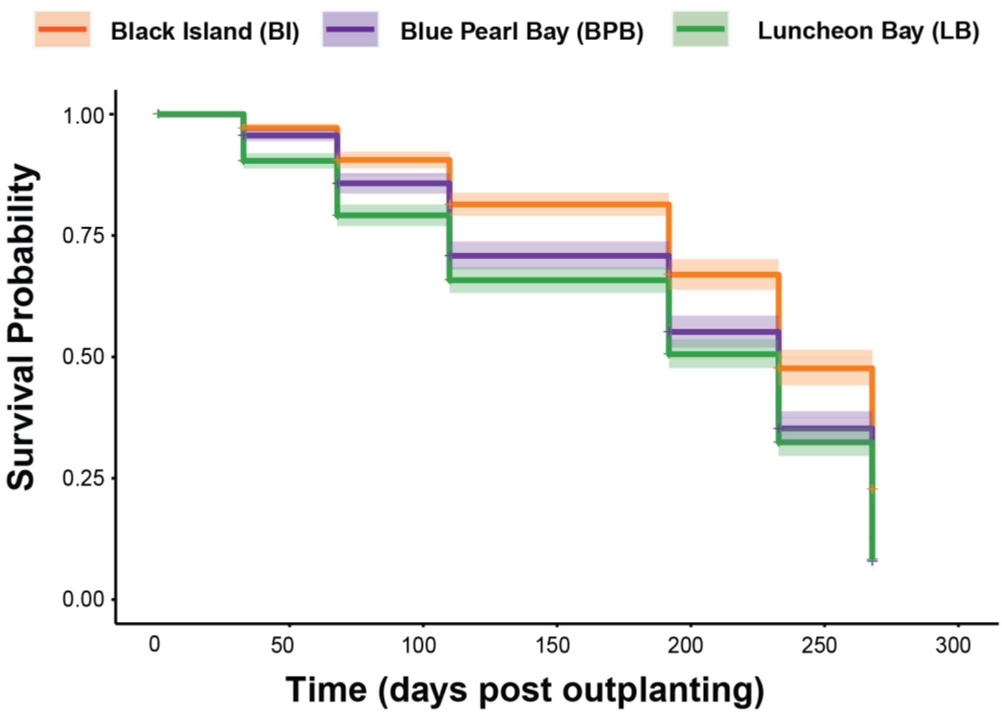

Figure S3.4. Kaplan-Meier survivorship curve (showing 95% CI) of coral outplant survivorship probabilities at the three CNPW sites over nine months with counts of alive outplants observed in triplicate plots at each timepoint as censored observations.

Table S3.7. Kaplan-Meier Survival probabilities at the six survey timepoints across the three CNPW sites. SE = standard error. CI = confidence interval.

Kaplan-Meier Survival Probability						
Time (Days)	**n.Risk**	**n.Event**	**Survival**	**SE**	**lower 95% CI**	**upper 95% CI**
Black Island (BI)						
1	1260	0	1.00	0.00	1.00	1.00
32	1260	36	0.97	0.0047	0.96	0.98
67	1050	71	0.91	0.0087	0.89	0.92
109	840	85	0.81	0.012	0.79	0.84
191	630	112	0.67	0.016	0.64	0.70
232	420	121	0.48	0.019	0.44	0.52
267	210	110	0.23	0.019	0.19	0.27
Blue Pearl Bay (BPB)						
1	1164	0	1.00	0.00	1.00	1.00
32	1164	51	0.96	0.0060	0.95	0.97
67	970	100	0.86	0.011	0.84	0.88
109	776	135	0.71	0.015	0.68	0.74
191	582	129	0.55	0.017	0.52	0.59
232	388	140	0.35	0.017	0.32	0.39
267	194	149	0.08	0.011	0.062	0.11
Luncheon Bay (LB)						
1	1362	0	1.00	0.00	1.00	1.00
32	1362	131	0.90	0.0080	0.89	0.92
67	1135	141	0.79	0.011	0.77	0.81
109	908	153	0.66	0.014	0.63	0.69
191	681	158	0.51	0.015	0.48	0.53
232	454	163	0.32	0.015	0.30	0.53
267	227	169	0.08	0.010	0.065	0.15

Table S3.8. Results of log-Rank test pairwise comparisons (showing p_{adjust} values) performed on Kaplan-Meier survival probability output between sites. A Bonferroni p-value adjustment was applied.

	BI	**BPB**
BPB	$8.7x10^{-13}$	
LB	$< 2x10^{-1664.}$	0.035

Table S3.9. Results of log-Rank test pairwise comparisons (showing p_{adjust} values) performed on Kaplan-Meier survival probability output between site-timepoint groups for Luncheon Bay (LB) and Blue Pearl Bay (BPB) only. A Bonferroni p-value adjustment was applied. ns = non-significant result.

	BPB_T32	**BPB_T67**	**BPB_T109**	**BBP_T191**	**BPB_T232**	**BPB_T267**
LB_T32	1.94×10^{-8}	8.08×10^{-35}	8.08×10^{-35}	8.08×10^{-35}	8.08×10^{-35}	8.08×10^{-35}
LB_T67	3.92×10^{-14}	1.00 (ns)	6.97×10^{-39}	6.97×10^{-39}	6.97×10^{-39}	6.97×10^{-39}
LB_T109	3.92×10^{-14}	7.66×10^{-33}	1.00 (ns)	3.7×10^{-44}	3.7×10^{-44}	3.7×10^{-44}
LB_T191	3.92×10^{-14}	7.66×10^{-33}	4.64×10^{-50}	1.00 (ns)	1.7×10^{-46}	1.7×10^{-46}
LB_T232	3.92×10^{-14}	7.66×10^{-33}	4.64×10^{-50}	8.11×10^{-47}	1.00 (ns)	6.33×10^{-49}
LB_T267	3.92×10^{-14}	7.66×10^{-33}	4.64×10^{-50}	8.11×10^{-47}	7.25×10^{-53}	1.00 (ns)

Table S3.10. Results of a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of original outplants) in triplicate fate-tracked plots between timepoints at Black Island. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom.

Black Island					
One-way repeated measures ANOVA Δ survivorship (of original outplants) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.024	5,10	2.89	1.11×10^{-5}	0.072
Intercept	19.54	1,2	0.00035	2.89	9.00×10^{-6}*

Table S3.11. Results of (a) a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of original outplants) in triplicate fate-tracked plots between timepoints at Blue Pearl Bay, (b) Tukey's HSD post-hoc pairwise comparisons performed on estimated marginal means of 'Δ alive' between timepoints. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom.

Blue Pearl Bay					
(a) One-way repeated measures ANOVA Δ survivorship (of original outplants) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.05	5,10	0.019	5.34	0.012
Intercept	20.26	1,2	0.00012	3.29×10^{-5}	3.04×10^{-6}
(b) Pairwise Comparisons					
Contrast	**Estimate**	**SE**	**df**	**t.ratio**	**p.value**
T32 – T67	0.0085	0.054	2	0.16	1.00
T32-T109	0.039	0.025	2	1.53	0.69

T32-T191	0.14	0.020	2	6.84	0.072
T32-T232	0.09	0.012	2	7.86	0.056
T32 – T267	0.11	0.031	2	3.68	0.22
T67-T109	0.03	0.060	2	0.51	0.99
T67-T191	0.13	0.046	2	2.83	0.33
T67-T232	0.08	0.046	2	1.70	0.60
T67-T267	0.11	0.025	2	4.30	0.17
T109 – T191	0.10	0.014	2	7.25	0.065
T109-T232	0.051	0.034	2	1.51	0.69
T109-T267	0.076	0.044	2	1.71	0.063
T191-T232	-0.049	0.025	2	-2.00	0.53
T191-T267	-0.024	0.031	2	-0.81	0.94
T232-T267	0.024	0.02	2	1.14	0.84

Table S3.12. Results of (a) a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of original outplants) in triplicate fate-tracked plots between timepoints at Luncheon Bay, (b) Tukey's HSD post-hoc pairwise comparisons performed on estimated marginal means of 'Δ alive' between timepoints. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom.

Luncheon Bay					
(a) One-way repeated measures ANOVA Δ survivorship (of original outplants) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.16	5,10	0.0057	57.038	5.01×10^{-7}
Intercept	20.18	1,2	0.00013	312409.88	3.20×10^{-6}
(b) Pairwise Comparisons					
Contrast	**Estimate**	**SE**	**df**	**t.ratio**	**p.value**
T32 – T67	0.25	0.035	2	7.11	0.067
T32-T109	0.25	0.035	2	7.00	0.069
T32-T191	0.26	0.027	2	9.50	0.039*
T32-T232	0.26	0.019	2	13.99	0.018*
T32 – T267	0.26	0.023	2	11.40	0.027*
T67-T109	-0.0036	0.0088	2	-0.41	1.00
T67-T191	0.011	0.011	2	1.00	0.88
T67-T232	0.011	0.016	2	0.68	0.97
T67-T267	0.0087	0.013	2	0.69	0.97
T109 – T191	0.014	0.017	2	0.84	0.93
T109-T232	0.015	0.018	2	0.82	0.94

T109-T267	0.012	0.016	2	0.77	0.95
T191-T232	0.00065	0.0099	2	0.077	1.00
T191-T267	-0.0018	0.0057	2	-0.32	0.99
T232-T267	-0.0024	0.0043	2	-0.57	0.98

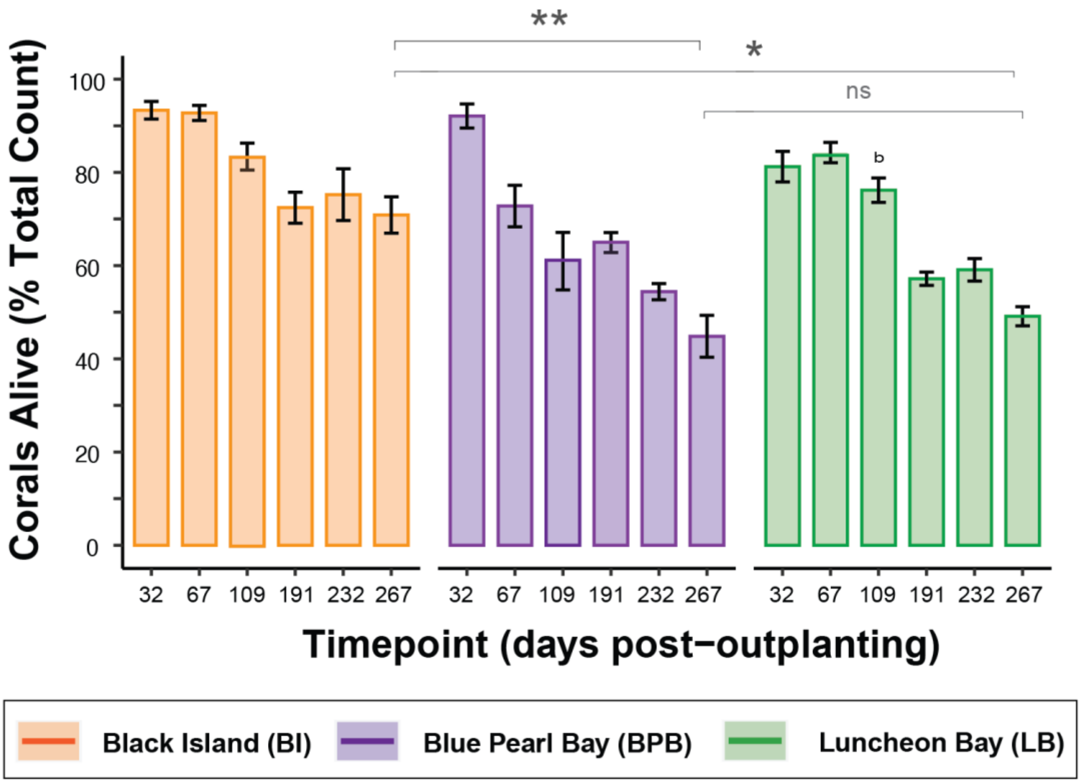

Figure S3.5. Mean (± standard error) proportion of coral outplants 'alive' relative to total count of observations at each survey timepoint in triplicate fate-tracked plots at three CNPW site. After nine months, outplant survivorship at BI (70.87 ± 3.88%) was significantly higher than at BPB (44.82 ± 4.48%) and LB (49.12 ± 2.03%) (Tukey's post-hoc; p = 0.0059, p = 0.014 respectively) (Table S3.13), and survivorship at BI was consistently higher than the other two sites (between 1.22 – 26.05%) across the survey period. Horizontal bars and asterisks represent significant Tukey's post-hoc comparisons (* = p <0.001, ** = p < 0.01, ns = non-significant) between sites at the final survey timepoint (267 days) following a one-way ANOVA test on arc-sin transformed proportions ($F_{2,6}$ = 14.36, p = 0.0052; Table S3.13).

Table S3.13. Results of a (a) one-way ANOVA comparing mean outplant survivorship (as a proportion of total count 'empty, 'dead', 'alive') between CNPW outplanting sites at the final survey time point (T6, 267 days) (b) Tukey's HSD post-hoc pairwise comparisons between sites. Proportion data was arc-sin transformed prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom. BI = Black Island, BPB = Blue Pearl Bay, LB = Luncheon Bay.

(a) One-Way ANOVA - Outplant survivorship at T6 (276 days)					
	df	**Sum Sq**	**Mean Sq**	**F-value**	**p-value**
Site	2	0.13	0.063	14.36	0.0052*
Residuals	6	0.026	0.0044		
(b) Tukey's post-hoc test					
Contrast		**Difference**	**Lower Bound**	**Upper Bound**	**p-value**
BPB-BI		-0.27	-0.44	-0.10	0.0059*
LB-BI		-0.23	-0.39	-0.06	0.014*
LB-BI		0.044	-0.12	0.21	0.71

Table S3.14. Results of a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of total count 'empty, 'dead', 'alive') in triplicate fate-tracked plots between timepoints at Black Island. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom, ns = non-significant.

Black Island					
One-way repeated measures ANOVA Δ survivorship (of total count) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.010	5,10	0.0061	3.26	0.052 (ns)
Intercept	18.86	1,2	0.00037	1.028×10^{-5}	9.73×10^{-6}*

Table S3.15. Results of a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of total count 'empty, 'dead', 'alive') in triplicate fate-tracked plots between timepoints at Blue Pearl Bay. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom.

Blue Pearl Bay					
One-way repeated measures ANOVA Δ survivorship (of total count) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.021	5,10	0.021	1.99	0.17
Intercept	19.62	1,2	0.00047	82830.90	1.21×10^{-5}*

Table S3.16. Results of (a) a one-way repeated measures ANOVA comparing mean proportional change (Δ) in outplant survivorship (of total count 'empty, 'dead', 'alive') in triplicate fate-tracked plots between timepoints at Luncheon Bay, (b) Tukey's HSD post-hoc pairwise comparisons performed on estimated marginal means of 'Δ alive' between timepoints. Proportion data was arcsin-square root transformed (+1 constant), prior to analysis. Results which returned a significant p-value indicated with an asterix (*). df = degrees of freedom.

Luncheon Bay					
(a) One-way repeated measures ANOVA Δ survivorship (of total count) between timepoints					
	Sum Sq	**Df**	**Error**	**F-value**	**p.value**
Timepoint	0.16	5,10	0.0072	6.94	0.0048
Intercept	0.025	1,2	0.00098	40335.32	3.20×10^{-6}*
(b) Pairwise Comparisons					
Contrast	**Estimate**	**SE**	**df**	**t.ratio**	**p.value**
T32 – T67	0.076	0.011	2	7.18	0.066
T32-T109	0.051	0.029	2	1.73	0.069
T32-T191	0.0018	0.012	2	-0.15	1.00
T32-T232	0.10	0.019	2	5.21	0.12
T32 – T267	0.041	0.019	2	2.16	0.48
T67-T109	-0.025	0.021	2	-1.18	0.82
T67-T191	-0.077	0.019	2	-3.98	0.19
T67-T232	0.025	0.027	2	0.93	0.91
T67-T267	-0.034	0.020	2	-1.63	0.63
T109 – T191	-0.053	0.032	2	-1.67	0.64
T109-T232	0.050	0.037	2	1.33	0.77
T109-T267	-0.0095	0.023	2	-0.42	1.00
T191-T232	0.10	0.0077	2	13.41	0.020*
T191-T267	0.043	0.012	2	3.523	0.24
T232-T267	-0.059	0.015	2	-3.90	0.21

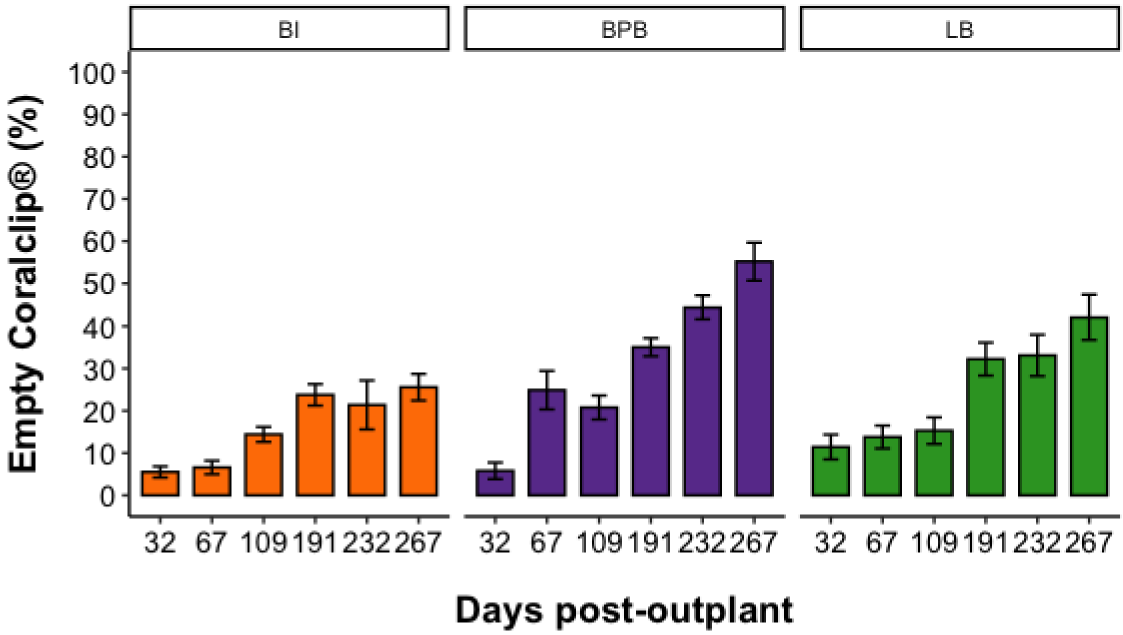

Figure S3.6. Mean (± SE) proportion of empty Coralclip® units relative to total count of corals 'alive', 'dead', and 'missing' (Coralclip® empty) in fate-tracked plots at CNPW sites for each survey timepoint (days post outplant). BI = Black Island (orange), BPB = Blue Pearl Bay (BPB), LB = Luncheon Bay (LB).

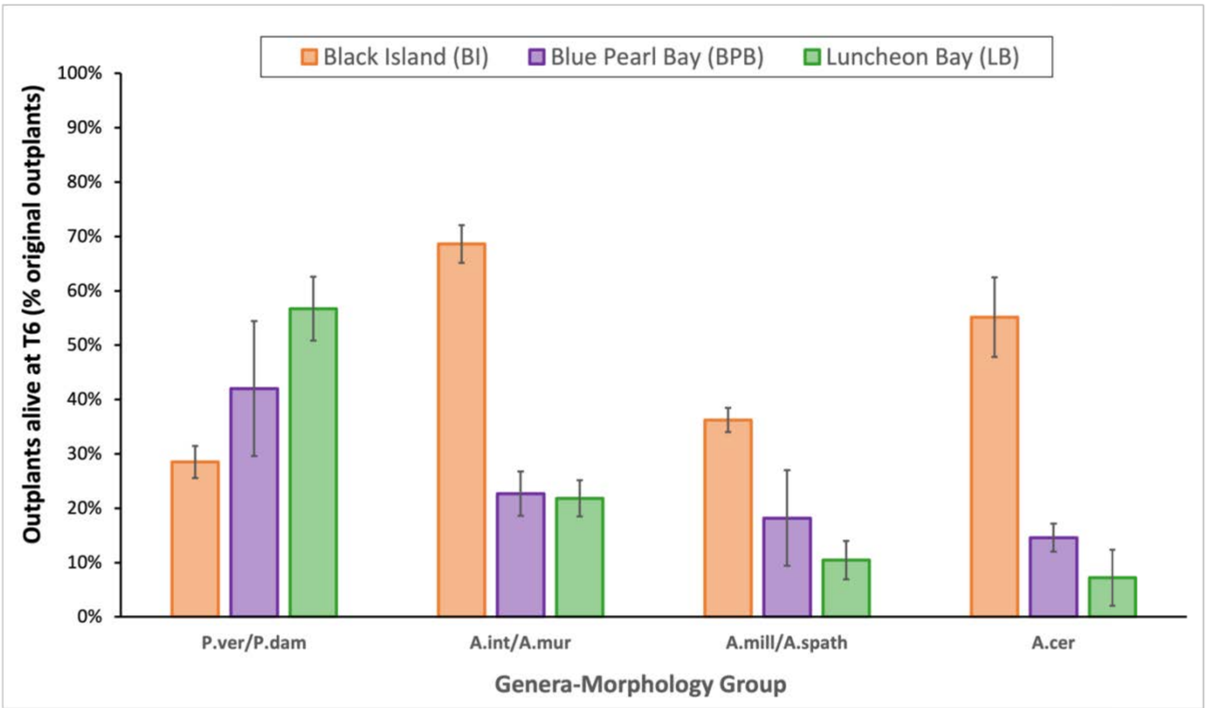

Figure S3.7. Mean (± SE) survivorship (%) across sites of outplants of similar genera-growth morphologies outplanted in triplicate fate-tracked plots at T6 (267 days) post-planting. Survivorship is expressed as the proportion of outplants alive relative to the original number outplanted. P.ver = *Pocillopora verrucose*, P.dam = *Pocillopora damicornis*, A.int = *Acropora intermedia*, A.mur = *Acropora muricata,* A.mill = *Acropora millepora*, A.spath = *Acropora spathulata*, A.cer = *Acropora cerealis.*

Table S3.17: Realised costs (PC_R, \$US coral^{-1}) of CNPW implementation relative to mean outplant survivorship (as a proportion of total count, Table 3.3b) in fate-tracked plots after nine months (T6, 267 days).

Mean Survivorship	PCR (True Cost)	PCR (Actual Cost – less in-kind)
Overall (54.94%)	$104.40	$59.05
BI (70.87%)	$80.93	$45.77
BPB (44.82%)	$127.97	$72.38
LB (49.12%)	$116.77	$66.04

Table S3.18: Illustrative comparison between planting cost (PC, \$US) and realised planting cost (PC_R, \$US) estimates for coral outplanting only (e.g., costs associated with other restoration activities excluded) between Coral Nurture Program (CNP) Whitsundays and CNP Cairns-Port Douglas (**Chapter 2**). Different units used for PC and PC_R for the two locations/studies are owing to methodological differences in value determination (see Supplement S3.1 below).

	CNP Whitsundays (this study)	CNP Cairns-PD (Chapter 2)
PC* (Total)	$57.36	-
Mean PC_R† (Total)	$178.15	-
n (corals outplanted)	4,425 (over 9 months)	30,556 (over 41 months)
PC (outplanting cost only)	US\$10.63 coral^{-1}	US\$2.30 ± 0.19 coral^{-1} trip^{-1}
PC_R (outplanting cost only)	US\$33.04 coral^{-1}	US\$2.94 ± 0.23 coral^{-1} trip^{-1}

*Total costs refer to "True costs" (i.e., including "in-kind" costs) (Table 4)
† Refers to realised planting cost (PC_R) estimates calculated with "True costs" and mean survivorship at T276 across CNPW sites (32.20%; Table 5).

Supplement 3.1 - Comparison note (Table S12):

The term "Illustrative" is used for the above comparison (Table S12), owing to necessary differences in costing methodologies and assumptions employed in the two studies resulting from their different operational phases. In the case of CNP Cairns-Port Douglas, "operational" costs were calculated per "trip" using operational reporting data (CNP activity logs, Scott et al. 2024; **Chapter 2**)*, versus* CNP Whitsundays (this study) where costing was conducted from program outset (i.e., "implementation") using funding reporting data, dive logs and program invoices. Here, outplanting activity was pooled. Methodological differences are outlined in further detail below:

<u>CNP Cairns-Port Douglas</u>

Mean "PC (Outplanting cost only) values were calculated for separate outplanting day "trips" (n = 110, n = 30,556 total outplants, August 2018 – December 2021) across five tourism operations rather than pooling costs and outplants for all outplanting trips as was done in this study. In this instance, outplanting-only activity was integrated within "Routine Planting

Days” on tourism trips, and hence vessel costs were excluded. A per-trip diving and planting gear cost was calculated, and costs were pro-rated over multiple uses assuming gear lifespans of between 2-5 years. Mean “PC_R” was estimated by adjusting PC values by the mean % outplant survivorship observed for the corresponding reef (ranging 68-88%, n = 5 Reefs)) through combined visual and metal detector roving surveys (Scott et al. 2024*;* **Chapter 2**). Here, outplants (n = 4, 723) were not fate-tracked from original deployment but comprised outplants of varying ‘ages’ (up to 3 years) within dispersed outplanting areas. Costs were calculated in $AUD (2018) and converted to $US using the mean daily exchange rate between August 2018 and December 2021 (US$1.00 = AUD$1.28).

CNP Whitsundays

PC (outplanting cost only) was derived by dividing the total outplanting cost across the three deployments described in this study ($US $47,073.90, based on “True costs”, with in-kind volunteer time included (Table 4)) by the total number of outplants (n = 4,425). Here, diving gear costs were either accounted for in vessel charter costs, or costs were derived per trip from gear hire invoices. Planting gear was costed once in its entirety at first use. Realised planting cost (PC_R) estimates were calculated using “True costs” and mean survivorship (as % corals alive of total original outplants) at T276 days (nine months post-outplanting) across CNPW sites (32.20%; Table 5). Costs were calculated in $AUD (2022) and converted to $US using the mean daily exchange rate between 1 January 2022 and mid-June 2023, where US$1.00 = AU$1.45 (MacroTrends, 2023).

Supplement 3.2 – Costing dataset

Screenshots from the Supplementary datasheet used to calculate the costs of CNPW activity from January 2021 to June 2023. The ‘Summary’ is presented first, where summed costs for each activity are detailed by ‘category’. Here, GST (+10%) was added to collective costs (except salaries in overheads), and currency was converted from $AUD to $US using the mean daily exchange rate between 1 January 2022 and mid-June 2023, where US$1.00 = AU$1.45 (MacroTrends, 2023). Subsequent tabs are divided by ‘Activity’: “A) Project planning and administration”, “B) Coral material collection”, “C) Nursery installation”, “D) Nursery stocking and maintenance”, “E) Outplanting”, “F) Monitoring” “G) Research”, “(H) *Ex-situ* training” and “I) Travel and accommodation”. Within each tab costs are calculated (in

AU$2022, excluding GST) for each ‘event’ or ‘trip’ and categorised as ‘labour’, ‘vessel’, ‘consumables and ‘capital items.

SUMMARY - EX GST ($AUD)							
Activity	**Labour**	**Capital**	**Vessel**	**Consumab**	**Overheads**	**(True Cost)**	**Actual Cost (less in-kind)**
Management & Administration	$500.00	$0.00	$3,000.00	$0.00	$125,450.80	$128,950.80	$35,817.80
Coral Material Collection	$5,956.25	$662.33	$21,360.00	$4,410.00	$0.00	$32,388.58	$30,413.58
Nursery Installation	$625.00	$12,110.11	$2,520.00	$9.78	$0.00	$15,264.89	$14,639.89
Maintenance	$2,228.13	$20.00	$7,770.00	$1,660.88	$0.00	$11,679.01	$11,310.26
Outplanting	$15,100.13	$1,800.82	$41,880.00	$3,240.00	$0.00	$62,420.94	$55,092.69
Monitoring	$4,422.00	$88.86	$17,965.00	$0.00	$0.00	$22,475.86	$20,653.86
Research	$1,437.50	$6.76	$5,460.00	$133.25	$0.00	$7,037.52	$6,150.02
Ex-situ Training	$472.00	$0.00	$0.00	$30.00	$0.00	$502.00	$30.00
for CNP researchers/consultant				$19,000.00		$19,000.00	$19,000.00
Total						**$299,719.59**	**$193,108.09**

* US$1.00 = AU$1.45 (conversion factor 0.69)

**Goods and Services Tax (GST) - +10% added to all cost categories except for staff salaries in "Overheads"

PC SUMMARY ($USD*, incl GST**)								
Activity	**Labour**	**Capital**	**Vessel**	**Consumable**	**Overheads**	**Total (True Cost)**	**Actual Cost (less in-kind)**	**% of total (True Cost)**
Project Planning, Management & Administration	$ 379.50	$ -	$ 2,277.00	$ -	$ 121,833.96	$ 124,490.46	$ 24,165.72	49.05%
Coral Material Collection	$ 4,520.79	$ 502.71	$ 16,212.24	$ 3,347.19	$ -	$ 24,582.93	$ 23,083.91	9.69%
Nursery Installation	$ 474.38	$ 9,191.57	$ 1,912.68	$ 7.42	$ -	$ 11,586.05	$ 11,111.68	4.57%
Nursery Stocking & Maintenance	$ 1,691.15	$ 15.18	$ 5,897.43	$ 1,260.61	$ -	$ 8,864.37	$ 8,584.48	3.49%
Outplanting	$ 11,460.99	$ 1,366.82	$ 31,786.92	$ 2,459.16	$ -	$ 47,073.90	$ 41,815.35	18.55%
Monitoring	$ 3,356.30	$ 67.44	$ 13,635.44	$ -	$ -	$ 17,059.18	$ 15,676.28	6.72%
Research	$ 1,091.06	$ 5.13	$ 4,144.14	$ 101.14	$ -	$ 5,341.47	$ 4,667.86	2.10%
Ex-situ Training	$ 358.25	$ -	$ -	$ 22.77	$ -	$ 381.02	$ 22.77	0.15%
Travel and Accomodation for CNP researchers/consultant	$ -	$ -	$ -	$ 14,421.00	$ -	$ 14,421.00	$ 14,421.00	5.68%
						$ 253,800.38	**$ 143,549.05**	
Cost/coral (PC) (n = 4,425)						**$ 57.36**	**$ 32.44**	
% of Total (True Cost)	9.19%	4.39%	29.89%	8.52%	48.00%			

PCR SUMMARY		
Realised Cost (PCR) at T6 (267 days)	**Total (True Cost)**	**Actual Cost (less in-kind)**
Relative to survivivorship as a proportion of original outplants in fate-tracked plots (Table 3a)		
Overall (mean across sites)	$178.12	$100.75
Survivorship at BI	$120.55	$68.18
Survivorship at BPB	$245.85	$139.05
Survivorship at LB	$223.18	$126.23
Relative to survivivorship as a proportion of total count fate-tracked plots (Table 3b)		
Overall (mean across sites)	$104.40	$59.05
Survivorship at BI	$80.93	$45.77
Survivorship at BPB	$127.97	$72.38
Survivorship at LB	$116.77	$66.04

A) Project Planning and Overheads

1. Site Scounting Trip						
Labour Costs						
Staff Type	n (staff days)	Cost/day	% day spent on activity	Total Staff Days	Total	
CNP consultant	2	$ 550.00	1	2	$ 550.00	Includes travel
Principal Investigator 1	1	-	1	1	-	
Local Coordinator	1	-	1	1	-	
Total	-	-	-	**4**	**$ 550.00**	
Total (without in-kind)	-	-	-		**$ 550.00**	
Vessel						
Vessel cost (includes fuel + skipper)	n (days)	Cost/day	% day spent on activity	Total vessel days	Total	
Vessel 1	1	$ 3,000.00	1	1	$ 3,000.00	
Total					**$ 3,000.00**	

2. Project Overheads						
Project Administration - preparation of PIP, permit application, planning, research supervision, in-water activities, writing reports, managing budgets, media and communications						
Labour (salary)	**Duration**	**Cost/year**	**Years**		**Total**	**Total (without in-kind)**
Principal Investigator 1 (0.1 FTE) (in-kind)	12 months	$ 20,833.00	1		$ 20,833.00	$ -
Principle Investigator 1 (0.1FTE) (cost to program)	12 months	$ 20,833.00	1		$ 20,833.00	$ 20,833.00
Principal Investigator 2 (0.1 FTE) (in-kind)	18 months	$ 18,500.00	1.5		$ 27,750.00	$ -
Local Coordinator (0.25 FTE) (partial cost to program)	18 months	$ 34,700.00	1.5		$ 52,050.00	$ 11,000.00
Total					**$ 121,466.00**	**$ 31,833.00**
Other	**Duration**	**Cost/year**	**Years**		**Total**	**Notes**
Queensland fisheries permit for coral collection/propogation	Annual cost	323.2	1.5		**$ 484.80**	
Permit for implementing restoration intervention in Great Barrier Reef Marine Park (GBRMPA)	Annual cost	$ -	1.5		**$ -**	No cost as project is under a research permit and is non-commercial in nature
Total					**$ 484.80**	

Project Planning + Overheads - Summary						
	Labour	**Capital**	**Vessel**	**Consumable**	**Overheads**	**Total**
	$ 500.00	$ -	$ 3,000.00	$ -	$ 121,950.80	**$ 125,450.80**
Without in-kind contributions	$ 500.00	$ -	$ 3,000.00	$ -	$ 32,317.80	**$ 35,817.80**

B) Coral Material Collection

Collection 1 - "Site Setup"						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	5	$ 312.50	0.2	1	-	Staff wages included in vessel charter cost
Research Students	8	$ 125.00	(6 x 0.2), (2 x 0.4)	2	$ 250.00	In-kind
Local Coordinator	3	-	0.2	0.6	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Volunteers	6	$ 312.50	0.2	1.2	$ 375.00	In-kind
Principle Investigator	7	-	0.2	1.4	-	PI staff costs accounted for in A) Overheads
CNP Consultant	2	$ 500.00	0.2	0.4	$ 200.00	
Total				**6.6**	**$ 825.00**	
Total (without in-kind)					**$ 200.00**	
Vessel						
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	6	$ 4,200.00	0.2	1.2	$ 5,040.00	Includes 2x crew wages
Total					**$ 5,040.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	31	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	10	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Total					**$ -**	
Capital	**n**	**Cost/unit**			**Total**	
Chisel	3	$ 10.91	-		$ 32.73	2-5 year lifetime
Hammer	3	$ 8.18	-		$ 24.55	2-5 year lifetime
Wire/bone cutters	3	$ 22.73	-		$ 68.18	2-5 year lifetime
50L storage buckets	4	$ 36.96	-		$ 147.84	2-5 year lifetime
Collapsable plastic crates	2	$ 24.26	-		$ 48.52	2-5 year lifetime
Total					**$ 273.29**	

Collection 2 - February/March - "Nursery Restocking Trips"						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	7	$ 312.50	0.5	3.5	$ 1,093.75	
Research Students	1	$ 125.00	0.5	0.5	$ 62.50	In-kind
Local Coordinator	2	-	0.5	1	$ -	In-kind, Local coordinator staff costs accounted for in A) Overheads
Total				**5**	**$ 1,156.25**	
Total (without in-kind)					**$ 1,093.75**	
Vessel						
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 1	1	$ 3,000.00	0.5	0.5	$ 1,500.00	
Vessel 2	1	$ 3,000.00	0.5	0.5	$ 1,500.00	
Total				**1**	**$ 3,000.00**	
Consumables						
Item	**n**	**Cost/unit**	**% day spent on activity**		**Total**	**Notes**
Dive tanks	10	$ 25.00	-		$ 300.00	
Dive gear	10	$ 300.00	0.5		$ 1,350.00	1 set owned personally
					$ 1,650.00	
Capital	**n**	**Cost/unit**			**Total**	
Chisel	4	$ 10.91	-		$ 43.64	2-5 year lifetime
Hammer	4	$ 8.18	-		$ 32.73	2-5 year lifetime
Wire/bone cutters	4	$ 22.73	-		$ 90.91	2-5 year lifetime
50L storage buckets	6	$ 36.96	-		$ 221.76	2-5 year lifetime
Total					**$ 389.03**	

Collection 3 - March "Training and Monitoring Trip"						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	5	$312.50	0.2	1	$187.50	Costs for 2 staff included in vessel charter cost for Vessel 3
Research Students	9	$125.00	0.2	1.8	$225.00	In-kind
Local Coordinator	3	-	0.2	0.6	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Total				**3.4**	**$412.50**	
Total (without in-kind)					**$187.50**	
Vessel						
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	3	$4,200.00	0.2	0.6	$2,520.00	Includes 2x crew wages
Vessel 1	1	$3,000.00	0.2	0.2	$600.00	
Total				**0.8**	**$3,120.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	23	-	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	23	-	0.2		-	included as part of vessel charter cost for Vessel 3
Total					$ -	
Capital	**n**	**Cost/unit**			**Total**	
Chisel	3	$ 10.91			-	Reuse - cost accounted for in Coral Collection 1
Hammer	3	$ 8.18			-	Reuse - cost accounted for in Coral Collection 1
Wire/bone cutters	3	$ 22.73			-	Reuse - cost accounted for in Coral Collection 1
50L storage buckets	4	$ 36.96			-	Reuse - cost accounted for in Coral Collection 1
Collapsable plastic crates	2	$ 24.26			-	Reuse - cost accounted for in Coral Collection 1
Total					-	

Collection 4 - Coralpalooza						
Labour						
Staff Type	**n (paid staff)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	9	$ 312.50	1	9	$ 2,500.00	Cost for 1 staff included in vessel charter cost for Vessel 3
Research Students	1	$ 125.00	1	1	$ 125.00	In-kind
Volunteer	3	$ 312.50	1	3	$ 937.50	In-kind
Local Coordinator	1	-	1	1	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Total	-	-	-	**14**	**$ 3,562.50**	
Total (without in-kind)	-	-	-		**$ 2,500.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 1	1	$ 3,000.00	1	1	$ 3,000.00	
Vessel 2	1	$ 3,000.00	1	1	$ 3,000.00	
Vessel 3	1	$ 4,200.00	1	1	$ 4,200.00	Includes 2x crew wages
Total			-	**3**	**$ 10,200.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	32	$ 25.00	1		$ 660.00	10 tanks included in vessel charter cost for Vessel 3
Dive gear	13	$ 300.00	1		$ 2,100.00	Cost/unit is for gear hire for 1 person - 6 sets included as part of vessel charter cost for Vessel 3
Total					**$ 2,760.00**	
Capital	**n**	**Cost/unit**			**Total**	
Chisel	6	$ 8.18	-		-	Reuse - cost accounted for in Coral Collection 1,2
Hammer	6	$ 22.73	-		-	Reuse - cost accounted for in Coral Collection 1,2
Wire/bone cutters	3	$ 36.96	-		-	Reuse - cost accounted for in Coral Collection 1,2
50L storage buckets	9	$ 24.26	-		-	Reuse - cost accounted for in Coral Collection 1,2
Total					$ -	

Coral Collection - Summary					
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
	$ 5,956.25	$ 662.33	$ 21,360.00	$ 4,410.00	**$ 32,388.58**
Without in-kind contributions	$ 3,981.25				**$ 30,413.58**

C) Nursery Installation

Nursery Installation 1 - "Site Setup"						
Labour						
Staff Type	**Staff Days**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	-	-	-		-	
Research Students	4	$ 125.00	0.2	0.8	$ 625.00	In-kind
Principal Investigators/Local Cordinator	5	-	0.2	1	-	In-kind, PI and LC labour costs are included in A)"Overheads"
Total	-	-	-	**1.8**	**$ 625.00**	
Total (without in-kind)	-	-	-		**$ -**	
Vessel						
Vessel cost (includes fuel +	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	3	$ 4,200.00	0.2	0.6	$ 2,520.00	
Total			-		**$ 2,520.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	9	-	0.2		-	included as part of vessel charter cost for Vessel 3
Dive gear	9	-	0.2		-	included as part of vessel charter cost for Vessel 3
Stainless steel wire	1 x 15m pack	$ 2.53			$ 2.53	
Mallet	1	$ 7.25			$ 7.25	
Total					**$ 9.78**	
Capital costs						
Item	**n**	**cost/unit**			**Total**	
Stainless Steel Frames	9	$ 1,220.00			$ 10,980.00	
Alumnium Diamond Mesh	9	$ 113.13			$ 1,018.18	
Steel REBAR stakes	36	$ 3.11			$ 111.93	
Total					**$ 12,110.11**	

Nursery Installation - Summary					
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
	$ 625.00	$12,110.11	$ 2,520.00	$ 9.78	**$15,264.89**
Without in-kind contributions	$ -				**$14,639.89**

D) Nursery Stocking & Maintenance

Nursery Stocking 1 - "Site Setup"

Labour

Staff Type	n (Staff Days)	Cost/day	% day spent on activity	Total Staff Days	Total	Notes
Operator Staff	6	$ 312.50	0.2	1.2	$ 375.00	
Research Students	6	$ 125.00	0.2	1.2	$ 150.00	In-kind
Local Coordinator	1	-	0.2	0.2	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Volunteers	2	$ 312.50	0.2	0.4	$ 125.00	In-kind
Principal Investigator(s)	6	-	(1 x 0.4), (1 x 0.2), (1 x 0.4)	1	-	In-kind, PI staff costs accounted for in A)
Total	**-**	**-**	**-**	**4**	**$ 650.00**	
Total (without in-kind)	-	-	-		**$ 375.00**	

Vessel cost (includes fuel + skipper)	n (days)	Cost/day	% day spent on activity	Total Vessel Days	Total	Notes
Vessel 3	3	$ 4,200.00	0.2	0.6	$ 2,520.00	Includes 2x crew wages
Vessel 1	1	$ 3,000.00	0.2	0.2	$ 600.00	
Vessel 2	1	$ 3,000.00	0.2	0.2	$ 600.00	
Total			-	1	**$3,720.00**	

Consumables

Item	n	Cost/unit			Total	Notes
Dive tanks	31	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	10	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Total					**$ -**	

Nursery Stocking 1 - "Site Setup"

Labour

Staff Type	n (Staff Days)	Cost/day	% day spent on activity	Total Staff Days	Total	Notes
Operator Staff	6	$ 312.50	0.2	1.2	$ 375.00	
Research Students	6	$ 125.00	0.2	1.2	$ 150.00	In-kind
Local Coordinator	1	-	0.2	0.2	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Volunteers	2	$ 312.50	0.2	0.4	$ 125.00	In-kind
Principal Investigator(s)	6	-	(1 x 0.4), (1 x 0.2), (1 x 0.4)	1	-	In-kind, PI staff costs accounted for in A)
Total	**-**	**-**	**-**	**4**	**$ 650.00**	
Total (without in-kind)	-	-	-		**$ 375.00**	

Vessel cost (includes fuel + skipper)	n (days)	Cost/day	% day spent on activity	Total Vessel Days	Total	Notes
Vessel 3	3	$ 4,200.00	0.2	0.6	$ 2,520.00	Includes 2x crew wages
Vessel 1	1	$ 3,000.00	0.2	0.2	$ 600.00	
Vessel 2	1	$ 3,000.00	0.2	0.2	$ 600.00	
Total			-	1	**$3,720.00**	

Consumables

Item	n	Cost/unit			Total	Notes
Dive tanks	31	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	10	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Total					**$ -**	

Nursery stocking 3 - "Monitoring and Training Trip" March 2023						
Labour						
Staff Type	**n (Staff Days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	5	$ 312.50	0.25	1.25	$ 390.63	2 x staff costs included in charter cost for Vessel 3
Research Students	1	$ 125.00	0.25	0.25	$ 31.25	In-kind
Local Coordinator	1	-	0.25	0.25	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Total	-	-	-	**1.75**	**$ 421.88**	
Total (without in-kind)	-	-	-		**$ 390.63**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 1	1	$ 4,200.00	0.25	0.25	$ 1,050.00	
Total			**-**	**0.25**	**$ 1,050.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	7	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	7	$ 300.00	0.25		-	included as part of vessel charter cost for Vessel 3
Cable Ties	37	$ 0.08	-		$ 3.03	
Total					**$ 3.03**	
Capital Equipment						
Chisel for removing encrusting organisms	4	$ 10.91	-		-	Reuse, cost accounted for in Collection 1
Scrubbing brushes for cleaning	4	$ 5.00	-		-	Reuse, cost accounted for in Outplanting 1
Total					**$ -**	

	Nursery Stocking and Maintenance				
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
	$ 2,228.13	$ 20.00	$ 7,770.00	$ 1,660.88	**$ 11,679.01**
Without in-kind contributions	$ 1,859.38				**$ 11,310.26**

E) Outplanting

Outplanting 1: "Site Setup" (September 2022)						
Labour						
Staff Type	**n (Staff Days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total Cost**	**Notes**
Operator Staff - Vessel 3	5	$ 312.50	0.2	1	-	2 staff included in vessel charter cost
Operator Staff - Other Operators	12	$ 312.50	(6*1.0) + (6*0.8)	10.8	$ 3,375.00	
Research Students	11	$ 125.00	(1*1) + (3*0.8) + (3*06) + (4*0.2)	22.2	$ 2,775.00	In-kind
Volunteers	6	$ 312.50	(3*1.0) + (1*0.2) + (2*0.8)	4.8	$ 1,500.00	In-kind
Local Coordinator	2	-	(1*0.8) + (1*0.2)	1	-	In-kind, local coordinator staff costs accounted for in A) Overheads
Principal Investigator(s)	3	-	0.8	0.8	-	In-kind, PI labour costs are included in A)"Overheads"
CNP Consultant	2	$ 500.00	1	2	$ 1,000.00	
Total				**42.6**	**$ 8,650.00**	
Total (without in-kind)					**$ 4,375.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 1	2	$ 3,000.00	(0.8 x 1 day) (1.0 x 1 day)	1.8	$ 5,400.00	
Vessel 3	2	$ 3,000.00	1	2	$ 6,000.00	
Vessel 3	7	$ 4,200.00	(2*0.6) + (1*0.7) + (1*0.4) + (1*0.3) + (1*0.2)	2.8	$ 11,760.00	Includes 2x crew wages
Total				**6.60**	**$ 23,160.00**	
Consumables						
Item	**n**	**Cost/unit**	**% day spent on activity**		**Total**	**Notes**
Dive tanks	60	$ 25.00			-	for Vessel 3
Dive gear sets	18	$ 300.00			-	2 personal set, 16 included as part of vessel charter cost for Vessel 3
Total					$ -	
Capital items						
Hammer	6	$ 8.18			$ 49.09	
Chisel	6	$ 10.91			$ 65.45	
Scrubbing Brush	12	$ 5.00			$ 60.00	
Coralclip	1,618	$ 0.28			$ 453.04	
Mesh Basket	3	$ 18.18			$ 54.55	
Total					**$ 682.13**	

Outplanting 2: 'Monitoring and Training Trip (March 2023)						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total Cost**	**Notes**
Operator Staff - vessel 1	5	$ 312.50	0.8	4	$ 1,250.00	2 pax included in vessel charter cost
Operator Staff - vessel 2	3	$ 312.50	(0.8 x 2 days) (0.55 x 1 day)	2.15	$ 671.88	
Research students	3	$ 125.00	(0.8 x 2 days) (0.55 x 1 day)	2.15	$ 268.75	In-kind
Local coordinator	3	-	(0.8 x 2 days) (0.55 x 1 day)	2.15	-	In-kind, Local coordinator staff costs accounted for in A) Overheads
Total				**10.45**	**$ 2,190.63**	
Total (without in-kind)					**$ 1,921.88**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**		**Notes**
Vessel 2	1	$ 3,000.00	0.6	0.6	$ 1,800.00	
Vessel 3	3	$ 4,200.00	(1*0.4) + (2*0.6)	1.6	$ 6,720.00	
Total				**2.2**	**$ 8,520.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	40	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear sets	12	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Total					$ -	
Capital Items						
Hammer	6	$ 8.18	-		-	repeat use from Outplanting 1
Chisel	6	$ 10.91	-		-	repeat use from Outplanting 1
Scrubbing Brush	12	$ 5.00	-		-	repeat use from Outplanting 1
Coralclip	1,088	$ 0.28	-		$ 304.64	
Mesh Basket	6	$ 18.18	-		-	repeat use from Outplanting 1
Total					**$ 304.64**	

Outplanting 3: Coralpalooza						
Labour						
Staff Type	**n (Staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total Cost**	**Notes**
Operator Staff	8	$ 312.50	1	8	$ 1,875.00	2 pax included in vessel charter cost
Research Students	2	$ 125.00	1	2	$ 250.00	In-kind
Post Doc	1	$ 347.00	1	1	$ 347.00	In-kind
Local Coordinator	1	$ -	1	1	$ -	In-kind, Local coordinator staff costs accounted for in A) Overheads
Volunteers	7	$ 312.50	1	7	$ 2,187.50	In-kind
Total				**19**	**$ 4,659.50**	
Total (without in-kind)					**$ 1,875.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**		**Notes**
Vessel 1	1	$ 3,000.00	1	1	$ 3,000.00	
Vessel 2	1	$ 3,000.00	1	1	$ 3,000.00	
Vessel 3	1	$ 4,200.00	1	1	$ 4,200.00	Includes 2x crew wages
Total				**3**	**$ 10,200.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	30	$ 25.00			$ 540.00	12 tanks included in vessel charter cost for Vessel 3
Dive gear sets	18	$ 300.00	1		$ 2,700.00	Cost/unit is for gear hire - note 3 sets owned personally, 6 sets included as part of vessel charter cost for Vessel 3
Total					**$ 3,240.00**	
Capital items						
Item	**Units**	**Cost/unit**			**Total**	
Hammer	12	$ 8.18			$ 98.18	
Chisel	6	$ 10.91			$ 65.45	
Scrubbing Brush	12	$ 5.00			$ 60.00	
Coralclip	1,719	$ 0.28			$ 481.32	
Mesh Basket	6	$ 18.18			$ 109.09	
Total					**$ 814.05**	

Outplanting - Summary					
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
	$ 15,500.13	$ 1,800.82	$ 41,880.00	$ 3,240.00	**$ 62,420.94**
Without in-kind contributions	$ 8,171.88				**$ 55,092.69**

Monitoring 1 - "Site Setup"						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**% day spent on activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	3	$ 312.50	(0.5*1) + (0.4*2)	1.3	-	Staff wages included in vessel charter cost
Research Students	16	$ 125.00	(6 x 0.2), (2 x 0.8), (2 x 0.5), (6 x 0.4)	6.2	$ 775.00	In-kind
CNP Consultant	1	$ 500.00	0.2	0.2	$ 100.00	
Total	-	-	-	**7.7**	**$ 875.00**	
Total (without in-kind)	-	-	-		**$ 100.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	9	$4,200.00	(1*0.1) + (5*0.2) + (3*0.4)	2.3	$9,660.00	Includes 2x crew wages
Total			**-**	**2.3**	**$9,660.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	27	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	7	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Total					$ -	
Capital Items						
Item	**n(units)**	**Cost/unit**			**Total**	
TG Olympus Tough TG6	1	$ 0.27			$ 0.82	daily cost of a $500 camera pro-rated over a lifetime of 5 years
Clipboards and pencils	4	$ 3.00			$ 12.00	
Transect Tape	3	$ 16.68			$ 50.04	
GoPro Hero 9 x 2	9 days	$ 0.22			$ 3.95	daily cost of a $400 camera pro-rated over a lifetime of 5 years
Star pickets	2	$ 5.91			$ 11.82	
Mallet	1	$ 7.25			$ 7.25	Re-use, cost accounted for in Nursery Installation
Total					**$ 85.88**	

F) Monitoring

Monitoring 2 - Monitoring and Training Trip (March 2023)						
Labour						
Cost Type	**n (paid staff)**	**Cost/day**	**day spent on activit**	**Total Staff Days**	**Total**	**Notes**
Research Students	2	$ 125.00	0.8	1.6	$ 200.00	In-kind
Total	-	-	-	**1.6**	**$ 200.00**	
Total (without in-kind)	-	-	-		**$ -**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**day spent on activit**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	2	$4,200.00	(2*0.2)+(1*0.15)	0.55	$2,310.00	
Total				**0.55**	**$2,310.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	13	-			$ -	Tank hire incl. in vessel charter cost
Dive gear sets	3	-			$ -	Dive gear hire incl. in vessel charter cost
Total					$ -	
Capital items						
Item	**n(units)**	**Cost/unit**			**Total**	
TG Olympus Tough TG6	1	$ 0.27			$ 0.82	daily cost of a $500 camera pro-rated over a lifetime of 5 years
Clipboards and pencils	4	$ 3.00			-	
Transect Tape	3	$ 16.68			-	re-use, cost previously accounted for in Monitoring 1
GoPro Hero 9	2	$ 0.22			$ 0.66	daily cost of a $400 camera pro-rated over a lifetime of 5 years
Total					**$ 1.48**	

Monitoring 3 - Intensive Monitoring (n=5 days)						
Labour						
Staff Type	**n (paid staff)**	**Cost/day**	**day spent on activit**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	8	$ 312.50	1	8	$2,500.00	
Research Students	4	$ 125.00	1	4	$ 500.00	In-kind
Post-doc	1	$ 347.00	1	1	$ 347.00	In-kind
Principal Investigator	1	$ -	1	1	$ -	In-kind, PI labour costs are included in A)"Overheads"
Local Coordinator	6	$ -	1	6	$ -	In-kind, LC labour costs incliuded in A) "Overheads"
Total	-	-	-	**20**	**$3,347.00**	
Total (without in-kind)	-	-	-		**$2,500.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**day spent on activity**		**Total**	**Notes**
Vessel 4	5	$1,199.00	1		$5,995.00	
Total					**$5,995.00**	
Consumables						
Item	**n**	**Cost/unit**			**Total**	**Notes**
Dive tanks	52	-	1		-	Tank hire incl. in vessel charter cost
Dive gear sets	18	-	1		-	Dive gear hire incl. in vessel charter cost
Total					-	
Captial items						
Item	**n(Units)**	**Cost/unit**			**Total**	
Clipboards and pencils	4	$ 3.00			-	re-use, cost previously accounted for in Monitoring 1
TG Olympus Tough TG6	1	$ 0.30			$ 1.50	daily cost of a $500 camera pro-rated over a lifetime of 5 years
Total					**$ 1.50**	

Monitoring Summary					
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
Total	$ 4,422.00	$ 88.86	$17,965.00		**$22,475.86**
Total (less in-kind)	$ 2,600.00				**$20,653.86**

G) Research

Research Activities 1 - "Site Setup"						
Labour						
Staff Type	**n (staff days)**	**Cost/day**	**activity**	**Total Staff Days**	**Total**	**Notes**
Operator Staff	2	$ 312.50	0.2	0.4	-	Staff wages included in vessel charter cost
Research Student	16	$ 125.00	(5*0.8) + (3*0.4) + (1*0.5) + (7*0.2)	7.1	$ 887.50	In-kind
CNP Consultant	2	$ 550.00	0.5	1	$ 550.00	
Principle Investigator	1	-	0.2	0.2	-	In-kind
Total	-	-	-	**8.7**	**$ 1,437.50**	
Total (without in-kind)	-	-	-		**$ 550.00**	
Vessel cost (includes fuel + skipper)	**n (days)**	**Cost/day**	**% day spent on activity**	**Total Vessel Days**	**Total**	**Notes**
Vessel 3	6	$ 4,200.00	(1*0.1)	1.3	$ 5,460.00	Includes 2x crew wages
Total			-		**$ 5,460.00**	
Consumables						
Item	**n(Units)**	**Cost/unit**			**Total**	**Notes**
Dive tanks	29	$ 25.00	-		-	included as part of vessel charter cost for Vessel 3
Dive gear	9	$ 300.00	-		-	included as part of vessel charter cost for Vessel 3
Galvanised Steel REBAR (marked plots)	18	$ 2.78	-		$ 50.07	
Epoxy Tubes	6	$ 12.27	-		$ 73.64	
Masonry nails	90	$ 0.11	-		$ 9.55	
Mallet	1	$ 7.25	-		-	Reuse - cost accounted for in Coral Collection 1
Hammer	1	$ 8.18	-		-	Reuse - cost accounted for in Coral Collection 1
Total					**$ 133.25**	
Capital						
Item	**n(Units)**	**Cost/unit**			**Total**	
TG Olympus Tough TG6	1	$ 0.27	-		$ 0.82	daily cost of a $500 camera pro-rated over a lifetime of 5 years
Clipboards and pencils	4	$ 3.00	-		-	Reuse - cost accounted for in Monitoring 1
GoPro Hero 9 x 3	9 days	$ 0.22	-		$ 5.94	daily cost of a $400 camera pro-rated over a lifetime of 5 years
Total					**$ 6.76**	

Research - Summary					
	Labour	**Capital**	**Vessel**	**Consumable**	**Total**
	$ 1,437.50	$ 6.76	$ 5,460.00	$ 133.25	**$ 7,037.52**
Wiithout in-kind	$ 550.00				**$ 6,150.02**

H) Ex-Situ Training

Training - ex-situ						
Labour						
Staff Type	n (staff days)	Cost/day	% day spent on activity	Total Staff Days	Total	Notes
Principal Investigator	1	-	1	1	-	In-kind, PI staff costs accounted for in A) Overheads
Post-doc	1	$ 347.00	1	1	$ 347.00	In-kind
Research Student	1	$ 125.00	1	1	$ 125.00	In-kind
Local Coordinator	2	-	1	2	-	In-kind
Total				**5**	**$ 472.00**	
Consumables						
Item	Unit	Cost/unit			Total	
Venue Hire 1	**1 evening**	$ 30.00			$ 30.00	
Total					**$ 30.00**	

I) Research Travel + Accommodation

Trips	n travellers	Cost/person/trip	Total
Scouting	2	1000	2000
"Site Setup:	7	1000	7000
Taxonomy training	1	1000	1000
Monitoring 1	1	1000	1000
Monitoring 2	1	1000	1000
Monitoring & Nursery restocking training	1	1000	1000
"Monitoring and Training"	3	1000	3000
Coralpalooza/Monitoring	3	1000	3000
Total			**$ 19,000.00**

Chapter 4: General discussion and concluding remarks.

4.1 Summary

Propagating corals to re-plant degraded reefs has become a global practice as reef managers and stakeholders aim to assist natural recovery in the face of current and projected declines in coral reef health (Boström-Einarsson et al., 2020). Biological and ecological feasibility of many such reef restoration efforts from the past two decades have been documented in the scientific literature (Boström-Einarsson et al., 2020; Shaver et al., 2020; Hughes et al., 2023); however few studies have reported project costs alongside outcomes to enable evaluations of cost-effectiveness (Bayraktarov et al., 2019; Suggett et al., 2023). *On Australia's Great Barrier Reef (GBR) in particular, coral restoration and adaptation interventions are emerging management strategies (McLeod et al., 2022) that are fundamentally lacking cost data to support ongoing and future application.* Research presented in this thesis addresses this gap with novel insight into the cost-effectiveness of coral restoration-based site recovery driven by a collaborative tourism industry-research partnership on the GBR: the Coral Nurture Program (CNP) (Howlett et al., 2022). Here, I identified the inherent context-dependent variability of the costs for coral propagation and outplanting, as well as the resulting outplant survivorship at sites across the two major GBR tourism hubs. Specifically, I examined (i) the operational costs of established propagation and outplanting activity via five tourism operations at sites with private moorings on the mid-outer shelf reefs of the Cairns-Port Douglas region (**Chapter 2**; Scott et al., 2024); and (ii) the early implementation and associated costs of CNP activity via three tourism operators at sites with public, shared moorings in the inshore reefs of the Whitsundays (**Chapter 3**) (Fig. 4.1). Within this final chapter, I integrate findings from across these preceding chapters, discuss their utility for guiding improved approaches for cost-reporting in coral restoration practice globally, and their importance to the ongoing application and adaptation of coral restoration activities on the GBR. Furthermore, I outline future research and practice needed to critically advance understanding amongst reef managers, stakeholders and investors of the cost-benefits, and hence socio-economic viability, of coral restoration.

4.2 Integrating costing approaches across operational-environmental contexts.

Previous cost-evaluations of coral restoration on the GBR focussed on coral outplanting costs at a single site (Suggett et al., 2020, 2023), and although the outplanting phase is a significant activity for restoration, it typically represents <50% of total costs in coral propagation and restoration practices (**Chapter 3**; Edwards et al., 2010; Toh et al., 2014; Humanes et al., 2021). In the first cost assessment of multi-site coral outplanting activity in Australia **(Chapter 2)**, I expanded upon this earlier work to examine the nature and variability in cost-effectiveness of coral outplanting across the five diverse tourism operations and reef systems of the Coral Nurture Program (CNP) in the Cairns-Port Douglas region. Importantly, I advanced understanding beyond outplanting cost-efficiency (Suggett et al., 2020) to cost-effectiveness (Fig. 4.2) by evaluating planting costs (PC) weighted by coral outplant survivorship across space and time, thereby yielding the 'realised' costs (PC_R) of retaining coral outplants at diverse reef sites (Fig 4.1). To achieve this, I co-developed and applied a new metal detector-based methodology to recover Coralclip® devices that cannot be visually seen during reef surveys (e.g., are overgrown by established coral outplants) (**Chapter 2**). This approach, utilising a relatively low-cost device (ca. US$200 per metal detector device), can assist in the longer-term fate-tracking and monitoring of restored coral assemblages outplanted using metal materials that are widely employed in coral restoration practice (Edwards et al., 2010; Suggett et al., 2020). Indeed, Coralclip® use now spans > 150,000 units deployed on reefs in >20 countries (J. Edmondson, personal communication, July 25, 2023), largely via stakeholder-led programs, but also in restoration interventions implemented via government agencies and research institutions, such as reef recovery following ship groundings by Queensland Parks and Wildlife.

By examining cost increases associated with wider restoration activities (e.g., training, nursery maintenance), **Chapter 2** highlighted that moving toward a more comprehensive 'whole life' (*sensu* Spurgeon, 2001) restoration costing framework is essential to support project budgeting and investment decisions (Iacona et al., 2018; Bayraktarov et al., 2019; Hein & Staub, 2021). As with the CNP, such frameworks may not be embedded in stakeholder restoration practice from project initiation, and thus **Chapter 2** demonstrates an approach that programs can adopt to opportunistically leverage existing data to derive 'operational' program costs (Fig. 4.1). However, initiation of CNP activity in the Whitsundays (CNPW) during my candidature presented an invaluable opportunity to quantify critical program ‘implementation’ cost attributes and evaluate how novel environmental and tourism operational conditions influence resulting cost-effectiveness estimates (**Chapter 3**) (Fig. 4.1). In **Chapter 3** I reported the first detailed account of early-stage implementation, outcomes, and associated costs of community-led coral restoration on the GBR. The costing approach corroborated and built upon key lessons from **Chapter 2**; specifically, that comprehensive cost-accounting is essential to avoid inflating cost-efficiency estimates, and that ‘effectiveness’ (as assessed as survivorship) is highly site-specific. I also demonstrated that the costs associated with program implementation (e.g., capital and planning costs) are significant, and importantly, that the methodology used to quantify costs (e.g., inclusion or exclusion of in-kind costs) and evaluate effectiveness metrics (e.g., survivorship) substantially moderate resulting cost evaluations. These findings therefore underscore the importance of transparency in cost-reporting methodologies of restoration activities and the need to establish a framework that can be easily adopted for cost reporting within stakeholder-led restoration programs to facilitate greater standardisation, and hence comparability to support resourcing decisions across methods (e.g., Feliciano et al., 2018; Abrina & Bennett, 2021; Mostrales et al., 2022).

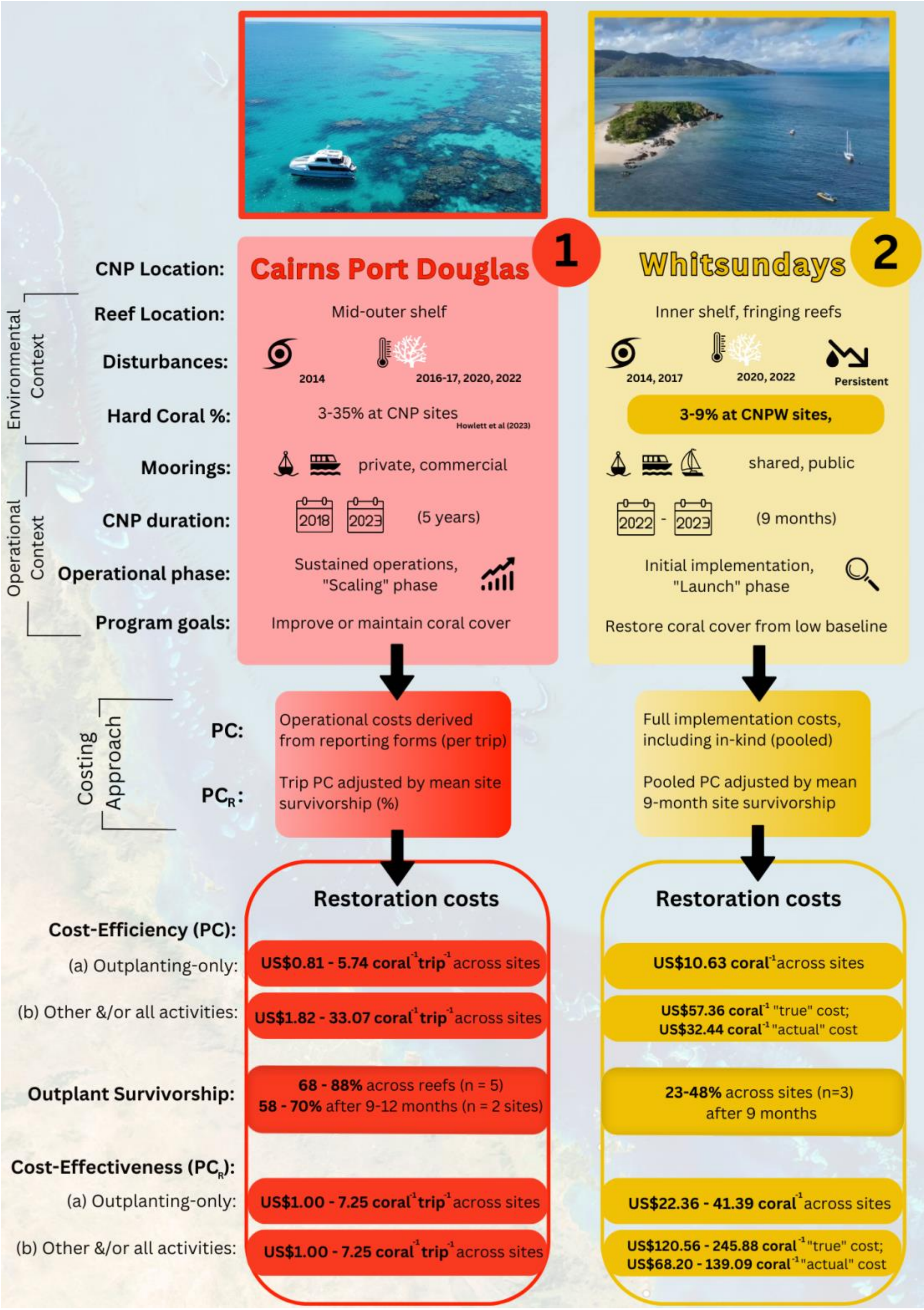

Figure 4.1. The overall findings (highlighted in dark red/yellow), and costing approaches employed to address the two aims of this thesis (adapted from Chapter 1, Fig. 1.1) within the operational-ecological contexts of the Coral Nurture Program Cairns-Port Douglas (Aim 1, Chapter 2) and Whitsundays (Aim 2, Chapter 3). Collectively, these findings demonstrate the within-and-between location variability and inherent context-and-method dependent nature of restoration costs and outcomes. PC = Planting cost, PC_R = Realised planting cost.

Across the two chapters and tourism hubs, several context-specific environmental and operational factors interacted to influence cost-efficiency and -effectiveness (PC, PC_R; Table 4.1) estimates. Full cost comparisons are not currently possible between the two regions (and chapters), given the (i) different operational phases (e.g., ‘sustained operations’ versus ‘launch’) and hence, (ii) different costing approaches employed (e.g., retrospective evaluation of ‘operational’ costs versus cost-accounting from program ‘implementation’) (Fig. 4.1). However, when the respective realised outplanting costs (PC_R) were compared between the two locations (e.g., mean US\$2.94 ± 0.23 coral^{-1} trip^{-1} for CNP-Cairns-Port Douglas vs. US\$22.36 – 41.39 coral^{-1} for CNPW) (**Chapter 3**), it was apparent that despite methodological differences, cost-effectiveness of CNPW operations is inevitably prone to be higher (i.e., greater PC/PC_R) than for CNP (Cairns-Port Douglas) owing to lower outplant survivorship in inshore reef environments and the operational need for ‘dedicated’ vessel use (Table 4.1). In identifying these factors and to some degree, quantifying their cost implications, this thesis therefore enables restoration practitioners and reef managers on the GBR (and in reef regions elsewhere) to understand, and perhaps forecast the cost-implications of specific socio-ecological contexts in restoration planning. For example, higher costs may be reflective of future restoration scenarios, where repeat disturbances challenge coral material availability and survivorship or where selected-for material is sourced from land-based aquaculture operations (Schmidt-Roach et al., 2020; Gibbs, 2021; Banaszak et al., 2023).

Table 4.1. Operational and environmental factors (adapted from Table 2.3, **Chapter 2;** Scott et al., 2024) identified in Chapter 2 (CNP, red) and 3 (CNPW, yellow) that influence and/or regulate planting output (PO), planting cost (PC) and realised costs (PC_R) across Coral Nurture Program (CNP) activity. Cost implications that impact operations to similar degrees across chapters (locations) are indicated by merged, orange cells. Where possible, cost implications for factors (as proportional or \$US costs) are given. CoO = Corals of Opportunity.

Activity	**Factor influencing PC and PC_R**	**Associated cost attributes impacting PC and PC_R**	**Operational/cost implications**	
			Chapter 2 (CNP - Cairns-Port Douglas)	**Chapter 3 (CNPW - Whitsundays)**
Planning and permitting	Time requirements for permit approval Pre-restoration site scoping and surveys and monitoring for permit requirements	• Overheads • Planting output • Vessel requirements • Staff time	Not costed	Staff hired in anticipation of activity are paid even if permitting delays restoration activity. Overheads (largely in-kind) associated with planning, program management and

				administration accounted for ~50% of costs
Site access	Distance to reef site from port	• Vessel size (costs) • Staff number • Fuel requirements • Outplant survivorship • Staff time for outplanting and nursery maintenance	Longer distance to reef sites requires large vessels (and fuel) that if included as "true costs" would be substantial	Smaller vessel sizes mean no extra crew to conduct CNP activity during "routine" tourism operations → necessitates "dedicated" vessel use (30% of total costs)
				Inshore reef environments experience greater nutrient and sediment loads, with increased potential for algae fouling of nurseries (more maintenance time) and outplant mortality.
	Vessel moorings (public or private)	• Planting output • Staff time for monitoring and maintenance	Private moorings mean sites are more regularly accessed (daily in most cases) to enable regular outplanting, nursery maintenance and site monitoring activity.	
	Underlying site condition	• Vessel cost	Degraded reef sites (low live coral cover) are less desirable for tourism visitation - necessitating "dedicated" vessel use. PC on "Dedicated Planting Days" was 92% greater than on "Routine Planting Days" where vessel costs were absorbed within tourism operations	Vessel costs accounted for 30% of total costs
	Tourism operation type (diving/snorkelling)	• Diving gear and SCUBA tank costs	Diving tour operators with access to dive gear and scuba tank compressors can 'absorb' diving gear costs within regular operations, whereas for snorkelling operations gear hire costs are additional.	
	External collaborator travel to restoration sites (e.g., researcher travel)	• Overheads and travel costs	Not costed	6% of total costs
Accessing coral outplant material	Source of coral material for outplanting and nursery stocking Distance between material source and outplant site	• Staff time for coral collection • Vessel time and fuel costs for coral collection	Typically high availability of CoO at sites requires less time for material collection. Operators that rely to a greater extent on nursery-propagated outplants tend to have a lower planting output.	Low availability of CoO at sites necessitates ex-situ coral material collection (10% of total costs).

Nursery Design, Propagation and Maintenance	Nursery Design	• Capital costs for nurseries	Private vessel moorings, greater water clarity, and lower tidal range meant that mid-water floating nurseries made from low-cost materials (ca. US$60 per frame) could be utilised.	Public vessel moorings, reduced water quality and high tidal range meant that fixed bottom table nurseries were required to enable shallower positioning of nurseries without interfering with boat movement (more expensive at ca. $1,000 per frame).
	Nursery cleaning requirements	• Staff time and vessel use requirements for nursery maintenance	PC on "Propagation and Maintenance days was 152% greater compared to "Routine Planting Days".	Nursery stocking and maintenance contributed ~ca 4% of total costs, however greater maintenance requirements are likely required to reduce impacts of elevated algal overgrowth on nurseries.
	Presence of corallivores and herbivores	• Staff time and vessel use requirements for nursery maintenance and stocking	At several sites, nursery cleaning and maintenance is not required due to the presence of herbivorous fish communities who 'clean' frames of biofouling algae.	Fixed-bottom nurseries are more vulnerable to Crown of Thorns predation. Predation by Bumphead Parrotfish required additional nursery stocking trips. Monitoring of herbivory is ongoing at sites, however data is unresolved on fish interactions.
Outplanting	Bare substrate availability and quality Outplanting experience level	• Planting output • Outplant survivorship • Staff time and vessel requirements	Owing to higher coral cover (Howlett et al., 2023) compared to CNPW sites and the longer timeframe of CNP activity, time spent locating available substrate may be higher. Longer timeframe of outplanting practice may result in more secure attachment and better siting of outplants due to adaptive practice over time (although this depends upon staff retention).	High availability of bare substrate but variable substrate quality impacts secure attachment. Planting output is generally slower during initial training.
Training	Staff time allocation to outplanting Researcher travel to facilitate training	• Staff time • Planting output	Staff time dedication to training resulted in PC costs >600% greater than "Routine Planting Day" costs.	Although training requirements were <1% total costs, training was ongoing throughout CNPW trips, and these costs were accounted for in other activities.

Monitoring, site maintenance	Monitoring 'depth'	• Staff time, vessel requirements • Planting output • Survivorship	Not costed	7% of costs.

4.3 Money Matters 2.0: Conducting cost-evaluations is critical but challenging.

This thesis has sought to address the critical knowledge gap, outlined in **Chapter 1**, relating to primary cost data in coral restoration, particularly in Australia (Bayraktarov et al., 2019). Globally, coral restoration cost-estimates are becoming increasingly available through research outputs and the annual reports of restoration companies (Stewart-Sinclair et al., 2021), yet remain limited relative to the number of projects now operational (Boström-Einarsson et al., 2020; Bayraktarov et al., 2019). Whilst the absence of cost data for coral restoration interventions in Australia is arguably from their early-stage maturity on the GBR (McLeod et al., 2022), paucity and inconsistency in cost data has similarly challenged resource allocation decisions across terrestrial conservation and restoration, despite decades of practice compared to marine restoration (Naidoo et al., 2006; Cook et al., 2017; Iacona et al., 2018; Pienkowski et al., 2021; White et al., 2022a). This begs the question: <u>If information on restoration costs is a primary factor underpinning intervention financial feasibility and investment, why is this data so frequently underreported?</u>

Several possible reasons may underpin a lack of cost reporting (see Naidoo et al., 2006; Edwards et al., 2010; Bayraktarov et al., 2015, 2019, 2020; Pienkowski et al., 2021; White et al., 2022a) spanning practical, technical and philosophical considerations (outlined in Table 4.2), and include: (i) a lack of experience or expertise capturing and applying economic data and analyses, (ii) the myriad complexities of cost-accounting **(Chapter 2,3)**, (iii) the additional time-and-financial costs required to collect and interpret cost data **(Chapter 2)**, (iv) lack of capacity to collect and report these data (relates to (i) and (iii)), (v) hesitancy in reporting costs when competing for funding, and (vi) the time-lag between restoration activity and measurable, meaningful outcomes. For the CNP, several of these resource and capacity constraints which can typically challenge stakeholder-led programs, were overcome through affiliation with a university, such as in-kind time, expertise and funding contributions for data collection, interpretation, and synthesis (Table 4.2) However throughout this thesis, I encountered and identified several of the practical complexities of cost-accounting (Table

4.2). Challenges particularly related to disaggregating costs and partitioning labour and resources across discrete restoration activities (e.g., as advised in Edwards et al., 2010) where this data was not recorded in routine CNP reporting forms **(Chapter 2),** or where multiple activities were performed by staff across various organisations **(Chapter 3)** (Table 4.2). However, importantly, through conducting cost-evaluations and encountering these challenges, restoration programs can adjust operating and/or data collection procedures to streamline accounting processes (Table 4.3), as was evidenced by the evolution in costing methodology in **Chapter 3**. Furthermore, because of cost-accounting challenges identified in **Chapter 2,** CNP reporting forms have since been amended for tourism operators to estimate proportional time allocation to different activities on CNP trips to facilitate future cost-evaluations.

Table 4.2. Technical, practical, and philosophical barriers identified across marine and terrestrial conservation practice that challenge consistent cost-reporting, and how they applied to the Coral Nurture Program (CNP) (this thesis).

Costing Challenge	Description	Coral Nurture Program Context
(i) **Lack of expertise in economics or experience using economic tools/data (Technical)**	Restoration programs may largely be led by community stakeholders, restoration practitioners and/or scientists trained in ecology or biology disciplines, and not trained to consider or collect appropriate data for social science or economic valuation tools (e.g., cost-benefit analysis) (Naidoo et al., 2006; Field & Elphick, 2019; White et al., 2022b).	Affiliation with a university enabled the time, financial resources, and expertise to conduct evaluations.
(ii) **Complexities of cost-accounting (Practical)**	Challenges arise when costing discrete activities across complex projects with several collaborators and funding sources, and where costs accrue at various organisational levels or different time horizons (Pienkowski et al., 2021). As such, it is difficult to consolidate a single cost estimate or estimate 'hidden' costs in a way that is comparable across interventions.	Complete life-cycle costs of discrete restoration activities (e.g., as per Edwards et al., 2010) were difficult to quantify given CNP activity is dispersed, additive and ongoing **(Chapter 2)**. Partitioning staff time, capital, consumable costs, and vessel use across outplanting and non-outplanting activities was more time-consuming than simply aggregating costs **(Chapter 2)**, especially where several activities were conducted by different staff paid through different funding sources on a single day **(Chapter 3).**

(iii) **Time and financial costs (Practical)**	Capturing economic data may be perceived as a trade-off from capturing ecological data (Field & Elphick, 2019; White et al., 2022b) or from restoration activity itself **(Chapter 2)**. Collecting costs to the level of detail recommended in cost frameworks can require substantial time and cost investment (Guerry et al., 2015).	Affiliation with a university enabled the time, financial resources, and expertise to conduct evaluations. Detailed reporting can present a time-cost trade-off to primary business activities (tourism) and restoration activity **(Chapter 2).**
(iv) **Lack of capacity (Practical)**	Time-and-financial costs and lack of expertise reduce capacity for practitioners to share their progress in peer-reviewed literature (Bayraktarov et al., 2020; Ferse et al., 2021) Resource constraints on practitioner- or NGO-led programs further disincentivise the documentation of experiences for shared learning (Iacona et al., 2018).	Affiliation with a university enabled the time, financial resources, and expertise to conduct evaluations and the incentive to report/publish findings.
(v) **Sensitivities in reporting costs (Philosophical)**	Reporting costly restoration activities may potentially reduce funding competitiveness (Edwards et al., 2010; Cook et al., 2017; Iacona et al., 2018) or expose projects to critique (e.g., Hughes et al., 2023), potentially impacting social licence.	Reporting costs and evaluating cost-effectiveness became an outcome written into grant proposals (Suggett et al., 2023).
(vi) **Time-lags between action and outcomes (Practical)**	Most grants funding restoration projects are delivered over 1-3 years (Hein & Staub, 2021), and consequently, preclude characterisation of ecological recovery or socioeconomic benefits over the timeframes with which they realistically accrue (Guerry et al., 2015; Boström-Einarsson et al., 2020).	CNP (Cairns-Port Douglas, Chapter 2) has been in operation for 5 years across various phased funding sources (Suggett et al., 2023). Evidence for ecological recovery has been documented at some sites (Roper et al., 2022; Howlett et al., 2023), and interdisciplinary approaches documenting socioeconomic benefits are being explored (e.g., Suggett et al., 2023). CNP Whitsundays is in the 'launch' or pilot phase, with a current funding horizon of 3 years. **Chapter 3** has documented early outcomes, but further socio-ecological evaluation is required as activity progresses.

In **Chapter 1**, I further outlined that cost comparisons between programs and methods are challenged by variability in costing approaches and outputs, which are governed by costs included (or not), how 'effectiveness' is evaluated, and the level of detail provided in contextual metadata (e.g., Table 1.1, **Chapter 1**) (Bayraktarov et al., 2019). In **Chapter 2 and 3**, I demonstrated that high variability in costs and cost-effectiveness is present even within a single program, challenging the derivation of a single cost-estimate and comparability between locations. Calls for standardised reporting frameworks and data-sharing platforms for intervention costs, outcomes and cost-evaluations are growing (e.g., Guerry et al., 2015; Bayraktarov et al., 2019; Ferse et al., 2021; Pienkowski et al., 2021; Eger et al., 2022). Such frameworks are being developed that aim to more explicitly capture important contextual information, assumptions, and metadata to enable use for comparative analyses for effective intervention design (e.g., Edwards et al., 2010; Beher et al., 2016; Cook et al., 2017; Iacona et al., 2018; Goergen et al., 2020; Gouezo et al., 2021; White et al., 2022b; Suggett et al., 2023). Furthermore, systematic cost-reporting is being encouraged in author guidelines for conservation/restoration journals (e.g., *Conservation Biology*) and cost-evaluations or demonstrations of triple-bottom line outcomes (social, environmental, economic) are increasingly required in grant proposals and reporting (Suggett et al., 2023). As such, incentives, and guidelines for restoration practitioners to report and evaluate economic data are increasing, with emphasis that interim or incomplete cost evaluations are better than no data at all (Guerry et al., 2015; Cook et al., 2017). However, such frameworks need to balance detail with ease of application to overcome capacity challenges (e.g., (iii), (iv), Table 4.2) (Iacona et al., 2018; Eger et al., 2022). Furthermore, capacity to meaningfully evaluate and report program cost-effectiveness will continue to be constrained by sufficient financial resources (Bayraktarov et al., 2019; Boström-Einarsson et al., 2020; Hein & Staub, 2021; Suggett et al., 2023) (Fig. 4.2).

Further challenges in comparable cost-evaluations exist where diverse approaches are used across studies/programs to evaluate ecological outcomes and effectiveness (Hein et al., 2017; Bayraktarov et al., 2020). In this thesis, I evaluated cost-effectiveness based on realised costs (PC_R, e.g., cost per surviving coral coral); however, in doing so, any loss of corals originally deployed via mortality or dislodgement immediately becomes an 'expense' and thus elevates PC_R (**Chapter 2, Chapter 3**). Whilst PC_R is a useful metric – particularly to capture early-stage activity where coral outplant losses can be highest (e.g., **Chapter 2, Chapter 3**; Morand et al., 2022) it may ultimately be limited in its entirety to describe 'realisation' of wider restoration successes and goals, e.g., ecological or aesthetic recovery. Characterising cost-effectiveness in relation to long-term CNP goals of assisted site recovery will necessitate further integrated metrics that evaluate costs relative to ecological changes (e.g., coral cover and composition (Roper et al., 2022; Howlett et al., 2023)) or recovery trajectories (e.g., recruitment, juvenile coral density, propagule quality (Roper et al., 2022, *unpublished data*)) at restored sites compared to control areas (i.e.., 'the counterfactual' or 'status quo' alternative scenario) (Ferse et al., 2021; Gouezo et al., 2021; White et al., 2022b; Hughes et al., 2023) (Table 4.3). Mismatches in monitoring metrics, depth and timeframes compared to program goals have been highlighted in several restoration reviews and practitioner surveys (e.g., Bayraktarov et al., 2015, 2019; Hein et al., 2017, 2019; Boström-Einarsson et al., 2020; Ferse et al., 2021). In all cases, the universal limiting factor is the insufficient longevity of restoration project funding (Suggett et al., 2023), which is typically reliant on grants lasting 1-3 years (Hein & Staub, 2021). These constrained timelines are at odds with the decadal-scale, stochastic recovery dynamics of reef ecosystems (see (vi), Table 4.2), and thus limit monitoring capacity to demonstrate the ecological and socioeconomic outcomes needed to justify intervention (Hein et al., 2017).

The mismatch in funding availability *versus* program goals presents a dilemma (or a 'funding-scale trap'; *sensu* Suggett et al. (2023)) for restoration projects: sustained financing to scale activity requires evidence to address investor/management uncertainty on project cost-effectiveness, however providing comprehensive socioeconomic and ecological evidence requires sustained financing (Fig. 4.2). To resolve this dilemma, Hein & Staub (2021) and Suggett et al. (2023) argue that greater communication between practitioners and funders on project objectives and realities is necessary, and that sustained financing structures are planned for from program outset (Table 4.3). Such tailoring of funding to practise will likely need to involve a phased goal-setting approach (Hein & Staub, 2021; Suggett et al., 2023), underpinned by clear specification of time-specific, quantifiable objectives (Carwardine et al., 2008; Beher et al., 2016; Hein et al., 2021), appropriate metrics to indicate progress (Goergen et al., 2020; Gouezo et al., 2021)), the timeframe required to meaningfully conduct monitoring to measure progress to demonstrate cost-benefit (Hein & Staub, 2021, Suggett et al., 2023), and hence realistic program budgets and funding horizons required to implement, measure and report on activity (Fig. 4.2). As projects progress through phases over time, this process should be iteratively repeated and adaptively developed to continually leverage funding and scale activity based upon a strong evidence-base for cost-effectiveness. If user-friendly, standardised reporting frameworks could be integrated into fundraising and reporting activities (Pienkowski et al., 2021) (Table 4.3), and sufficient resources made available to incentivise practitioners to do so, primary cost data will increasingly become available to integrate into management decisions and guide adaptive practice for projects elsewhere (Fig. 4.2)

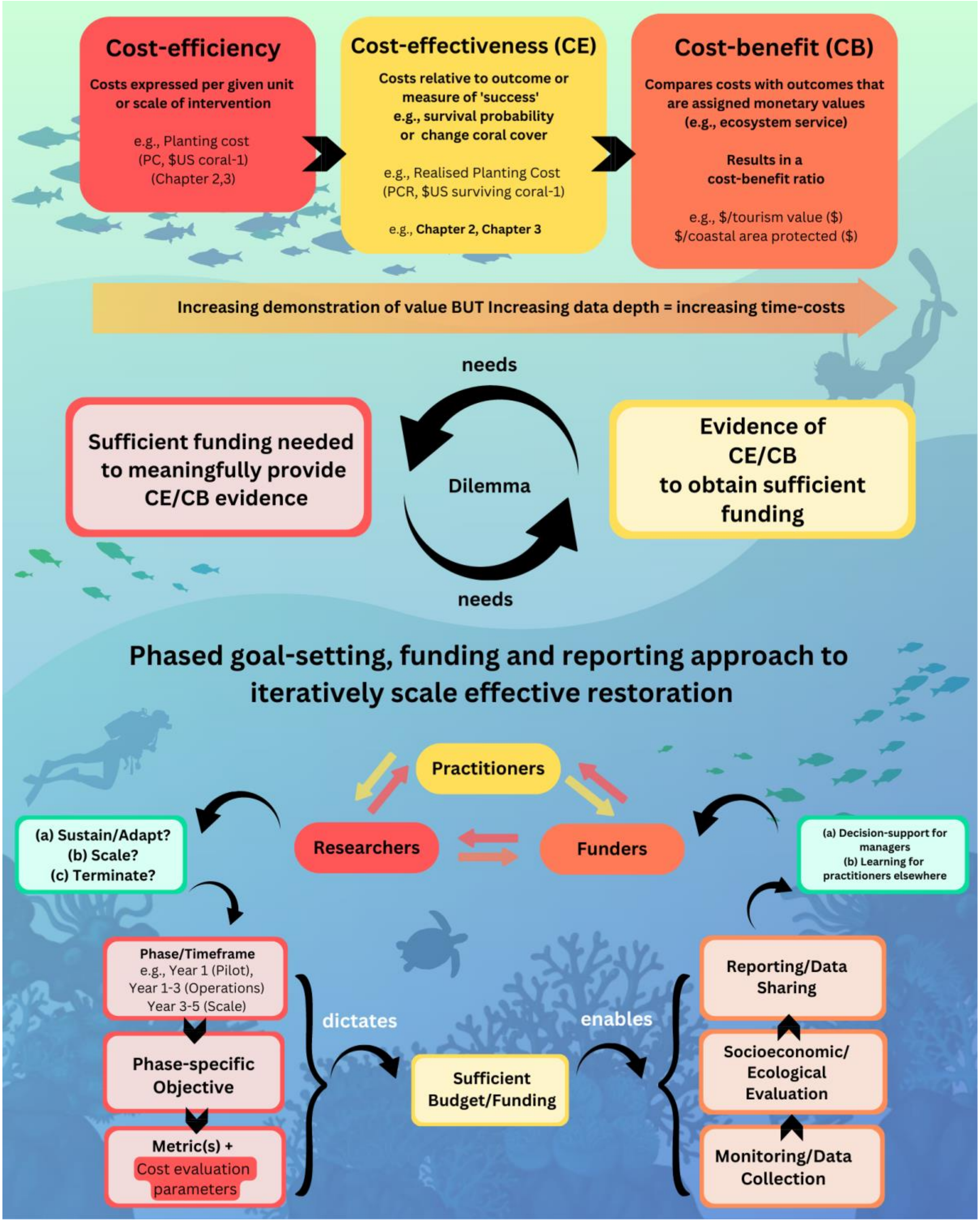

Figure 4.2. The cost-evaluation continuum outlined in Chapter 1 (Fig. 1.2) depicting progressive demonstration of value from cost-efficiency to cost-effectiveness (CE) ("*" this thesis) to cost-benefit (CB), that simultaneously necessitates increasing data depth and time-costs. The need to invest time and resources into cost-evaluations to justify and obtain further investment presents a dilemma for restoration practitioners, as restoration budgets focussed on activity or delivering long-term goals are often insufficient to deliver effective goal-based cost evaluations within realistic timeframes. An iterative, phased goal setting, funding, and reporting approach co-developed by practitioners and funders (and researchers, if involved) can ensure goal-oriented cost-

evaluations are budgeted for in restoration funding delivery to enable informative cost-reporting for investors, managers, and the restoration community.

4.4 Future cost-evaluations should consider and quantify restoration benefits.

Cost-effectiveness evaluations are somewhat limited as demonstrations of value for investors, policymakers, and the general public, without explicitly linking restoration outcomes to their benefits for society and economies (Iftekhar et al., 2017; Eger et al., 2022; Suggett et al., 2023). Coral reef restoration generates multiple socioeconomic benefits that to date, have been documented (e.g., Fadli et al., 2012; Kittinger et al., 2016; Hein et al., 2019; Bayraktarov et al., 2020), but rarely quantified in relation to costs. The CNP similarly delivers several socioeconomic benefits, such as capacity-building for tourism operators and volunteers in restoration skillsets and ecological knowledge (**Chapter 2 and 3**), collaboration and cohesion amongst tourism operators towards mutual objectives **(Chapters 3),** livelihood retention during tourism downturns (Howlett et al., 2022; Suggett et al., 2023), and education on reef threats and restoration approaches (contributing to social licence) for thousands of tourists who travel to the reef with CNP operator partners (Howlett et al., 2022). Restoration is increasingly being framed as a 'socio-ecological endeavour' (sensu Fischer et al., 2021; see also Fernández-Manjarrés et al., 2018; Uribe-Castañeda et al., 2018; Palou Zúniga et al., 2023; Suggett et al., 2023) that generates value (and occasionally costs) to coastal communities and economies. At the same time, ecosystem service, natural capital values and disclosure of biodiversity risks of trade/business activity (e.g., the Taskforce for Nature-related Financial Disclosures, (TNFD, 2023)) are increasingly being centralised in public and private decision-making (Guerry et al., 2015). Thus, implementing interdisciplinary research approaches that can quantify cost-benefits are an obvious priority for restoration practice moving forward (Woodhead et al., 2019) (Fig. 4.2; Table 4.3).

Table 4.3: Future critical reef restoration research and practice priorities to advance understanding of restoration costs and cost-benefits.

Future area for research and/or restoration practice	**Key steps and methods**	**Potential collaborative partners/beneficiaries**
Evaluate current monitoring processes to ensure data required for cost evaluations are being captured alongside ecological data.	Determine current uncertainties and knowledge-gaps around costs and evolve monitoring/data capture processes to resolve uncertainty.	Restoration Practitioners

Determine cost-effectiveness based upon broader ecological metrics over longer time horizons (depending on goals).	Survey restoration areas using photomosaics (e.g., Neufeld & Fundakowski, 2020) or aerial methods (for shallow reefs, e.g., Peterson et al., 2023) to quantify changes in coral cover or size-frequency distributions of key coral taxa.	Restoration practitioners Research community Funding agencies Reef managers
Budget for and include time-costs associated with measuring and reporting restoration outcomes and communicate realistic time horizons with funders.	Quantify staff-time and costs (including hidden 'in kind' costs) associated with *in-situ* monitoring, data processing and reporting.	Restoration practitioners Funding agencies
Develop practitioner-focussed guidance on cost-reporting that is widely disseminated through reef restoration networks.	Develop guidelines similar to those for reef restoration design (e.g., Shaver et al., 2020, 2022) and monitoring (Goergen et al., 2020). Disseminate findings through workshops and seminars.	Reef restoration consortiums (e.g., International Coral Reef Initiative, Nature Conservancy, Coral Restoration Consortium).
Map the local-scale, socio-ecological system for GBR tourism hubs: Cairns-Port Douglas and Whitsundays regions	Conduct interdisciplinary research to understand the key stakeholders, beneficiaries and collaborative agencies involved in reef restoration in the Cairns Port Douglas and Whitsundays regions. Identify and map key ecosystem services (ES) and beneficiaries. Utilise local-scale valuation and benefit-transfer approaches to quantify current ES values.	Researchers (spanning economics, social science, and ecology). Government/management agencies Funding agencies Reef stakeholders including Traditional Owners, recreational and commercial reef users.

One such approach is through cost-benefit analyses (Fig. 4.2), which utilise economic tools such as choice-modelling and contingent valuation (Costanza et al., 2017) to parameterise market and non-market ecosystem service (ES) values (e.g., supporting, regulating, provisioning and cultural services such as habitat, coastal protection, fisheries, cultural identify and practices) (Millennium Ecosystem Assessment, 2005). On the GBR over 30 studies have sought to measure and document various ES values (Rolfe & De Valck, 2021). Recent work is attempting to overcome the paucity of studies on the reef's non-market aesthetic and spiritual values (Stoeckl et al., 2011) by integrating the whole range of benefits flowing to coastal users (including non-market and First Nations Peoples values) into more traditional valuation approaches (e.g., (Stoeckl et al., 2021; De Valck et al., 2023). However, as ES values and beneficiaries (as well as restoration intervention costs (Naidoo et al., 2006; **Chapter 2, 3**)) are spatially heterogeneous and highly context-specific (Table 4.1), challenges arise when using benefit transfer approaches to transpose empirical restoration cost and ES benefit estimates derived for one location to another (see Costanza et al., 2017; Rolfe & De Valck, 2021; Stoeckl et al., 2021). Therefore, for localised coral restoration strategies where costs and values are governed by specific and dynamic ecological, social and economic contexts (e.g., as documented in this thesis), locally tailored, stakeholder-informed valuation approaches (Costanza et al., 2017; Abrina & Bennett, 2021) are perhaps better suited to quantify potential values generated by restoration interventions (Gouezo et al., 2021; Suggett et al., 2023).

Integrated cost-benefit decision frameworks based upon socio-ecological systems (SES; also socio-ecological networks, SEN) approaches (e.g., Uribe-Casteñada et al., 2018; Gouezo et al., 2021; Suggett et al., 2023) may present practical tools for conceptualising, measuring and hence valuing the flow of ecosystem service values and benefits that potentially arise from (and justify) reef restoration interventions (Gouezo et al., 2021; Palou Zúniga et al., 2023; Suggett et al., 2023). An SES explicitly represents the complex and dynamic nature of the interactions, dependencies, and feedbacks between the social and ecological components of a given context (Felipe-Lucia et al., 2022). Suggett et al (2023) proposed an integrated framework for reef restoration interventions that links the underlying SES, restoration effectiveness and the financing landscape to (i) identify the diverse stakeholder networks that derive, influence and prioritise ES values (ii) identify and quantify baseline ES values that frame restoration goals (i.e., asset values) (iii) quantify the extent to which restoration interventions retain or improve ES provision (i.e., value proposition) relative to associated costs (i.e., cost-benefit) (iv) identify the funding requirements and horizons, and hence financing sources that can enable restoration goals (Fig,. 4.2). For the CNP, using an estimate of tourism ES value for Opal Reef (Spalding et al., 2016, 2017), Suggett et al. (2023) suggested that for every US$1 spent on reef restoration, US$10 in tourism value is retained. Resolving complete SES networks and valuing the full suite of associated ES values will not be without challenges, such as those outlined for reporting restoration costs in Table 4.2. However, I would argue that for high-value restoration locations, such as the Cairns-Port

Douglas and Whitsundays GBR tourism hubs that were the focus of this thesis, the asset value of reef sites (e.g., Spalding et al., 2017; De Valck & Rolfe, 2018), the dependence of local communities on this asset value (Marshall et al., 2017; Prideaux et al., 2018), and hence their contributions to local and national economies (O'Mahoney et al., 2017) warrants further interdisciplinary research focus and investment (Table 4.3) (Suggett et al., 2023). Furthermore, as reefs continue to degrade under increasing disturbance frequency (Kleypas et al., 2021), ES provision and local valuation is increasingly likely to underpin reef intervention prioritisation frameworks into the future (Gouezo et al., 2021). Resolving the socio-ecological cost-benefit of restoration efforts is thus a critical next step to inform the 'complex decision challenge' inherent to coral reef management in the Anthropocene (Anthony et al., 2020).

4.5 Concluding remarks.

This thesis represents the first multi-site assessment of restoration costs and cost-effectiveness on Australia's Great Barrier Reef. Such cost data are critical to establishing the knowledge base upon which reef managers and investors need to make increasingly time-critical intervention decisions (Sivapalan & Bowen, 2020), yet remains underreported in restoration practice in Australia, and globally. Here, I have provided valuable insights into the context-dependency of restoration costs and outcomes and contributed primary cost, outplant survivorship and benthic data to inform ongoing adaptive practice, future integration and tailoring of restoration activities within stakeholder activities on the GBR. Importantly, through examining coral propagation activity adopted by eight diverse tourism operations across the GBR's two major tourism gateways, this thesis has reinforced the eagerness and capacity of the tourism industry to adopt more proactive site stewardship practices, regardless of potential intrinsic motivations underpinned by property rights (i.e., public, or private vessel moorings) (Hein et al., 2020; Gibbs & Newlands, 2022). Overall, the two chapters in this thesis underscore the importance of collaboration and 'buy-in' across multiple partners, spanning stakeholder, reef management, funders, and researchers, to enable restoration (Bartelet et al., 2023; Palou Zúniga et al., 2023; Suggett et al., 2023). Several such collaborative restoration models are increasingly being implemented on the GBR to upskill reef tourism operators and Traditional Owners as part of the Reef Restoration and Adaptation Program (RRAP) and the Great Barrier Marine Park Authority's tourism strategy and "Reef Blueprint" ((GBRMPA, 2017, 2021). As interest and urgency to invest into and implement reef interventions grows, resolving a cohesive framework for transparent cost evaluations that can be adopted across restoration programs is becoming increasingly time-critical to demonstrate socio-ecological benefits, and thus unlock emerging funding mechanisms (Eger et al., 2022; Suggett et al., 2023). Collectively, work presented in this thesis has demonstrated different formats with which costings can be achieved, highlighting the highly variable nature of cost-effectiveness inherent to reef restoration.

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Chapter 1

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