T4: CURRENT PRACTICES

A report provided to the Australian Government by the Reef Restoration and Adaptation Program

Boström-Einarsson L[1], Ceccarelli D[1], Babcock RC[2], Bayraktarov E[3], Cook N[4], Harrison P[5], Hein M[1], Shaver E[6], Smith A[4], Stewart-Sinclair PJ[3], Vardi T[7], McLeod IM[1]

[1]TropWATER, James Cook University
[2]CSIRO
[3]University of Queensland
[4]Reef Ecologic
[5]Southern Cross University
[6]The Nature Conservancy
[7]NOAA

September 2019

Inquiries should be addressed to:

Dr Ian McLeod
James Cook University
Ph: +61 7 4781 5569
Ian.mcleod@jcu.edu.au

This report should be cited as

Boström-Einarsson L, Ceccarelli D, Babcock RC, Bayraktarov E, Cook N, Harrison P, Hein M, Shaver E, Smith A, Stewart-Sinclair PJ, Vardi T, McLeod I.M. (2019) Reef Restoration and Adaptation Program: Current Practices. A report provided to the Australian Government by the Reef Restoration and Adaptation Program (55 pp).

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Acknowledgement

This work was undertaken for the Reef Restoration and Adaptation Program, a collaboration of leading experts, to create a suite of innovative measures to help preserve and restore the Great Barrier Reef. Funded by the Australian Government, partners include: the Australian Institute of Marine Science, CSIRO, the Great Barrier Reef Marine Park Authority, the Great Barrier Reef Foundation, The University of Queensland, Queensland University of Technology and James Cook University, augmented by expertise from associated universities (University of Sydney, Southern Cross University, Melbourne University, Griffith University, University of Western Australia), engineering firms (Aurecon, WorleyParsons, Subcon) and international organisations (Mote Marine, NOAA, SECORE, The Nature Conservancy).

In addition, this work was achieved in collaboration with, and cofounded by, the National Environment Science Program Tropical Water Quality Hub.

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1. PREAMBLE

The Great Barrier Reef

Visible from outer space, the Great Barrier Reef is the world's largest living structure and one of the seven natural wonders of the world, with more than 600 coral species and 1600 types of fish. The Reef is of deep cultural value and an important part of Australia's national identity. It underpins industries such as tourism and fishing, contributing more than $6B a year to the economy and supporting an estimated 64,000 jobs.

Why does the Reef need help?

Despite being one of the best-managed coral reef ecosystems in the world, there is broad scientific consensus that the long-term survival of the Great Barrier Reef is under threat from climate change. This includes increasing sea temperatures leading to coral bleaching, ocean acidification and increasingly frequent and severe weather events. In addition to strong global action to reduce carbon emissions and continued management of local pressures, bold action is needed. Important decisions need to be made about priorities and acceptable risk. Resulting actions must be understood and co-designed by Traditional Owners, Reef stakeholders and the broader community.

What is the Reef Restoration and Adaptation Program?

The Reef Restoration and Adaptation Program (RRAP) is a collaboration of Australia's leading experts aiming to create a suite of innovative and targeted measures to help preserve and restore the Great Barrier Reef. These interventions must have strong potential for positive impact, be socially and culturally acceptable, ecologically sound, ethical and financially responsible. They would be implemented if, when and where it is decided action is needed and only after rigorous assessment and testing.

RRAP is the largest, most comprehensive program of its type in the world; a collaboration of leading experts in reef ecology, water and land management, engineering, innovation and social sciences, drawing on the full breadth of Australian expertise and that from around the world. It aims to strike a balance between minimising risk and maximising opportunity to save Reef species and values.

RRAP is working with Traditional Owners and groups with a stake in the Reef as well as the general public to discuss why these actions are needed and to better understand how these groups see the risks and benefits of proposed interventions. This will help inform planning and prioritisation to ensure the proposed actions meet community expectations.

Coral bleaching is a global issue. The resulting reef restoration technology could be shared for use in other coral reefs worldwide, helping to build Australia's international reputation for innovation.

The $6M RRAP Concept Feasibility Study identified and prioritised research and development to begin from 2019. The Australian Government allocated a further $100M for reef restoration and adaptation science as part of the $443.3M Reef Trust Partnership, through the Great Barrier Reef Foundation, announced in the 2018 Budget. This funding, over five years, will build on the work of the concept feasibility study. RRAP is being progressed by a partnership that includes the Australian Institute of Marine Science, CSIRO, the Great Barrier Reef Foundation, James Cook University, The University of Queensland, Queensland University of Technology, the Great Barrier Reef Marine Park Authority as well as researchers and experts from other organisations.

2. INTRODUCTION, BACKGROUND AND OBJECTIVES

2.1 Threats to coral reefs

The primary driver of global species loss is habitat decline (Tilman et al., 1994, Pimm et al., 1995, Segan et al., 2016), as critical habitats disappear and take with them the resources necessary for species to persist (Bender et al., 1998, Gibbons et al., 2000, Stuart et al., 2004). Reductions in species richness, population declines, and extinctions alter the structure of animal, plant and microbial communities, fundamentally changing ecosystem functioning (Larsen et al., 2005, Dobson et al., 2006). While the loss of terrestrial habitats has been recognised as an important environmental management issue for centuries, marine ecosystem decline has only become apparent in recent decades. In this timeframe, coral reef ecosystems have suffered an unprecedented loss of habitat-forming hard corals (Gardner et al., 2003; Bruno and Selig 2007; Wilkinson 2008; Burke et al., 2011). For example, coral cover on Caribbean reefs declined from an average of 36 percent in 1970 to 16 percent in 2012 (Jackson et al., 2014), and average coral cover on Australia's Great Barrier Reef dropped from 28 percent in 1985 to 14 percent in 2012 (De'ath et al., 2012). In particular, reefs close to urban areas are subject to a suite of chronic and acute anthropogenic disturbances (Alongi 2002), including increased nutrient outputs from agriculture (Fabricius and De'ath 2004), elevated levels of suspended sediment caused by deforestation and development (Richmond 1993), destructive fishing practices (Burke et al., 2011) and over-harvesting of reef species (Jackson et al., 2001). In addition, outbreaks of corallivorous crown-of-thorns starfish, (*Acanthaster planci*, Moran et al., 1992; Pratchett et al., 2017), coral disease (Harvell et al., 2002) and tropical storms (De'ath et al., 2012) are known to be major drivers of coral reef decline in many regions. However, since the first recorded global mass-bleaching (a stress response where corals lose their symbiotic algal partners and energy providers) event in 1998, climate change has emerged as the primary threat to coral reefs (Hoegh-Guldberg 1999, Pandolfi et al., 2003, Bellwood et al., 2004, Bruno and Selig 2007). This was further emphasised during the recent global marine heat wave, which led to the most extensive coral bleaching event in history, including remote and pristine reefs (Hughes et al., 2018a).

2.2 Natural recovery

Dynamic systems like coral reefs have an innate capacity for natural recovery. Reefs have historically made complete recoveries from major natural disturbances such as cyclones (Hughes and Connell 1999; Lukoschek et al., 2013) and crown-of-thorns starfish outbreaks (Pratchett et al., 2014). However, mass coral bleaching is increasing in frequency, intensity and severity (Hughes et al., 2018b), and extreme weather events are predicted to become more powerful and damaging (Cheal et al., 2017), eroding the time and capacity for recovery between catastrophic events. Combining these shocks with chronic pressures such as pollution, sedimentation and overfishing may drive phase shifts, where previously coral-dominated reefs are overgrown with other organisms (e.g. algae or sponges), leading to alternate stable states where corals are rare or absent (Done 1992; Anthony 2016). Furthermore, the widespread nature of impacts to coral reefs can impact connectivity and system properties of reef networks, affecting processes that are critical for coral recovery; larval supply, settlement and recruitment of coral larvae (Harrison and Wallace 1990; Richmond 1993, Hock et al., 2017) and post settlement survival are often compromised by chronic or repeated disturbance events (Baker et al., 2008, Fabricius et al., 2017, Graham et al., 2015), making natural recovery unlikely, or impossible, in many locations (dela Cruz and Harrison 2017). In cases where natural recovery of the coral community does occur (Gilmour et al., 2013; Adjeroud et al., 2018), the recovery-state may be fundamentally different from the pre-disturbance community composition (e.g. Brown 1997; Berumen and Pratchett 2006; Hughes et al., 2018b). Given the current global ecological and political situation and future predictions of increasing stressors, combating habitat loss on multiple levels is likely to be the fundamental issue for ecologists and managers in the Anthropocene. This has led to an increasing impetus and interest in interventions that may boost the resilience of reefs, or aid in the preservation and restoration of coral reef structure and function (van Oppen et al., 2017; Anthony et al., 2017).

2.3 Habitat protection or intervention?

Until very recently, marine conservation has favoured passive habitat protection over restoration. However, recent research has shown that optimal conservation outcomes may include both habitat protection and restoration (Possingham et al., 2015). Restoration is common practice in terrestrial ecosystems and is well accepted for coastal habitats such as wetlands (Wolanski and Elliott 2015) or shellfish reefs (Brambaugh et al., 2006; Gillies et al., 2018), but has been controversial for coral reefs, both in academia and amongst marine managers. Critics of interventions often argue that coral restoration detracts focus from mitigating climate change and other threats to the marine environment (e.g. Bruno and Valdivia 2016, Hughes et al., 2017). Proponents of coral restoration counter (1) that interventions can serve to protect coral biodiversity in the short-term, while mitigation of large-scale threats such as climate change and water quality take effect (e.g. Anthony et al., 2017), (2) are necessary for the recovery of endangered coral species such as the elkhorn coral *Acropora palmata* and the staghorn coral *Acropora cervicornis* in the Caribbean (Young et al., 2012; Chamberland et al., 2017; Lirman and Schopmeyer 2016), and (3) increase environmental stewardship and interest in protecting coral reefs by including local communities in restoration projects (e.g. Marshall et al., 2012, Hesley et al., 2017).

Local-scale restoration action could bridge the temporal gap between large-scale action and the substantial lag effects predicted for indirect management actions. This is particularly important for climate change, where the temperature is predicted to increase for several more decades even in a zero-carbon emission scenario (Meehl et al., 2005, IPCC 2018). Given that disturbed reefs are likely to suffer from Allee effects and a reduction in genetic diversity following large-scale disturbance events (Knowlton 2001, Hoegh-Guldberg et al., 2007, Mora et al., 2016), preserving coral species and genetic diversity through active restoration could 'buy time' for recovery following the removal of stressors. Furthermore, local management actions can boost the resilience of corals to more substantial threats, including climate change. For example, in a recent study, Shaver et al., (2018) experimentally demonstrated increased resistance to bleaching, and increased recovery, in corals where corallivorous snails had been removed.

Despite widespread reservations, active coral restoration has been increasingly used as a tool to restore coral populations in recent decades (reviewed by Rinkevich 2005; Edwards and Gomez 2007; Edwards 2010; Omori 2010; Young et al., 2012). However, owing to the disconnect between coral restoration practitioners, coral reef managers and scientists, most coral restoration work to date has been undertaken with little or no scientific input or detailed monitoring. Therefore, a substantial proportion of coral restoration projects and methods have not been documented in the scientific literature. A paucity of documentation, coordination and sharing of knowledge reduces our ability to learn from past successes and failures and increases the risk of repeatedly testing similar methods and hypotheses. To counteract this, we aimed to synthesise the available knowledge in a comprehensive global review of coral restoration methods. This review builds upon the work of previous authors who have reviewed aspects of coral restoration (e.g. Rinkevich 2005, 2014; Yeemin et al., 2006; Zimmer 2006; Ammar 2009; Chou et al., 2009; Edwards and Gomez 2007; Edwards 2010; Omori 2010; Johnson et al., 2011; Young et al., 2012; Bayraktarov et al., 2016; Lirman and Schopmeyer 2016; Barton et al., 2017; Hancock et al., 2017; Hein et al., 2017). While previous reviews have been helpful, they have often been limited in their geographic scope (e.g. primarily focused on a specific region), methods assessed (e.g. focus on fragmentation only), species included (e.g. focused on *Acropora* only), data sources (based on unpublished case studies), or have not critically evaluated the efficacy of specific methods.

To provide a more comprehensive review of the collective knowledge of coral restoration methods, we aim to augment the data collected from a traditional scientific literature search with information from sources outside traditional academia. Specifically, we targeted restoration practitioners with an online survey and access online sources for specific details about restoration methods and new developments. We aim to provide a synthesis of the current methods of coral restoration, highlight common problems and potential areas of concern and identify knowledge gaps.

3. METHODS

While most reviews have focused entirely on either reviewing published literature or assembling case studies, a large proportion of restoration projects do not result in peer reviewed publications. For example, many environmental non-governmental organisations (eNGOs) conduct large scale projects without any formal documenting of outcomes. Therefore, just reviewing peer reviewed literature may introduce a bias towards projects with a scientific output as the key objective compared to real-world restoration objectives. They also have the potential to suffer from positive result bias, with the consequence that descriptions of unsuccessful projects cannot be used to inform the development of future restoration practices.

To mitigate this, we assembled case studies and descriptions of coral restoration methods from four sources: 1) the primary literature (i.e. published peer-reviewed scientific literature), 2) grey literature (e.g. scientific reports and technical summaries from experts in the field), 3) online descriptions (e.g. blogs and online videos describing projects), and 4) an online survey targeting restoration practitioners (www.coralrestorationsurvey.com). We included only those case studies which actively conducted coral restoration (i.e. at least one stage of scleractinian coral life-history was involved). This excludes indirect coral restoration projects, such as disturbance mitigation (e.g. predator removal, disease control etc.) and passive restoration interventions (e.g. enforcement of control against dynamite fishing or water quality improvement). To the best of our abilities, we avoided duplication of case studies across the four separate sources, so that each case in our review and database represents a separate project.

Information gathered from each case study was entered into a database (except for online sources where detailed information was unavailable or unreliable), where data were organised into six categories (1) the information source, (2) the case study particulars (e.g. location, duration, spatial scale, objectives, etc.), (3) specific details about the methods, (4) coral details (e.g. genus, species, morphology), (5) monitoring details, and (6) the outcomes and conclusions. These six categories were further split into details so that the final database includes more than

40 columns of data for each case study (access the full database here). This database will be provided online, in the form of a spreadsheet and interactive figure, as a resource for further exploration of the methods, techniques and concepts we review here. The interactive figure can be viewed here, and will be linked throughout this document.

While our expanded search enabled us to avoid the bias towards published literature, we acknowledge that using sources that have not undergone rigorous peer-review potentially introduces another bias. Many government and NGO reports undergo an informal peer review, however survey results and online descriptions may present a subjective account of restoration outcomes. However, the academic peer-review system can also be flawed, so that a published journal article does not necessarily reflect a more accurate representation of data compared to a survey result. While this issue is not possible to resolve in a review, we have avoided introduction of a potential further bias by not interpreting results or survey answers, instead only reporting exactly what authors have stated.

Primary literature

We used multiple search engines to achieve the most complete coverage of the scientific literature. First, we searched the scientific literature using Google Scholar with the keywords "coral* + restoration". Because the field (and therefore search results) is dominated by transplantation studies, we then conducted separate searches for other common techniques using "coral* + restoration + [technique name]". We then complemented this search by using the same keywords in ISI Web of Knowledge. We then manually selected studies that fulfilled our criteria for active coral restoration described above. In those cases where a single paper describes several different projects or methods, these were split into separate case studies. Finally, we consulted prior reviews of coral restoration to obtain case studies from their reference lists (e.g. Rinkevich 2005, 2014; Yeemin et al., 2006; Zimmer 2006; Ammar 2009; Chou et al., 2009; Edwards and Gomez 2007; Edwards 2010; Omori 2010; Johnson et al., 2011; Young et al., 2012; Bayraktarov et al., 2016; Lirman and Schopmeyer 2016; Barton et al., 2017; Hancock et al., 2017; Hein et al., 2017).

Grey literature

While many reports appeared in the Google Scholar literature searches, we also consulted The Nature Conservancy (TNC) database of reports for North American coastal restoration projects (http://projects.tnc.org/coastal/). This was supplemented with reports listed in the reference lists of other papers, reports and reviews, and during our online searches.

Online records

Small-scale projects conducted without substantial input from researchers, academics, non-governmental organisations (NGO) or coral reef managers often do not result in formal written accounts of methods. To access this information, we conducted online searches of YouTube, Facebook and Google, using the search terms "Coral restoration". We used information provided in videos, blog posts and websites to describe a further ~50 projects. Due to the unverified nature of such accounts, we have limited the data collected compared to peer reviewed literature and surveys. At the minimum, the location, the methods used and reported outcomes or lessons learned were included in this review.

Online survey

In order to access information from projects not published elsewhere, we designed an online survey targeting restoration practitioners. The survey consisted of 25 questions (Appendix 1) querying restoration practitioners regarding projects they had undertaken. These data were entered into the database in two separate versions (1) a publicly available version which anonymises the participants, and (2) a locked version used for calculations in this review. Although we encouraged participants to fill out a separate survey for each case study, it is possible that participants included multiple separate projects in a single survey, which may reduce the real number of case studies reported.

Cost of restoration

Funding for coral reef restoration is often hindered by uncertainties of risks around the costs and feasibility of restoration (Edwards & Gomez, 2007). The estimates of costs of restoration projects were provided by Dr Elisa Bayraktarov and Phoebe Stewart-Sinclair from the University of Queensland and were sourced from their database of costs and feasibility of marine coastal restoration (Bayraktarov et al., 2016), and unpublished data (2014 - 2018) provided by the University of Queensland on 27th of June 2018 as described below. This database is separate from the one generated and used for this report overall.

We reviewed primary literature, reports, and published data repositories to build a database of coral reef restoration projects of the last 40 years until. We conducted a systematic literature search using Web of Science (Core collection; Thomson Reuters, New York, New York, USA) and Scopus (Elsevier, Atlanta, Georgia, USA) databases. Databases were searched for peer-reviewed articles using the search terms "(coral reef* OR coral*) AND restor*", as well as "(coral reef* OR coral*) AND rehab*" to collate all literature on restoration and rehabilitation available until 16th of March 2018. The search was confined by focusing on the key terms in the title and resulted in a total of 141 studies. In a second step, an EndNote (Version X7.0.2; Thomson Reuters, New York, New York, USA) search was performed within the full text (any field + PDF with notes) of the sources using the search terms (cost* OR feasib* OR surviv*) to only account for studies either indicating cost or feasibility/survival. This second step resulted in narrowing down our total studies to 128. The overall literature search and review led to the creation of a coral reef database containing data from 86 studies, out of which 71 were published within the database from 2016 (Bayraktarov et al., 2016) and contained published literature until November 2014. Data from plots and figures were extracted graphically by using WebPlotDigitizer (available online). Not all information required for the database was directly available in every report, therefore, additional information was derived where possible. Where only cost per coral colony was provided, calculations for the restoration cost per area were estimated by assuming a transplanting schedule with four coral colonies outplanted per m^2 or 40,000 coral transplants per hectare (Edwards and Gomez, 2007). Accounting for a median survival of 60.9 percent (averaged over the reported pre-transplant, transplant, and post-transplant survival in the coral reef restoration database section), a total of 65,681 coral transplants would be required to populate one hectare. The latter value was used for converting cost per colony to cost per unit area. All reported restoration costs were adjusted for inflation in each respective country based on consumer price index (CPI) to a base year of 2010 prices. Data required for economic conversion were downloaded from World Bank Development Indicators (Group, 2014). For some countries and/or years, CPI data were unavailable; such observations were excluded from further analyses, but original data are available in the database. If the CPI for a particular year was unavailable, the next closest year selected was data collection year. Otherwise, if data collection year was unavailable, the publication year minus one year was used. For restoration costs incurred in 2018, the previous year CPI (2017) was used for conversion. For studies where local currencies were reported, data were first converted to U.S. dollars using the foreign exchange rates from the Penn World Tables (Heston, Summers, & Aten, 2012) and later adjusted to the respective countries' inflation based on CPI to a base year of 2010 prices.

Data analysis

Percentages, counts and other quantifications from the database reference the total number of case studies with data in that category. Case studies where data were lacking for the category in question, or lack appropriate detail (e.g. reporting 'mixed' for coral genera) are not included in calculations. Many categories allowed multiple answers, (i.e. coral species); these were split into separate categories for calculations (e.g. coral species *n*). For this reason, absolute numbers may exceed the number of case studies in the database. However, percentages reflect the proportion of case studies in each category. We used Tableau to visualise and analyse the database (Desktop Professional Edition, version 10.5, Tableau Software).

4. CORAL RESTORATION IN A CHANGING WORLD

4.1 What is restoration?

The Society for Ecological Restoration International Science & Policy Working Group (2004) defines restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed". Also, "Restoration attempts to return an ecosystem to its historic trajectory". Restoration projects ideally require no attendance once they are mature. In comparison, rehabilitation emphasises "the reparation of ecosystem processes, productivity and services…" but does not necessarily mean a return to pre-existing biotic conditions. Rehabilitation projects may require some attendance once they are mature. A restored ecosystem "contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy" (all definitions from Society for Ecological Restoration International Science & Policy Working Group 2004). Following these definitions, building a coral nursery is not a restoration project in itself, but a tool for restoration.

Restoration can be passive or active, whereby passive restoration (also 'natural regeneration' or 'indirect restoration') "relies on increases in individuals, without direct planting or seeding, after the removal of causal factors alone", while active restoration (also 'direct restoration', and often shortened to just 'restoration') relies on reintroductions or augmentations (McDonald et al., 2016). Broadly speaking, these two types of restoration also correspond to the level of degradation sustained by the environment, where passive restoration can be applied to sites with less damage, and active restoration is considered necessary in areas where unassisted natural recovery is unlikely. Finally, an intervention is the action "undertaken to achieve restoration, such as substratum amendment, exotics control, habitat conditioning, reintroductions" (McDonald et al., 2016).

In this review, we have excluded passive interventions such as predator removal (e.g. crown-of-thorns starfish and *Drupella* control), unless they were conducted in conjunction with active restoration. Instead, we review active restoration interventions which reintroduce (e.g. coral fragment transplantation) or augment (e.g. substrate stabilisation) coral reefs, for the purposes of restoring the reef ecosystem. In the published literature and elsewhere, there are many terms that describe the same intervention. For clarity, we provide the terms we have used in the review, their definitions and alternative terms (Table 1).

Table 1: The intervention terms used in the review, their definitions and other common terms

Intervention	Definition	Other common terms
Direct transplantation	*Transplanting coral colonies or fragments without intermediate nursery phase*	*Coral tipping, post-disturbance repair*
Coral gardening	*Transplanting coral fragments with an intermediate nursery phase*	*Population enhancement, asexual propagation*
Coral gardening - Nursery phase	*Transplanting coral fragments with an intermediate nursery phase (used to describe case studies that only detail the nursery phase)*	
Coral gardening - Transplantation phase	*Transplanting coral fragments with an intermediate nursery phase, including outplanting juveniles raised in the nursery (used to describe case studies that only detail the transplantation phase)*	*Outplanting*
Coral gardening - Micro-fragmentation	*Transplanting micro-fragments from massive corals, with an intermediate nursery phase*	
Substratum addition - Artificial reef	*Adding artificial structures for purposes of coral reef restoration*	*Other terms: Engineered structures*
Substratum stabilisation	*Stabilising substratum to facilitate coral recruitment or recovery*	
Substratum enhancement - electric	*Enhancing artificial substrata with an electrical field or direct current*	
Substratum enhancement - Algae removal	*Enhancing substrata by removing macroalgae*	
Larval enhancement	*Using sexually derived coral larvae (often produced from eggs and sperm in in-situ flow-through facilities) to release at restoration site, after intermediate holding phase*	*Larval propagation, sexual propagation*

4.2 Where and how are interventions occurring?

We identified 329 case studies on coral restoration, of which 195 were from the scientific literature, 79 were sourced from the grey literature (i.e. reports and online descriptions), and 55 were responses to our survey for restoration practitioners (Figure 1). We identified 52 countries (Figure 1) in which coral restoration projects have occurred, with most projects conducted in the USA, Philippines, Thailand and Indonesia (together representing 40 percent of projects). Ten coral restoration intervention types are represented in the database, with the overwhelming majority of these involving coral fragmentation or transplantation of coral fragments (70 percent, Figure 1).

4.3 Temporal and spatial duration of interventions

Coral restoration case studies are dominated by short-term projects, with 65.6 percent of all projects reporting less than 18 months of monitoring of the restored sites. Overall, the median length of projects was 12 months, but this varied between project types. Survey respondents (i.e. coral restoration practitioners) tended to report longer projects (median 24 months), while grey literature and peer-reviewed projects reported a median monitoring period of 13 and 12 months respectively (Figure 2). This inconsistency between projects may be due to the short time-scale available for most research projects, linked to student projects or short-term funding (i.e. peer-reviewed and grey literature). The projects may be ongoing however publications report the duration of data collection until the time of publishing as opposed to survey respondents, who are more likely to report on the entire duration since restoration activities began (i.e. 'time in the water'). Similarly, most projects are relatively small in spatial scale, with a median size of restored areas of 500 m^2 (Figure 2). Research projects published in the peer-reviewed literature reported a median spatial scale of 300 m^2, while survey respondents and grey literature both reported larger spatial scales (median 500 m^2). Median values were used to describe spatial and temporal scales due to a substantial right-side (positive) skew in both data sets, with long tails.

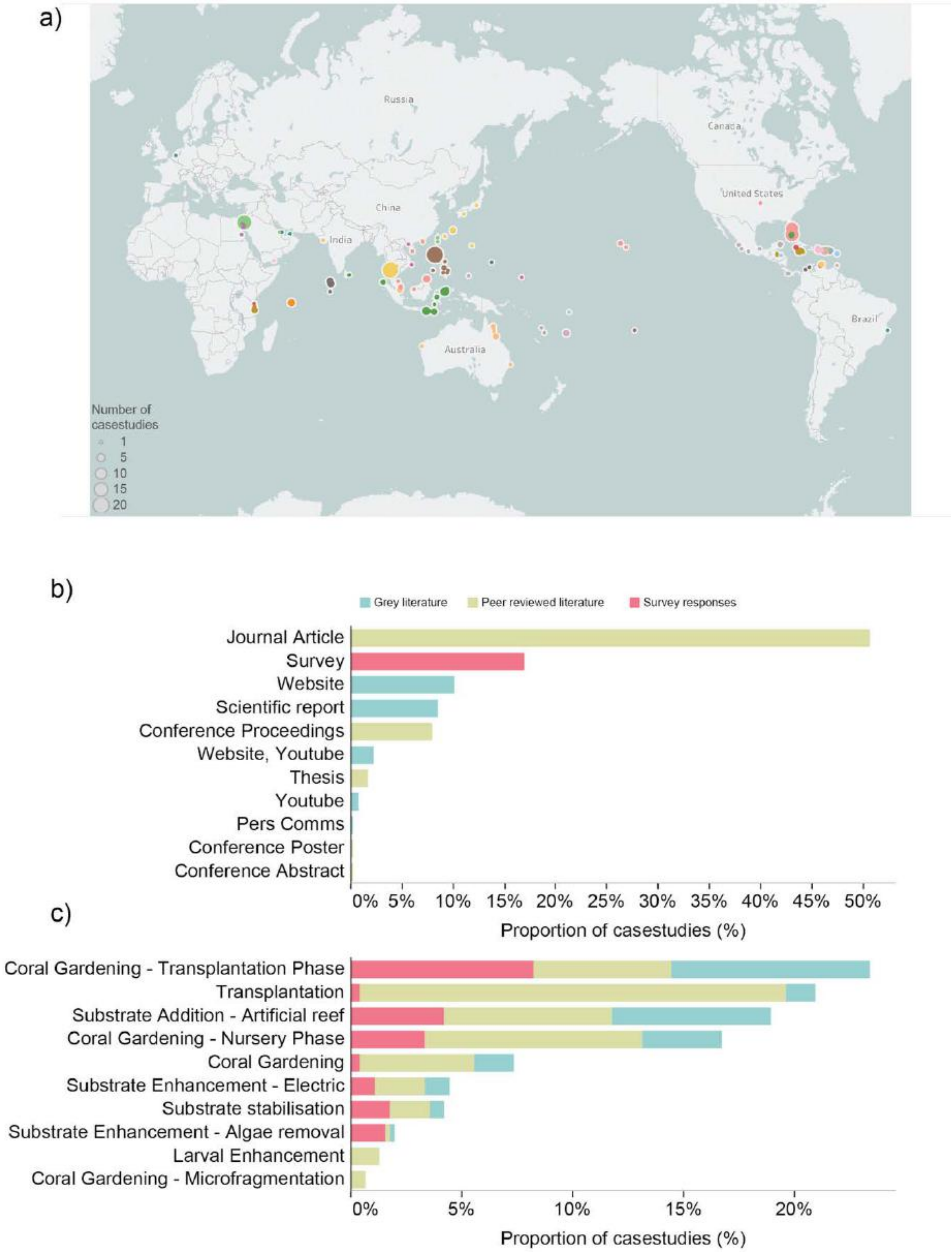
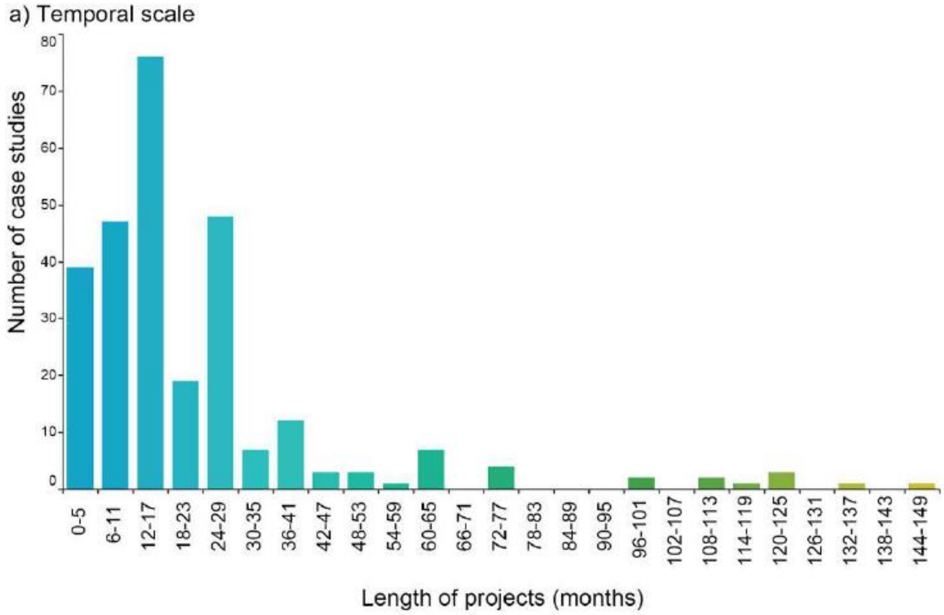


Figure 1: The a) location, b) source and c) type of intervention included in the review.



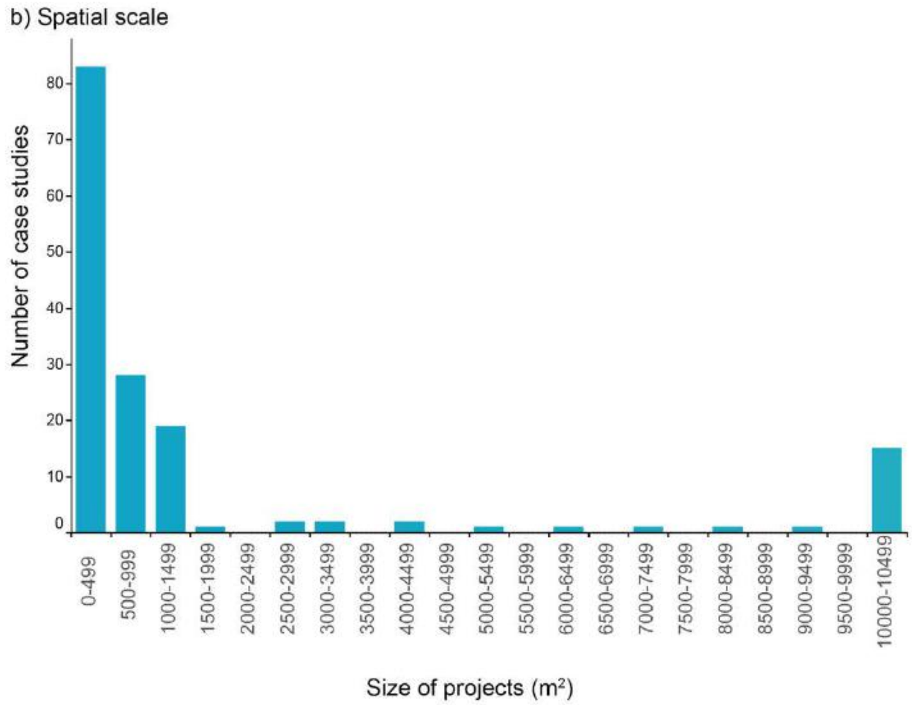


Figure 2: The a) temporal and b) spatial scale of coral restoration projects included in the review. Note that the x-axis in panel b has been truncated to only display projects up to one hectare. Full figure can be viewed in the online visualisation.

It is clear from our data that there is a general mismatch between the scales at which disturbances occur and the temporal scale of monitoring. The median length of monitoring is 12 months, and it is feasible to see a 12-month period without a major bleaching event, destructive storm or disease outbreak. However, most corals on a reef will experience these in a lifetime. While practitioners cannot be expected to subject their corals to every disturbance on the reef, these short monitoring times may artificially inflate the growth or survival rate, by intentionally or unintentionally avoiding stressors. For example, Fadli et al. (2012) described a successful restoration project in Indonesia, where coral cover, diversity and fish abundance improved dramatically on artificial reef modules after three years of deployment. However, almost 100 percent of these corals died in a bleaching event approximately six months after the conclusion of the study. While these authors reported this event in their publication, others may not know the event occurred at all or have little incentive to publish a failed experiment. This is a likely bias in our dataset overall, albeit one that is difficult to quantify. In an attempt to quantify this, we plotted the average survival against the length of monitoring, and we would expect a negative relationship between monitoring length and average survival if this bias was substantial. We found no such evidence in the data (Figure 4, note low replication of longer projects).

The longest study monitored a transplantation project for 12 years (Garrison and Ward 2012), and studies that lasted ten years or more (n=5) tended to be monitoring programs on artificial reefs or restoration sites with transplanted corals; these also tended to be larger in spatial scale (> 1,000 m^2) than the short-term studies. Similarly, studies with a larger spatial scale (greater than one ha or 10,000 m^2, n=25) were mainly monitoring projects of artificial reefs or coral transplantation sites. Unfortunately, despite being long-term projects of larger spatial scales, only a small proportion (28 percent) of these reported survival of corals (n=7, average survival 80 percent).

4.4 Corals used in interventions

Overall, coral restoration projects focused primarily (65 percent of studies) on fast-growing branching corals. Over a quarter of projects (26 percent) involved the coral genus *Acropora*, while nine percent of studies included a single species - *Acropora cervicornis* (e.g. Bowden-Kerby 2008; Mercado-Molina et al., 2014; Schopmeyer et al., 2017). Among all the published documents, the top five species (22 percent of studies) were *Acropora cervicornis, Pocillopora damicornis, Stylophora pistillata, Porites cylindrica* and *Acropora palmata* (Figure 3). Much of the focus on *Acropora cervicornis* and *Acropora palmata* is likely to have resulted from these important reef-forming species being listed as threatened on the United States Endangered Species List and as Endangered on the International Union for Conservation of Nature Red List of Endangered Species (IUCN 2018). Almost three quarters (72 percent) of case studies reported using more than one species in their restoration projects, while the remaining 28 percent used a single species. A diverse range of species are thus represented in the dataset, with 221 different species from 89 coral genera (Figure 3).

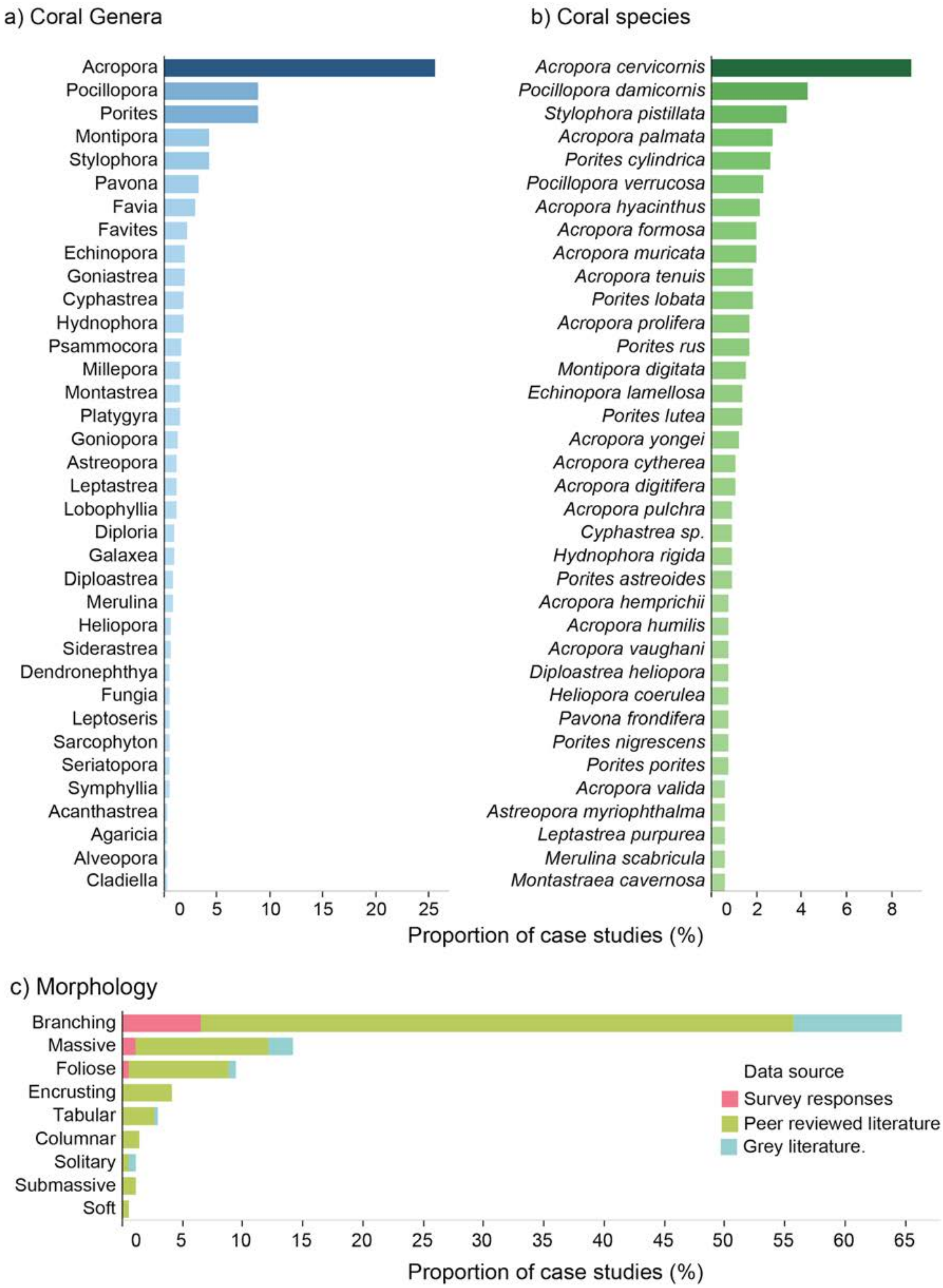


Figure 3: The a) species b) genera and c) growth morphologies of corals used in coral restoration interventions. Note: The y-axis for genera and species is substantially truncated for visual purposes. The complete species list can be viewed in the online database. A large proportion of survey respondents did not report species or genera (but opted for 'mixed'). The numbers reported here are therefore from the total number of case studies that reported on species or genera.

The average survival of corals varied between species and genera. Although we did not link survival to individual genera in cases where multiple genera were included in the same case study, some broad patterns are discernible. The overall average survival in this review was 69 percent (± SEM 1.5 percent), and most genera with substantial replication tended to fall in the 60-70 percent survival range (Figure 4). In fact, 11 out of 12 genera with >10 case studies reporting on survival fit within the 60-70 percent mortality range, while the single exception (*Hydnophora*) reported 75 percent survival. Low replication is a common factor in all genera with an average survival >90 percent, with no genera exceeding three case studies in this group (Figure 4). We argue that this highlights the variability between species within genera, and the critical role of environmental conditions in shaping the outcome of restoration projects. While individual projects may have successful outcomes, these data suggest that no particular genus is more or less suited to coral restoration than any other. However, an average survival of 69 percent is substantially higher than that reported in terrestrial ecological restoration, where outplant success tends to fall below 50 percent (e.g. Godefroid et al., 2011; Sweeney et al., 2002). Additionally, in contrast to terrestrial studies, and predictions in coral restoration reviews, there is no evidence of survival declining with increasing length of studies (Figure 4).

While these data argue against generic specificity in suitability to restoration, many studies suggest that some coral species and genotypes have higher survival and grow better than others in a reef restoration context (van Woesik et al., 2018). However, few studies have compared survival and growth (or other measures of successful restoration, such as reproductive variables) between different species. One of the few multi-species comparisons, conducted in the Philippines, found that survivorship was higher for slow-growing massive and robust corals than branching and foliose morphologies (Dizon and Yap, 2006a). However, the highest survival rate was measured for *Acropora palifera* (94 percent) and the lowest for *Acropora microphthalma* (eight percent), both branching species. Growth rate also differed between most species across all restoration sites in the study, with the highest growth rates in *A. microphthalma* and *Echinopora lamellosa* (Dizon and Yap, 2006a).

A separate study found that the highest mortality occurred in pocilloporids (between 71-82 percent) and *Acropora muricata* (71 percent), and the lowest mortality in *Pavona frondifera*, the blue coral *Heliopora coerulea* and *Porites cylindrica* (Dizon et al., 2008). A transplantation project of corals to seawalls using *Porites lobata*, *Pocillopora damicornis*, *Hydnophora rigida*, *Diploastrea heliopora* and *Goniastrea minuta* found that almost half of *P. lobata* fragments survived after two years, but *P. damicornis* and *H. rigida* suffered complete mortality after two months. Ninety percent of *G. minuta* and ten percent *D. heliopora* survived after 12 months (Ng et al., 2015). Transplanted fragments of *Porites rus* can be more environmentally tolerant than *P. cylindrica* (Yap 2004), but *P. cylindrica* tends to have faster growth rates (Custodio and Yap 1997). A comparison between the soft coral *Dendronephthya hemprichii* and *Stylophora pistillata* found different preferences for depth and substratum orientation (Oren and Benayahu, 1997). *Acropora prolifera* was more tolerant to being placed on sandy substrata than *A. cervicornis* (Bowden-Kerby 1997), and also had higher growth rates and branching rates in a different study (Chilcoat 2004). *Porites cylindrica* was more tolerant of differences in water movement and the species composition of the receiving community than *P. frondifera* (Cabaitan at al., 2015). All of this variability in responses within and between species usually interacts with environmental variables at the different reef sites and the specifics of the different restoration methods used (e.g. Dizon and Yap 2006b, Palomar and Gomez 2009).

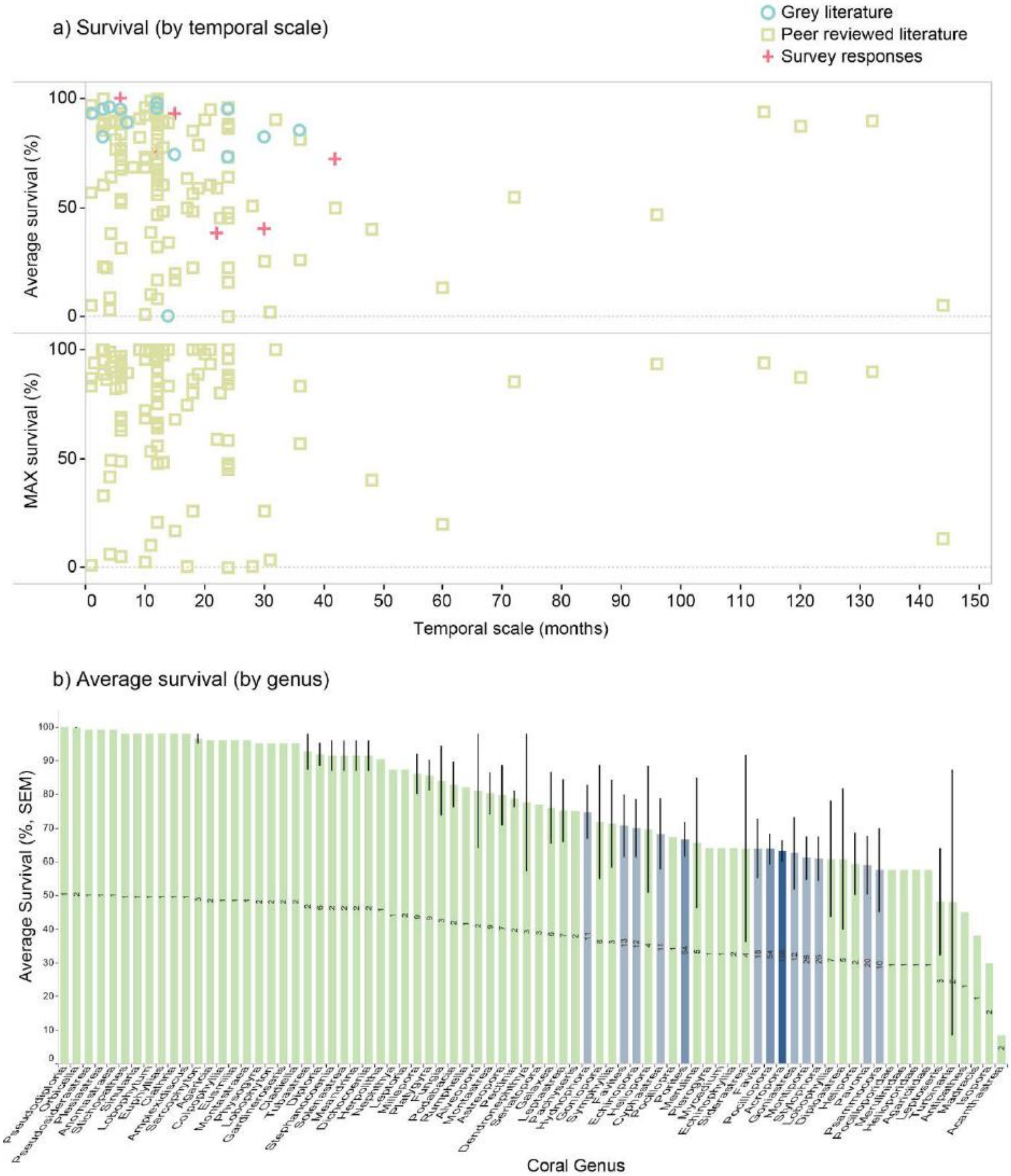


Figure 4: The average (a, top) and maximum (a, bottom) survival of corals in projects by monitoring time (months). Each data point is a study which reported both survival and monitoring time. Average survival (b) reported by case study by genera (±SEM). The numb number on each bar is the number of case studies reporting survival for that genus. Genera with >ten case studies are highlighted in blue. Note: survival represents an average reported by the case study, is not linked directly to each individual genus, and could therefore include survival estimates from other genera. Caution should be taken when interpreting these estimates. Refer to individual papers for details online.

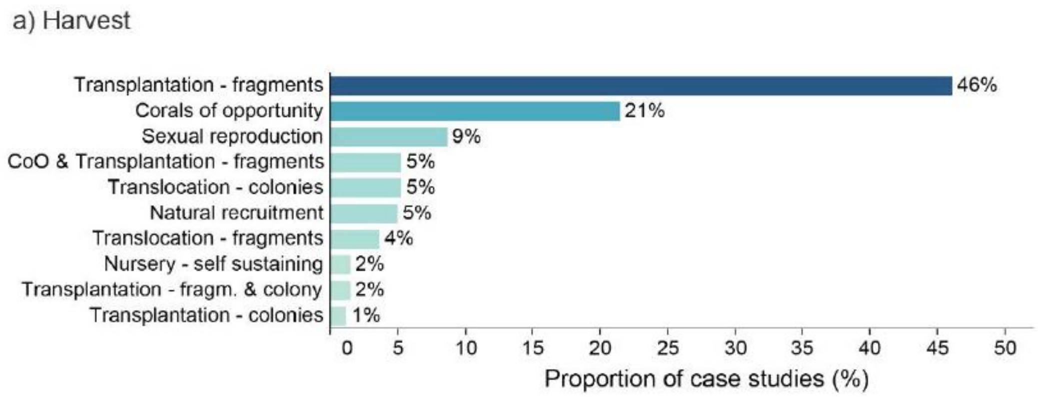
4.5 Intervention types

4.5.1 Transplantation

The earliest developed and most common method of coral restoration (used in 70 percent of reviewed projects) involves transplantation of coral fragments, which, in essence, could be seen as a simulation of asexual reproduction through fragmentation (Maragos 1974; Edwards and Gomez 2007). This technique is also called 'asexual propagation' or 'fragmentation'. This method bypasses the post-settlement demographic bottleneck of coral larvae, characterised by high mortality and slow growth, common in the life-histories of many hard-coral species (Clark and Edwards 1995, Lindahl 1998, Zimmer 2006; dela Cruz and Harrison 2017). Since pioneering fragmentation experiments to measure coral growth in the early 20th century (Vaughan 1916), the technique has evolved into multiple interventions specifically aimed at coral restoration. Options that have been explored include direct transplantation, the use of an "intermediate" nursery phase (also described as "coral gardening"), and micro-fragmentation. Regardless of the methods employed, interventions with coral transplantation have three steps in common, 1) the collection of corals, 2) transport, and 3) transplantation (also often called "outplanting") onto the restoration site. Below we describe the general methods shared by each transplantation technique in terms of harvest, transport, attachment and outplanting design, and then explore details about each specific intervention type.

4.5.2 Harvest

The primary method of sourcing corals is by harvesting fragments from nearby donor reefs (46 percent of case studies, Figure 5). Fragments are often removed from donor corals using surgical bone cutters, wire cutters (Johnson et al., 2011) or simply by hammer and chisel. While larger fragments generally have higher survival rates (e.g. Bowden-Kirby 2001), collecting large fragments can be detrimental to donor colonies. It is commonly accepted that harvesting ten percent or less of live tissue from donor colonies prevents significant sub-lethal effects (e.g. Edwards and Gomez 2007, Schopmeyer et al., 2017). In 22 percent of case studies, a non-destructive method of coral collection was used, whereby fragments that are already detached from corals are collected ('corals of opportunity', e.g. Schuhmacher et al., 2000, Bruckner and Bruckner, 2001, Monty et al., 2006). These coral fragments were dislodged either through natural processes such as wave action, fish activity or by mechanical disturbances such as ship groundings.



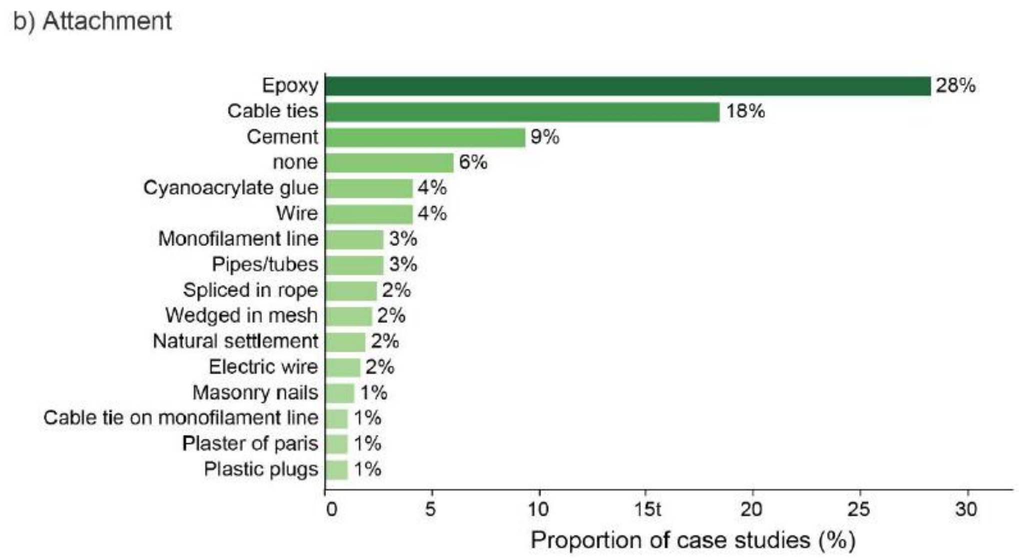


Figure 5: The sources of coral used in coral restoration case studies (a), and their method of attachment (b). Note: for visual purposes attachment methods with less than two case studies were excluded from the visualisation. The complete list can be viewed in the online database.

Recommendations for harvest

1. Do not harvest more than ten percent of donor colony.
2. The size range of fragments to be cut from the donor colony is ideally 10-30cm.
3. Where possible, harvest fragments that have already broken off, but are still alive or partially alive.
4. Use tools that cause minimal damage to the donor colony.
5. Consider the rest of the coral community when selecting donor colonies.

4.5.3 Transport

A wide variety of methods have been used to transport coral fragments from the donor to the restoration sites, and largely depend on the conditions at each site and the distance involved. If the restoration site is close by, fragments can be transported underwater in crates or bins by scuba divers (e.g. Clark and Edwards 1995, Okubo et al., 2004), but if the site is further away, transport by boat may be necessary. While it is generally recommended to reduce exposure to air and sunlight by transporting fragments in seawater and providing shade (Johnson et al., 2011), responses to transport method vary between species. For example, Kaly (1995) found that while *S. pistillata* and the gorgonian *Rumphella* sp. responded negatively to exposure to air during transport (two hours under a wet tarpaulin), *Acropora gemmifera* and *Favia stelligera* did not show an equivalent negative response. Similarly, Harriott and Fisk (1995) reported no significant differences in survival between corals transported in water and those exposed to air for less than an hour. However, when exposure exceeded two hours, survival rates dropped significantly.

Recommendations for transport

1. If restoration site is close, transport can be done by divers underwater.
2. If transport is by boat, exposure to air should be minimised.
3. If transport is greater than two hours, corals should be transported in seawater with shade.

4.5.4 Attachment

Once corals are at the restoration site, it is generally accepted that attaching fragments to hard substrata results in higher survival than merely placing them onto the seabed (e.g. Becker and Mueller 2001, Forrester et al., 2011). Being firmly attached to the substratum allows for coral tissue overgrowth and attachment to the benthos, while even slight movement of fragments can prevent attachment (e.g. Kaly 1995). Survival of loose fragments may also depend on the substratum type. For example, in an experiment testing different attachment methods, no fragments survived when scattered on sand, while approximately 80 percent of fragments scattered on coral rubble survived (Bowden-Kerby 1997). The most common method of attaching corals was with epoxy (30 percent), primarily in the form of a putty that hardens when two components are mixed underwater. Other common attachment methods included cable ties (i.e. ‘zip ties’ 20 percent) and cement (11 percent, Figure 5). Cable ties can be attached to nails or stakes driven into the substratum or dead corals present on the restoration sites (e.g. Ross 2012; Hernándes-Delgado et al., 2014; Lirman et al., 2014). Underwater cement can be premixed and squirted into cracks and crevices where fragments can be lodged (e.g. Alcala et al., 1982, Dizon et al., 2006). Alternatively, cement can be cast into disks *ex-situ* with the base of coral fragments encased in the cement, or attached with glue (e.g. Ferse 2010, Johnson et al., 2011, Bowden-Kerby 2014). Ultimately, providing that fragments are attached in a way that does not permit movement, there appear to be negligible differences in growth or mortality between different types of attachment (Forrester et al., 2011, Dizon et al., 2008). This is reflected in our data as well, where the average survival of corals between the most common attachment methods was between 60-70 percent (online figure). This suggests that, in terms of growth and survival, the method of attachment can be tailored to what is available and suitable for the reef site and project. However, these methods differ markedly in terms of time and labour costs (i.e. scattering fragments is substantially faster than gluing fragments to precast cement pucks). Finally, it should be noted that attachment methods need to be suited to the species to be restored and wave energy at the restoration site. Slow growing species in high energy environments require the most secure attachment, while fast growing species in low wave energy environments will need the least.

Recommendations for attachment

1. Secure fragments to encourage self-attachment to substratum - even small movements will prohibit attachment
2. Hard substratum is preferable over soft substratum
3. Attachment method can be tailored to environmental conditions and material availability - experiments show no major difference in survival or growth as long as fragments are properly secured.

4.5.5 Outplanting design

Finally, the density, pattern and species composition planted onto the restoration site can also affect the survival and growth of transplanted corals. However, due to the paucity of studies that have explored these factors experimentally, most such considerations appear to be species-specific. For example, when transplanting corals in monospecific versus mixed-species groups, Cabaitan et al., (2015) demonstrated that species composition affected survival for some corals (*P. frondifera*), but not others (*P. cylindrica)*. Similarly, while some studies report that the highest growth and survival is achieved when corals are transplanted in the same orientation as they were growing in when harvested (Okubo et al., 2005, Nakamura et al., 2011, Gomez et al., 2014), others report no effect of transplant orientation on the survival of coral fragments (Bongiorni et al., 2003, Becker and Mueller 2001). Recent studies have explored how the density and arrangement of coral fragments can influence their growth and survival, and reef biodiversity (dela Cruz et al., 2014). For example, Ladd et al., (2016) demonstrated that fragments of *A. cervicornis* tend to grow faster and lose less live tissue at moderate transplantation densities (three corals m^{-2}) than when planted closer together. However, the relationship between transplantation density and the health of transplants was not linear, with lower density groups (0.75 and 1.5 corals m^{-2}) exhibiting slower growth than intermediate density groups. Similarly, Griffin et al., (2015) demonstrated that *A. cervicornis* fragments tended to grow more vertically in larger groups, presumably due to increased competition for light and food with neighbouring fragments. In support of the intermediate densities for outplanting, some practitioners have found that predation by fire-worms and corallivorous snails increased when colonies were outplanted close together (Case Study 4, Johnson et al., 2011). These experiments were conducted using a single branching coral species (*A. cervicornis*), and recommendations may vary with other species or growth morphologies. Indeed, Shaish et al., (2010) found no difference in survival or growth of *Montipora digitata* fragments transplanted ten or 20 cm apart.

Recommendations for outplanting design

1. Transplant colonies at densities of approximately $3m^{-2}$, to increase growth, survival and potentially reduce susceptibility to coral predators
2. Consider transplanting corals in the same orientation as they were in when collected

4.5.6 Direct transplantation

Direct transplantation is one of the earliest coral restoration methods to be developed (e.g. Maragos 1974, Birkeland et al., 1979), and involves the harvesting of corals (fragments or whole colonies) at a donor site for transplantation at a recipient restoration site, without an intermediate nursery phase. There are 94 descriptions of direct transplantation in the review database, representing 21 percent of all records. This intervention was more common in the peer-reviewed literature, with only two and five records emerging from the survey and grey literature, respectively (refer to general description of harvest and attachment above). Three quarters of case studies of direct transplantation harvested fragments (60 percent) or whole colonies from nearby reefs (16 percent). This method assumes that the donor reef can withstand harvesting (Epstein et al., 2001) and that the receiving degraded reef is subject to conditions that are favourable for coral growth and reef establishment (Dizon and Yap, 2006). While the harvesting of whole colonies may be less sustainable than harvesting of fragments, it is most common in programs aimed at salvaging corals from planned construction activities that would otherwise kill them (e.g. biodiversity offsets, Plucer-Rosario & Randall 1987, Newman & Chuan 1994, Thornton 2000, Gayle et al., 2005, Seguin et al., 2008, Yeemin et al., 2006, Kilbane et al., 2008, Kenny et al., 2012, Young et al., 2012, Rodgers et al., 2017, Kotb 2016).

The success of direct transplantation depends on the size and health of the fragments, the method of transportation and attachment, and other extrinsic factors such as environmental conditions in the months following the transplantation, when coral fragments are stressed and vulnerable (van Woesik et al., 2017). Overall, direct transplantation studies reported an average survival of 64 percent, with 20 percent reporting >90 percent survival of transplanted corals (online figure). Direct transplantation has primarily involved fast-growing corals, with more than three-quarters of case studies using branching coral morphologies. Fish abundance, biomass and diversity can increase rapidly when compared to denuded control reefs (Cabaitan et al., 2008; dela Cruz et al., 2014; Opel et al., 2017), but very few studies have monitored the longer-term results of direct transplantation on coral reef communities beyond the survival and growth of the fragments themselves. Fish and invertebrate communities can develop in density and species richness to mimic undisturbed reef in a relatively short time (Yap, 2009); coral reef fishes can respond rapidly to the change in benthic composition and rugosity at a restored reef (Opel et al.2017). Coral colonies relocated from dredging or construction areas may thrive in a suitable new location (Kotb, 2016; Rodgers et al., 2017). However, the paucity of studies that have monitored restored corals for longer than 12 months highlights the need for caution when generalising the potential to re-create viable coral populations and communities.

Among peer-reviewed studies that reported the results of direct transplantation, 71 percent reported a successful outcome based on the parameters measured by the authors (usually survival and growth). Comparatively, 100 percent, and 60 percent of survey respondents and grey literature documents, respectively, reported a successful outcome. Success, in the short term (12 months or less), was usually expressed as a survival rate of at least 50 percent and some measure of positive growth. Interestingly, one study reported a successful outcome despite a survival rate of only 40 percent, but this was an average of attached and unattached fragments, and the authors concluded that attaching fragments to the substratum led to much higher survival (Forrester et al., 2011).

Due to the wide variety of species used in direct transplantation studies, and therefore the low replication within species, it is difficult to draw conclusions at a species level. Similar to the overall findings from the study, the genera with the highest survival tended to be those with low replication, suggesting that survival may be less species and genera specific and more related to the intervention type, and to the environmental conditions at the restoration sites. Out of the three morphologies represented in the dataset, solitary corals had the highest survival (survival 77 percent, Salvat et al., 2002, Bouchon et al., 1981), followed by massive morphologies (64 percent), and foliose corals (60 percent, online figure). A few studies that measured variables beyond survival and growth reported successful spawning (Okubo et al., 2004), increased coral cover (Miyazaki et al., 2010; dela Cruz et al., 2014), increased fish species richness and abundance (Cabaitan et al., 2008; dela Cruz et al., 2014; Rodgers et al., 2017), fish recruitment (Bowden-Kerby 1997) and macroinvertebrate abundance (dela Cruz et al., 2014).

The main lessons shared by direct transplantation concern which species, transport and attachment methods and receiving environments tend to yield the best results in terms of survival and growth of fragments. Transplanted fragments and colonies fare better when attached to the substratum with adhesive or tied to metal stakes or poles; unless the restoration site is in perpetually still waters, re-attachment rates of unattached fragments are too slow to withstand the effects of water movement (Bowden-Kerby 1997; Guest et al., 2011). Successful establishment of transplanted corals is often species-specific, and tends to work best with fast-growing species (Garrison and Ward 2012; Miyazaki et al., 2010; dela Cruz et al., 2014). The density of transplanted fragments and the identity of neighbouring species can also affect the survival and growth rates of direct transplants (Raymundo 2001; Lindahl 2003). The seasonality of transplantation and the quality of the receiving environment were also found to be important (Yap and Gomez 1984, 1985), but this is true of any transplantation or translocation project.

Recommendations for direct transplantation:

1. Do not harvest more than ten percent of donor colony.

2. If transport is greater than two hours, transport in seawater and provide shade.

3. Secure fragments to encourage self-attachment to substrate - even small movements will prohibit attachment.

4. Attachment method can be tailored to environmental conditions and material availability - experiments show no major difference in survival or growth as long as fragments are properly secured.

5. Transplant colonies at densities of approximately three m^{-2}, to increase growth, survival and potentially reduce susceptibility to coral predators.

6. Insufficient data exist for most species or genera, however *Acropora cervicornis* specifically, the *Acropora* genus broadly, and branching corals in general tend to achieve approximately 60-70 percent survival. For this reason, and based on the stoplight approach of Schopmeyer et al., (2017), we suggest setting 70 percent survival in outplanted corals as a benchmark target of success.

7. We highly recommend and encourage monitoring of outplanted corals for as long as possible. In particular, we encourage monitoring and reporting of additional ecological metrics (beyond purely biological, e.g. growth and survival of corals), including topographic complexity, fish abundance and richness, invertebrate biodiversity, and coral cover of different coral groups. See Hein et al., (2017), for a comprehensive review of metrics of success that can and should be monitored when possible.

8. Due to the inherent unsustainability and limited scalability of long-term harvesting of donor coral fragments, we suggest that direct transplantation is most suitable to small-scale restoration projects in response to a one-off acute disturbance.

4.5.7 Coral gardening

Continuous harvesting of coral fragments may have detrimental effects on donor corals and populations. In response to this, a more sustainable model has been developed where coral recruits or small fragments are raised in intermediate nurseries, prior to outplanting on restoration sites. While growing in the nursery, the coral fragments are being regularly maintained (e.g. by removing any algal growth) and are safe from predation, storm surges or wave energy. Coral gardening, also referred to as “coral aquaculture” or “coral farming”, is essentially mariculture of coral fragments for the purpose of coral reef restoration. The technique was modelled on silviculture, where trees are grown from seeds in nurseries, and later outplanted to restore degraded forests (Rinkevich 1982, 1995, 2000). This technique is used for both commercial and reef restoration purposes. Additionally, this technique is commonly used to deliver community-based conservation and stewardship activities with a strong focus on socio-economic values. The general goal of coral gardening for restoration purposes is to protect corals from damaging conditions during their most vulnerable stages, with the intention of planting them onto damaged reefs once they have reached a size threshold at which their survival post outplanting would be high (Rinkevich, 2005). In this review, 46 percent of case studies involved coral gardening, with a majority of records focusing on the transplantation phase of the concept (transplantation phase 23 percent, nursery phase 17 percent, both phases seven percent). This technique has been extensively reviewed in prior publications, and so we briefly summarise the patterns observed in the current review. For detailed descriptions of methods and techniques we direct the reader to Edwards and Gomez (2007), Johnson et al., (2011), and the Coral Restoration Module of the Reef Resilience website www.reefresilience.org.

Nursery phase

Corals are raised in either field-based (*in situ*), or land-based *(ex situ*) nurseries, depending on local conditions. Field-based nurseries are best placed in sheltered environments where they can be closely monitored and where conditions are favourable for the survival and growth of coral fragments. Land-based nurseries consist of aquaria or tanks where environmental conditions can be controlled. A multitude of different field-based nurseries have been designed, tailored to different environmental conditions. Coral nurseries can be in the form of structures placed on the substratum such as concrete bases, tables or frames (e.g. Mbije et al., 2010; Toh et al., 2017), or mid-water structures such as ropes (Shaish et al., 2008, Bowden-Kerby 2001) or PVC ‘trees’ (e.g. Nedimyer et al., 2011), or dead coral bommies (dela Cruz et al., 2015). Fixed tables and nursery trees were the primary coral nursery method among survey respondents, while there was more diversity in nursery types in the published literature. One of the few studies that compared the merits of field-based versus land-based nurseries found little difference between the two methods. Becker and Mueller (2001) explored the difference in survival and growth of *A. palmata* and *A. cervicornis* in *ex situ* and *in situ* nurseries and found no difference in extension rates of *A. cervicornis*, but greater linear growth in *A. palmata* fragments reared in the field compared to tanks. Both species exhibited larger basal growth of fragments in tanks compared to field nurseries. One study explored the use of bamboo as material for a nursery but reported a failure of the coral fragments surviving due to disintegration of the material underwater (Ferse 2010).

Nurseries are stocked by removing tissue and skeleton (from a few polyps to small branches) from healthy wild coral populations, collecting “corals of opportunity” (corals fragmented through disturbance), or collecting propagules from adult colonies spawning in captivity (Edwards and Gomez, 2007). Species considered favourable for coral gardening are those with high growth rates, rapid healing capacity, a natural tendency to use fragmentation for sexual reproduction and tolerance to a range of ambient conditions (Young et al., 2012). For example, *A. cervicornis* is used in coral nurseries throughout the Caribbean, because, besides its historical importance for providing habitat structure and its current endangered status, it possesses these qualities (Lirman et al., 2014). Increasingly, coral nurseries are developed to be self-sustaining after the first wild collection of a 'mother'-colony, which are used to produce subsequent generations of coral fragments (e.g. case studies in Johnson et al., 2011), although this does raise long term issues around the genetic diversity of restored populations.

Advocates of using a coral nursery phase for reef restoration point to improved growth and survivorship rates of fragments, compared to direct transplantation; survival rates of >75 percent are common (Putchim et al., 2008; Shaish et al., 2008). This is not echoed in our dataset, where direct transplantation studies reported 64 percent average survival, while coral gardening studies (i.e. those with an intermediate transplantation phase) reported an average 66 percent survival in the outplanting phase (including studies that report only on outplant success, and those that include both stages of coral gardening). The average survival of fragments in case studies that reported survival of corals in nurseries (n=34) was 73 percent. However, another important factor in assessing the use of nurseries is the inherent cost and maintenance required to keep corals in a healthy condition before they are transplanted. For instance, all structures require a degree of cleaning and maintenance to prevent growing corals being overgrown or smothered by fouling organisms (Precht 2006). Recently, trials of attracting invertivorous fishes to nurseries proved successful and cost-effective (Frias-Torres and van de Geer, 2015), but studies that seek cost-cutting mechanisms are scarce.

There is a need to question whether the direct comparison of survival rates between direct transplantation and coral gardening is valid, given that coral gardening experiences mortality rates at each stage (i.e. nursery and outplanting). The ultimate survival of fragments may be more accurately expressed as a proportion of nursery survivors. This is particularly true in nurseries that are not self-sustaining (i.e. rely on harvesting fragments from the reef for each generation of outplants). Accordingly, we calculated the actual survival for fragments from case studies by expressing the average outplant survival (66 percent) as a proportion of the average nursery survival (73 percent). This reveals that the true overall survival of corals in these case studies could be as low as 48 percent. While this calculation may not be valid for all case studies in this review, it does highlight the challenge in comparing survival rates between intervention types, and the need for standardisation in reporting outcomes.

Recommendations for the nursery phase

1. Field based nurseries tend to be more successful and less expensive than laboratory nurseries
2. Choose a site that is always sheltered
3. Fixed tables and coral trees seem to give the best return for investment
4. Use material of long durability underwater for your nursery
5. Grow fragments to a size that offers a high likelihood of survival when outplanted (species-specific)
6. Keep the nursery free of corallivores, pests and fouling organisms

7. Evaluate site specific conditions to select the location for nursery, considering access to reef fishes which may help reduce macroalgae, but also allow corallivores access to vulnerable fragments

4.5.8 Transplantation phase

Ideally, nursery-reared corals are transplanted (or ‘‘outplanted’’) from nurseries to reef restoration sites to bridge spatial gaps between existing populations, enhance local coral abundance, supplement genetic and genotypic diversity, promote natural recovery through the restoration of sexually reproductive populations, and create habitat structure for the colonisation of sessile and mobile reef organisms (Lirman and Miller, 2003).

Survival of nursery-reared corals transplanted to restoration sites is dependent on a number of factors, including the size of fragments, their genotype and health, the season of outplanting, techniques used to secure colonies to the reef, the physical environment of the restoration site, the presence and abundance of coral predators, and the substratum type and benthic community (see also “Direct transplantation” above; van Woesik et al., 2018). For instance, macroalgal cover in the receiving environment can overgrow outplanted colonies (van Woesik et al., 2018), and sediment can smother fragments and impede growth after transplantation (Ng and Chou, 2014). In the absence of disturbance, a study comparing different coral restoration studies using *A. cervicornis* in the Caribbean reported high survival rates of transplanted corals within the first two years (Schopmeyer et al., 2017). Before outplanting, nursery-grown corals have to be resilient and large enough to sustain themselves without the need of any further human intervention.

Similar to direct transplantation studies, the success of transplanted corals in creating viable coral populations is usually only monitored for a short time. Fifty percent of all studies monitored restored coral populations for 12 months or less, and very few of these studies recorded anything other than biological variables such as survival and growth of transplanted corals. Survival rates need to be measured in the context of the proportion of fragments or colonies surviving the nursery stage (see above). Thus, it appears that the emphasis of coral gardening is still on optimising nursery techniques and ensuring survival in the first year of outplanting, rather than on following the results of outplanting towards the creation of a viable coral community that enhances or restores the overall reef system. Interestingly, Chamberland et al., (2015) monitored outplanted corals for 31 months, and reported a 3.4 percent survival rate as a successful outcome. It may be that this is the level of survival to be expected in the long-term, but data to support this are lacking.

There is a consensus among coral gardening practitioners that corals reared in nurseries are best allowed to reach a certain size before outplanting, and that this size may vary between species (Raymundo et al., 1999). Many practitioners focused their discussion about “lessons learned” on the cost and effort of coral gardening. Others gave details on the necessary maintenance of corals in nurseries, and how to address problems such as algal overgrowth and dislodgement by predatory fishes. When ecological variables were monitored, there was generally an increase in coral cover (Nakamura et al., 2010), fish abundance (Mbije et al., 2013; dela Cruz et al., 2014) and natural coral recruitment (Mbije et al., 2013). In areas where corals or coral fragments had been outplanted, ongoing success was most often hampered by natural disturbances, such as crown-of-thorns starfish outbreaks (Mbije et al., 2013).

In contrast to ex-situ nurseries, which are isolated and relatively sterile, in situ coral nurseries and transplantation sites are open to recruitment of reef organisms, including fishes and invertebrates that may harm young corals (Horoszowski-Fridman et al., 2015, Frias-Torres et al., 2015). Whilst in some cases fishes may inadvertently clean algae and biofouling organisms from around the growing corals, they may also prey on the corals, damage them during grazing (Horoszowski-Fridman et al., 2015), or, in the case of territorial damselfishes, grow algae that can smother or compete with the corals (Williams et al., 2018). Further, there is ample anecdotal evidence that reef fishes like parrotfish target vulnerable outplants over existing coral colonies of the same species. Similarly, multiple case studies mentioned predation by *Drupella* snails on both outplanted corals and *in situ* nurseries (e.g. Clarke and Edwards 1994, Van Treeck et al., 1997, Shafir et al., 2006, Dizon et al., 2008), suggesting that these gastropods are attracted to corals disturbed during interventions. Site specific conditions should therefore be considered during the planning phase of restoration projects.

Recommendations for the outplanting phase

There is a degree of overlap between recommendations for outplanting, and those for direct transplantation (above).

1. Secure fragments to encourage self-attachment to substratum - even small movements will prohibit attachment.

2. Attachment method can be tailored to environmental conditions and material availability - experiments show no major difference in survival or growth as long as fragments are properly secured.

3. Transplant colonies at densities of approximately three m^{-2}, to increase growth, survival and potentially reduce susceptibility to coral predators.

4. Insufficient data exist for most species or genera, however *Acropora cervicornis* specifically, the *Acropora* genus broadly, and branching corals in general tend to achieve approximately 60-70 percent survival. For this reason, and based on the stoplight approach of Schopmeyer et al., (2017), we suggest setting 70 percent survival in outplanted corals as a benchmark target of success.

5. We highly recommend and encourage monitoring of outplanted corals for as long as possible, or at least until evidence is acquired that the restored coral reef is resilient, can sustain itself and does not require any further interventions. In particular, we encourage monitoring and reporting of additional ecological metrics (beyond purely biological, e.g. growth and survival of corals), including topographic complexity, fish abundance and richness, invertebrate biodiversity, and coral cover of different coral groups. At the very least, monitored metrics should be tailored to the objectives of the project. See Hein et al., (2017), for a comprehensive review of metrics of success that can and should be monitored when possible, and Suding et al., 2015 for a global perspective of restoration frameworks.

6. Due to the inherent unsustainability and limited scalability of long-term harvesting of donor coral fragments, we suggest that direct transplantation is most suitable to small-scale restoration projects in response to a one-off acute disturbance.

4.5.9 Genetic diversity in coral gardening

If the goals of restoration are to include resilience to existing or future stresses, the consideration of genetic diversity is crucial (Carne et al., 2016). Acroporids, which are used preferentially in coral gardening, naturally reproduce asexually through fragmentation, so the recommended genetic diversity ratio reflects the proportion of unique genotypes per number of colonies sampled in a specific stand or thicket (Carne et al., 2016). The clonal processes preferentially used in coral gardening inherently limit resilience; assisted fertilisation (Calle-Triviño et al., 2018) or creating nursery stocks from the larvae of brooding corals (Linden and Rinkevich 2011) could be valuable tools for maintaining genetic diversity in coral gardening. The NOAA recovery plan (NOAA 2017) suggests a target genetic diversity ratio of 0.5 for both *A. cervicornis* and *A. palmata* (Carne et al., 2016).

Three peer-reviewed studies identified in this review have specifically tested the viability or performance of different genets of the same species in a coral gardening or transplantation context, especially for the purposes of recommending stronger genets for transplantation (e.g. Ross 2014). In *A. cervicornis*, it is understood that there are strong differences in growth rate and susceptibility to temperature, fouling and abrasion stresses between genets (Ross 2014, Ladd et al., 2017); there can be up to a six-fold difference in relative growth based on genotype alone (Bowden-Kerby at al. 2008). There are also differences in thermal tolerance, which will become one of the most important factors in a warming ocean (Ladd et al., 2017). Differences between genets that are apparent in the nursery may be less obvious in outplanted corals (Ross 2014), as characteristics of the receiving environment may have overriding effects on survival and growth (Bowden-Kerby et al., 2008). Furthermore, genotypes considered poor or strong survivors or growers may not perform consistently in different locations or years, and should not be dismissed (Goergen et al., 2018).

Acroporids are broadcast, hermaphrodite spawners, and cross-fertilisation can be limited by distance between colonies and populations; multiple genets of each species are ideally placed in proximity to each other to facilitate heterozygosity in mass spawning events (Baums 2008, Young et al., 2012). It is also important to consider that the genotypic composition of restored coral populations is at least as important as genotypic diversity for restoration success (Ladd et al., 2017). The effects of different levels of genetic diversity on the long-term persistence and function of a restored reef remains to be explored.

Recommendations for genetic diversity

1. Choose a variety of genotypes for each species.
2. Nurseries should not exclude “weaker” genotypes with lower productivity, as these characteristics may not be temporally or geographically consistent.

4.5.10 Micro-fragmentation

Less than five percent of transplantation studies have been conducted with slow growing life histories. Massive corals have largely been overlooked, mainly due to their slow growth and thicker skeletons, which are less amenable to fragmenting (Page and Vaughan 2014). However, recent research from Mote Marine Laboratory, based on decades of aquarist experimentation, has developed a ‘micro-fragmentation’ technique that enables massive and encrusting corals to be mass-produced and outplanted using concepts developed for coral gardening (Page and Vaughan 2014; Forsman et al., 2015).

A diamond blade saw is used to cut small fragments (one cm^2) of massive corals, which are then mounted on tiles. The tiles are kept in artificially lit and aerated aquaria for several weeks, after which they are placed in large outdoor flow-through aquaria. After approximately 12 months, the fragments can either be further sub-divided to generate new micro-fragments or outplanted. Micro-fragments that are secured to reef substrates or dead coral bommies in an array will readily fuse together to form a larger colony (i.e. 're-skinning'). The technique has been tested on ten massive coral species, with emphasis on three species determined to be most suitable for large-scale field trials (*Montastrea cavernosa*, *Orbicella faveolata* and *Diploria clivosa*; Page and Vaughan 2014). The research outcomes show high survival and rapid growth of fragments (>99 percent survival, Page and Vaughan 2014; Forsman et al., 2015). Recent research has shown that re-skinned corals reached sexual maturity more rapidly (~18 months) than naturally growing corals (~ten years) (T. Vardi *pers. comms*).

To date, no study has reported results of micro-fragmentation outside controlled aquarium environments. However, some are beginning to test the effects of fragment size and different environmental conditions on fragment health. Hall et al., (2015) found that pH interacts with fragment size to affect physiology and recovery from lesions in a species-specific way. Forsman et al., (2015) tested growth and survival of fragments in two tanks; one cleaned and maintained tank, and one established mesocosm tank that contained other corals, reef organisms and fishes. They reported faster growth but lower survival of fragments in the established tank, suggesting that results may vary when tested in a natural reef environment, however these results require further replication.

Similar to coral gardening, and many active interventions currently in use, micro-fragmentation relies on an intermediate phase of *ex situ* nursery rearing. Intuitively, substantive cost reduction and efficiency increases could be gained by forgoing the intermediate nursery phase, but this could also lead to decreased survivorship. This could be an important avenue for future research, as no current published literature exists on benefits of nursery rearing versus direct transplantation on the reef for micro-fragmentation studies.

Recommendations for micro-fragmentation

1. Cut fragments of at least one cm^2.

2. Fragments are best placed onto receiving colony in an array, rather than as single fragments.

4.5.11 Artificial reefs

About one fifth of projects (21 percent) described in the review involve the creation of substratum, such as artificial reefs. The creation of substratum involves structures that are placed on the seabed deliberately, sometimes to mimic characteristics of a natural reef, or for the purpose of increasing potential habitat for reef assemblages, fisheries yield and production, recreational diving opportunities and the prevention of trawling. The structures are usually placed on, or attached to, substratum that has been damaged (e.g. ship grounding sites) or that is otherwise unsuitable for coral settlement (e.g. loose rubble or soft sediment). In many cases, artificial reefs are deployed in conjunction with other methods, such as coral transplantation; this tends to enhance the subsequent colonisation of the artificial reefs by other organisms (e.g. Perkol-Finkel and Benayahu 2004, Ferse 2009, Fadli et al., 2012). Recently, the effects of artificial reefs on coral reef communities has also been studied on existing structures such as breakwaters, groynes, and jetties (Burt et al., 2009), which can yield important information for the creation of substratum for reef restoration.

Several materials have been tested since the onset of artificial reef creation, but the most favoured reef material is concrete, usually in the form of cubes, blocks and pipes (Baine 2001). Other materials used, in order of decreasing preference, are gabbro, granite, sandstone, and terra-cotta (Burt et al., 2009). Increasingly, engineered structures (e.g. EcoReefs, BioRock, ReefBalls) are designed with greater 3-dimensional complexity, in attempts to mimic coral reef habitats more closely (e.g. www.reefball.com). It is generally understood that the choice of material will influence the success of coral settlement, and hence the development of the entire benthic community (Burt et al., 2009). Additionally, the complexity of artificial reef structures will affect the settlement of benthic organisms (Thanner et al., 2006, Burt et al., 2009). Despite the large number of artificial reefs deployed worldwide for a variety of reasons, very few studies have monitored their development beyond the early stages of colonisation by benthic organisms (Hannes and Floyd 2009).

In the past decade, Mars Incorporated have developed a modular approach to restoring corals, particularly suitable to deploy on unstable substrate. The technique uses small, modular, open structures consisting of steel bar fabricated into a hexagonal structure resembling a spider (each spider covers 0.7m^2). The structure is then coated in a rough textured protective coating (consisting of resin and coarse sand) for the coral to adhere to. Corals of opportunity are tied to the 'spider' structure with cable ties, after which the spiders are deployed to degraded reefs. The units are tied together to form a 'web' and then staked into the rubble substrate, creating a robust structure able to withstand storms. Once staked down, the spider structures stabilise and help consolidate the rubble. Depending on the populations of herbivores present, some maintenance may be required in the first few months to remove macro algae. The organisation has installed more than 17 000 spiders over approximately three hectares of coral reefs in Indonesia and claims that spiders have been very effective. Early studies showed that coral cover was increased from ten percent to over 60 percent after only three years (Williams et al., 2018) in spite of several human and natural impacts reducing coral cover from time to time during that period. The technique has been employed since 2011 and has been used at larger scales in Indonesia where reefs are affected by dynamite fishing and cyanide fishing (Williams et al., 2018). The structures used in this process are presumably incorporated into the reef structure over time, however the longer-term consequences and stability of this modular system remain to be evaluated.

Realistically, the success with which an artificial reef functions as a complete ecosystem will require long-term comparisons between artificial reefs and natural reefs subject to the same environmental conditions (Perkol-Finkel and Benayahu 2004, Hannes and Floyd 2009). A study that compared artificial reefs and nearby natural reefs for five years found that at the end of this period, the similarity of the scleractinian and octocoral community composition was 70 percent and 63 percent, respectively (Hannes and Floyd 2009). Another study that monitored the development of coral reef communities on two types of artificial reefs and compared them with nearby natural reefs found that it took four years for the benthic and fish assemblages on the artificial reefs to stabilise, highlighting the need for such long-term studies to assess the success of this method of habitat creation (Thanner et al., 2006). Blakeway et al., (2013) found that colonisation of an artificial reef in turbid waters by natural recruitment resulted in significant numbers of recruits but that after six years the cover on the artificial reef (2.4 percent) was approximately a tenth of that on co-occurring natural reefs. The process of colonisation may therefore be a slow process on turbid inshore reefs.

4.5.12 Substratum stabilisation

The direct physical restoration of damaged substratum mostly involves stabilising rubble over an area that has been affected by storms or ship groundings. The rationale is that corals settling onto a damaged reef cannot successfully recruit to lose substratum, as survival rates are low (Lindahl, 2003). While substratum stabilisation has been used relatively often in US territorial waters, funded by insurance claims following ship-strikes, there is a paucity of published literature that clearly describes methods and techniques (four percent of case studies in this review). The most common method is to install mesh or netting over the rubble to prevent further movement. This is generally a precursor to the transplantation of corals onto the damaged area (Lindahl, 2003) and / or the additional deployment of artificial structures. Other methods include metal spikes driven into loose substratum (Fox et al., 2005), rock piles on unstable degraded reef areas (Fox et al., 2005), and open cement structures placed to contain loose substratum (Hudson and Diaz 1988, Clark and Edwards 1995).

Only four published studies and one unpublished report describing substratum stabilisation were available, preventing analyses of general trends or outcomes. Substratum stabilisation was mentioned as a method of preparing for coral transplantation in several studies, but the transplantation was the focus of the study, and the effects of stabilisation were not assessed. One study that evaluated a range of different stabilisation mechanisms found that hard substrata were more successful in attracting coral recruits than soft structures such as nets, and that the recruits also survived better on hard substratum (Fox et al., 2015).

In addition to stabilising substrates, the use of artificial structures may avoid issues with unstable substrates. For example, the six-legged artificial reef structures (‘spiders’) described above circumvent the problem of unstable and soft substrates, by elevating transplanted corals above the substrate however they probably should be deployed with care to avoid high energy environments that create rubble in the first place and may dislodge structures as well as re-mobilising rubble.

4.5.13 Substratum enhancement with electricity

In five percent of substratum enhancement studies, electricity was used to attempt to encourage faster growth and higher survival of coral transplants. This concept and technique were pioneered by Wolf Hilbertz in the 1970s and developed into a commercial product (Biorock). The aim of the technique is to mimic the chemical and physical properties of reef limestone, by encouraging the precipitation of calcium and magnesium on artificial substrates (Goreau 1996). A direct electrical current is established between electrodes, and calcium carbonate and magnesium hydroxide precipitates at the cathode, while oxygen and chlorine are produced at the anode (Hilbertz 1976). The purpose of this mineral accretion is to potentially increase calcification of coral polyps, and therefore boost colony growth and resilience to stressors. The authors and patent holders (Hilbertz 1976, Goreau and Hilbertz 1996) have published books and reports on the apparent effectiveness of this technique, suggesting that it increases the growth rates, survival, stress resistance, and physiology of corals (Goreau 2014).

Sabater and Yap (2002) described increased growth and attachment in *P. cylindrica* fragments when connected to a setup similar to that described by Goreau and Hilbertz (1996). A range of other studies have described increased survival of fragments on mineral accretion frames (Van Treeck and Schuhmacher 1997; Schuhmacher et al., 2000; Sabater and Yap 2002, 2004; Eisinger 2005; Eisinger et al., 2009). However, multiple experiments have failed to describe similar positive effects of exposing coral fragments to an electrical field. For example, Romatzki (2014) found that *A. pulchra* and *A. yongei* coral fragments exposed to similar strength electrical currents as those described by previous researchers grew slower than control colonies. Similarly, Borell (2010) described negative effects on growth of one species of coral (*A. yongei*) but positive effects on another (*A. pulchra*) growing on a cathode, suggesting that results may vary even between congeneric coral species. The disagreement between studies prohibits clear conclusions about the mineral accretions' method. However, at least some studies indicate that coral fragments exposed to an electrical field will attach more rapidly to frames and structures (e.g. Sabater and Yap 2002, Borell 2010, Romatzki 2014, Goreau 2014; Van Treeck and Schuhmacher 1997; Schuhmacher et al., 2000; Sabater and Yap 2002, 2004; Eisinger 2005; Eisinger et al., 2009), which could increase survival to some degree.

Recommendations for substratum interventions

1. Concrete appears to be the most favourable material for artificial reefs, but consider using materials with a smaller carbon footprint (see discussion)
2. Place artificial reefs within dispersal distance of natural reefs
3. Structural complexity may be more important than the material artificial reefs are made from
4. Substratum interventions tend to work best when deployed in combination with other interventions; i.e. stabilising loose substratum, adding artificial reefs and attaching coral colonies or fragments
5. Monitoring needs to occur for a reasonable period (at least four years) until significant changes can be detected
6. Modular structures can be used for outplanting corals of opportunity or nursery-grown corals to large reef areas

4.5.14 Larval enhancement

The process of larval enhancement (also known as 'larval propagation', 'sexual propagation' or 'larval re-seeding') aims at increasing the rates of larval production and settlement leading to increased recruitment success. These methods are designed to overcome the natural demographic bottlenecks where coral fertilisation rates may be limited on reefs with low coral cover and asynchronous spawning, and planktonic development of embryos and larvae may result in a high proportion of coral larvae being swept away from reefs and therefore failing to settle or recruit (Harrison and Wallace 1990; Richmond 1993; Jones et al., 2009). As such, the methods are mainly aimed at enhancing larval settlement and recruitment at sites that have experienced recruitment failure, or drastically reduced recruitment levels. Sexual reproduction methods have advantages over asexual propagation methods because they increase genetic diversity among restored coral populations, thereby enabling increase rates of adaptation and improved resilience (van Oppen et al., 2017). Larval enhancement also has potential for increased scales of restoration on degraded reefs.

There are two main types of larval enhancement strategies to enhance recruitment. First, interventions can collect or rear embryos and larvae to increase settlement on artificial or engineered structures that are later placed on reefs. For example, work in Japan showed that culture of spawned coral gametes can provide access to large numbers of embryos and larvae (Omori 2005), some of which can be induced to settle on settlement surfaces that can be outplanted onto reefs (Omori et al., 2008, Iwao et al., 2010). Settlement of *ex situ* cultured larvae and transplantation of settled juvenile corals onto reef areas has also been successfully trialled in the Philippines, with increased survival rates evident among larger and older age classes of recruits when outplanted onto reefs (Raymundo and Maypa 2004; Villanueva et al., 2012; Baria et al., 2012; Guest et al., 2014). Recently, Chamberland et al., (2017) 'seeded' concrete tetrapods, with Caribbean *Favia fragum* larvae that had been fertilised and reared *ex situ.* The 'seeding units' were scattered onto a degraded reef area, after a four-week juvenile coral rearing period. Approximately ten percent of settled larvae survived, and 56 percent of seeding units harboured at least one *F. fragum* individual after one year. The authors concluded that the main advantage of this method over others is the speed of outplanting compared to methods which attach coral fragments individually.

Another larval-based method was used by Edwards et al., (2015) in Palau who reared *A. digitifera* larvae *ex situ* and then allowed larvae to settle on 1.2 x 0.9 m concrete pallet balls and settlement tiles while enclosed in a tent. Although the authors recorded high initial larval settlement rates on tiles, there were no significant differences in the mean number of coral recruits surviving on larval enhanced versus control pallet balls after 13 months. The authors speculated that this may be due to high post-settlement mortality combined with high rates of natural coral recruitment in the healthy reef area used for the study (Edwards et al., 2015).

In a second larval enhancement technique, coral gametes are collected during spawning, embryos and larvae are reared in holding tanks or on the reef, and then larvae are released directly onto the reef in enclosures that retain them during the settlement period (Heyward et al., 2002, Suzuki et al., 2012; dela Cruz and Harrison 2017). Heyward et al., (2002) collected spawned acroporid gametes and reared embryos and larvae in small 1.8 m floating ponds for six days at Ningaloo Reef, Western Australia. When larvae were competent to settle, they were funnelled into a floorless mesh tent, anchored to the reef substrata, and pumped onto an enclosed 1.8 x 1m reef area for 20 minutes, and then the tent was moved to an adjacent area for 12 hours. Although the numbers of larvae pumped onto the experimental reef plots and initial settlement rates on tiles were not quantified, recruitment rates on tiles in the larval enhancement plots monitored after six weeks were up to 100-fold higher compared to control sites. However, this study was not conducted with a coral restoration objective, so no ongoing monitoring was undertaken, and therefore the longer-term juvenile survival and restoration outcomes are not known.

More recently, longer-term replicated larval enhancement and recruitment trials have been completed successfully on highly degraded reef areas in Northern Luzon, Philippines (dela Cruz and Harrison 2017, Harrison et al., 2016, and unpubl. data). dela Cruz and Harrison (2017) demonstrated that mass larval settlement on degraded reef areas (4 x 6 m) can significantly enhance recruitment and re-establish a breeding population of *A. tenuis* colonies after three years. Spawned gametes were collected from thirty gravid colonies, and embryos and larvae were reared in *ex situ* tanks for four days, then they were transferred onto replicate reef plots and retained in low-cost fine mesh larval enclosures for five days. High rates of larval settlement were recorded on settlement tiles in the larval enhancement plots whereas none settled on control tiles. As expected for broadcast spawning marine invertebrates, mortality rates of settled corals were highest in the first five months after settlement, then survivorship stabilised as recruits reached visible size (dela Cruz and Harrison 2017). Rapid growth lead to early onset of sexual reproduction in many colonies and these corals are now dominating the larval enhancement plots, and have spawned annually over the past three years, thereby contributing to larval production on these reefs (de la Cuz and Harrison, unpubl. data). Production costs of the three-year-old adult corals were <US$21. Additional larval enhancement reef trials with larvae from other *Acropora* species and brain corals have resulted in similar patterns of settlement, recruitment and growth on other reef areas in the Philippines (Harrison et al., unpubl. data). Larger scale larval enhancement trials using 100 m^2 reef patches enclosed in floating mesh curtains resulted in successful larval settlement on reef patches in the southern Great Barrier Reef during 2017 and long-term monitoring of recruitment, survival and growth is underway (Harrison et al., unpubl. data).

4.6 Relative costs

The median cost of coral restoration from all observations with reported costings in the dataset (n = 64) was 471,621 US$/ha (at base year 2010) ± 18,066,547 (median ± standard deviation). Out of a total of 338 observations, 18.9 percent reported on cost and 16.0 percent described what restoration components these costs included. From all observations reporting on cost, only 37.5 percent reported on both capital (planning, purchasing, land acquisition, construction, financing) and operating cost (maintenance, monitoring, and equipment repair/replacement) highlighting that the cost values reported here should be considered as relative cost only.

Regarding the eight restoration techniques covered in this report, the median cost for substrate addition - artificial reef were highest (n = 15) with 3,911,240 ± 36,051,696 US$/ha (Table 2). The cheapest restoration technique reported was the nursery phase of the coral gardening approach with median cost of 5,616 ± 22,124 (US$/ha). Presumably this difference is due to high level of technical and logistical requirements driving prices up for artificial reefs, while coral nurseries are relatively low-tech and often utilise volunteer labour.

Table 2: Range of relative cost (US$/ha at 2010) for coral reef restoration projects from published and grey literature with sample size of observations (n), median ± standard deviation (SD), minimum and maximum

Restoration technique		Restoration cost (2010 US$/ha)		
	n	Median (± SD)	Minimum	Maximum
Coral gardening	3	351,661 (± 136,601)	130,000	379,139
Coral gardening - nursery phase	5	5,616 (± 22,124)	2,808	55,071
Coral gardening - Transplantation phase	2	761,864 (± 1,033,831)	30,835	1,492,893
Direct transplantation	21	73,893 (± 867,877)	4,438	3,680,396
Enhancing artificial substrates with an electrical field	0			
Larval enhancement	6	523,308 (± 1,878,862)	6,262	4,333,826
Substrate addition - artificial reef	15	3,911,240 (± 36,051,696)	14,076	143,000,000
Substrate stabilisation	8	467,652 (± 9,015,702)	91,052	26,100,000

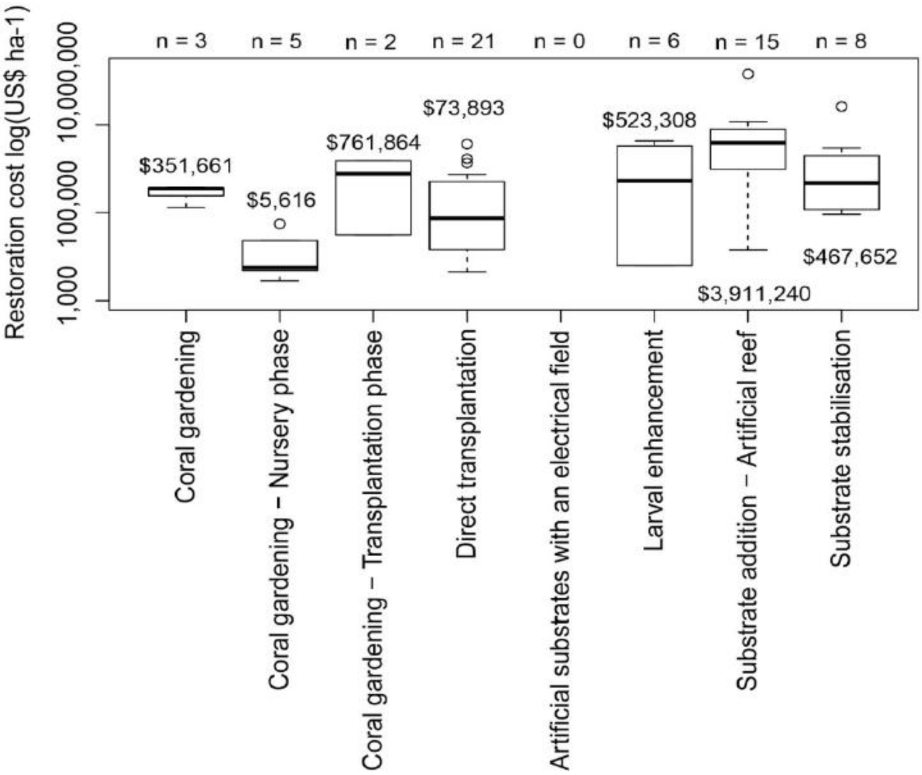


Figure 6: Range of relative costs reported for coral reef restoration projects categorised by restoration methods represented as box and whisker plots with sample size (n), minimum, quartiles, median, maximum, and outliers. Restoration cost (US$/ha at 2010) are displayed on a logarithmic scale.

Recommendations for estimating costs of restoration

Accurate reporting on the total costs of coral reef restoration is inconsistent or omitted entirely within the published and grey literature. For costs that have been reported, there is a large variation within and between intervention type, location, and study species, thus the costs presented here are to be interpreted as relative values only. A survey needs to be designed and used to audit coral reef restoration practitioners to investigate accuracy and bias within restoration costs reported in published literature and elicit total restoration costs in a standardised way. This work is envisioned in collaboration with the University of Queensland.

5. SUMMARY OF FINDINGS

The field of coral restoration has evolved in the past decades, and this review documents the successes and failures reported by scientists, managers and practitioners. Coral restoration has shared some of the common ‘growing pains’ associated with ecological restoration more broadly. For example, Lake (2001) highlighted the five obstacles to scaling ecological restoration of freshwater lakes and rivers in Australia, and there is an almost complete overlap with issues described throughout this report. These include (1) the reluctance of resource managers to undertake large and long term restoration projects; (2) the poor design of many restoration projects; (3) the lack of adequate and ongoing monitoring of projects; (4) the lack of reporting on the progress and outcomes of projects; and (5) the common challenges associated with increasing spatial and temporal scale in restoration projects (paraphrased from Lake 2001). However, recent advances illustrate how ecological restoration is increasingly informed by ecology, and ecologists are increasingly learning the lessons acquired by practitioners. Australia has a unique opportunity to learn from decades of coral restoration experience overseas, to avoid common pitfalls, and to make informed decisions for coral restoration in Australian waters. This review has summarised five decades of coral restoration projects and research, and to our knowledge is the most comprehensive review of existing coral restoration techniques. The accompanying interactive database provides an evidence-based resource for researchers, managers and practitioners to draw from and build on.

All methods described in this review have documented successful coral growth, and relatively high levels of survival. We direct the reader to each specific section for descriptions of the potential application and limitations associated with each method. There are however, two primary challenges, which, if resolved, present opportunities for improved project success going forward. These are (1) improvements in clear objective setting and monitoring of outcomes, and (2) issues with scaling up projects.

Objectives and monitoring

Objectives for coral restoration usually have a broad ecological scope that aligns with principles of reef resilience (Hein et al., 2017). For example, the aims of coral restoration usually focus on accelerating reef recovery (Rinkevich 2005, Garrison and Ward 2008, Ferse 2010), re-establishing a functioning reef ecosystem (Edwards and Gomez 2007, Hunt and Sharp 2014), or mitigating population declines and endangered species management (e.g. *Acropora* recovery plan, Johnson et al., 2011). However, there is a clear mismatch between stated objectives and what is actually measured during monitoring of projects. For example, of projects with an ecological objective (n=129), 45 percent did not measure any relevant metrics to evaluate project success relating to that objective. This mismatch not only prevents scientific evaluation of project outcomes, but also carries the risk of reducing public support for coral restoration in general, by building expectation and then not providing evidence to evaluate success or failure.

Coral restoration projects targeted at reef recovery, should have the re-establishment of breeding populations as a fundamental aim. Once these population are established, they should not need ongoing interventions, and can enhance natural larval production and recruitment processes. This objective is a key missing component in almost all coral reef restoration projects, and evidence for this is rarely monitored. In this review, for example, we found that outcomes of coral restoration projects are largely monitored with biological metrics of corals. Coral growth and survival were cited as outcomes in approximately 60 percent of the published literature and constituted most outcome metrics in the responses to survey questions. Other popular biological metrics used as outcomes included the condition and self-attachment of fragments. While these metrics are important for assessing the feasibility of coral restoration and for refining techniques, ecological monitoring is required to assess the ecological outcomes of restoration efforts and whether or not the initial aims and objectives are met. Ecological metrics would also enable better assessments of the ecological processes occurring at restored areas (e.g. fish and invertebrate colonisation and increased structural complexity; Shaver and Silliman 2017, Ladd et al., 2018), and thus inform adaptive management of restoration efforts in the longer term.

By necessity, we have used survival of corals to compare studies in this report. However, it is becoming increasingly clear that this is not always a relevant metric to assess the success of restoration projects. First, published survival data is likely to be biased towards higher survival. There is limited incentive to publish or report failed experiments for researchers, practitioners and managers alike, presumably due to a fear of discouraging funders and the general public. Second, shorter monitoring times may artificially inflate survival further by not reflecting the true long-term fate of restored corals. Finally, as the field moves towards sexual propagation as a source of coral propagules, which naturally have low survivorship, it becomes an almost meaningless metric, further highlighting the usefulness of a more holistic suite of ecological monitoring tools rather than biological metrics.

Socio-cultural and economic outcomes should also be assessed as part of coral restoration objectives, such as whether or not coral restoration can promote alternative livelihoods (Edwards 2010, Young et al., 2012), or promote local conservation stewardship (Fisk and Job 2008, Edwards 2010). Not only are socio-economic and governance dynamics of the region and stakeholders involved likely to influence restoration outcomes but benefits of coral restoration at the socio-cultural and economic scales are likely to widen the scale of coral restoration benefits (Ammar 2009, Hein et al., 2017). Finally, practitioners should involve local communities as much as possible throughout the project to build capacity and stewardship to ensure that the restored coral reef can flourish after the interventions.

While the need to broaden *what* we monitor is clear, there is also a need to standardise *how* we monitor the outcomes of restoration projects. A lack of standardisation in how to record mortality, survival, and growth makes these metrics difficult to compare between studies. Even the basic unit of the organism in question lacks standardisation. Coral gardening projects often refer to “fragments” without quantifying what the size (average, min, or max) of these units are (Young et al., 2012). The same lack of precision applies to the term “coral colony”. This absence of specificity in defining fragments or colonies in terms of their size hampers our ability to decipher survivorship. Survivorship of a small (e.g. 10 cm^2) fragment or colony, is expected to be far less than that of a 40 cm^2 unit, and yet, in most reports, percent survival of outplants is not clearly tied to size of outplants. The metric that best illustrates the lack of standardised reporting is growth of corals, where a multitude of creative metrics have been used - including linear extension (e.g. O’Neil et al., 2015), height (e.g. Tagliafico et al., 2018), ‘ecological volume’ (e.g. Shaish et al., 2010, Mbije et al., 2013, Toh et al., 2014), branch width (e.g. Bowden-Kerby 2008, Griffin et al., 2012), number of branches (e.g. Yap 1984, Soong and Cheng 2003, Chilcoat 2004), basal width (e.g. Becker and Mueller 2001) as well as combinations of width, length, height and partial mortality, maximum colony diameter, number of branches, and virtually every other dimension one could think of. The metrics used have such little overlap that comparisons of actual growth among studies are virtually impossible. Similarly, mortality or survival are often reported without explaining what specifically is being measured. Most studies use a simple binary scoring system (i.e. ‘live’, ‘dead’), while some base survivorship on arbitrary thresholds (e.g. >50 percent partial mortality = dead), and others quantify partial mortality of individual coral fragments (e.g. Lindahl

2003, Kenny et al., 2012). Established methodological approaches to coral demography (e.g. Babcock 1991) should be used which will not only help standardise metrics but also facilitate modelling approaches to assessing the utility of the wide range of possible restoration approaches that exist.

While the lack of standardisation is problematic, it is an understandable consequence of numerous groups and practitioners operating in isolation. This issue is now widely recognised, and groups like the Reef Resilience Network and the Coral Restoration Consortium are developing and sharing best practice guidelines for coral restoration worldwide. We suggest practitioners adopt the following guidelines for reporting the outcomes of restoration projects: 1) Be explicit when describing how your metrics were calculated, and what the calculations are based on, 2) avoid inventing new metrics for simple demographic parameters, unless the new metric is a radical improvement on existing methods, and 3) avoid complex equations in favour of simple calculations based on raw data.

The ideal monitoring program would be comprehensive and holistic, including ecological, social and economic metrics. However, all projects are limited in terms of funding and logistical capacity and will most likely be unable to monitor the complete range of metrics. Ultimately, practitioners should adopt a monitoring program that is clearly linked to the stated objectives of the project. Further, the use of proxies and indicator metrics may help reduce costs and time required. For example, coral cover and complexity may provide a suitable indicator of restored habitat value for other reef species. In addition, the number or proportion of breeding corals may be a more useful measure of restoration success than survival and growth of outplants. Proxies and indicators need to be thoroughly tested and evaluated before widespread use.

Some projects have stated ecological objectives, when in reality they are primarily aimed at social (e.g. local capacity building, stewardship) and or economic (e.g. job creation, edutourism) objectives. These are relevant and important, and there is no problem with projects being primarily focused on socio-economic objectives. However, much of the public and scientific distrust of coral restoration could well stem from a mismatch between what is publicly portrayed as the objectives and goals of projects, and the executed reality. We encourage practitioners to clearly state realistic objectives and to design projects and monitoring programs relevant to those objectives, to avoid artificially inflating expectations. Media agencies will often further inflate stated objectives (e.g. “local restoration project will save Great Barrier Reef/Caribbean reefs”) so caution should be used when describing projects to media and the public.

Scaling up and future consideration

The future of coral reef restoration requires substantial spatial scaling up of projects, if restoration is to meet future challenges to coral reefs. Coral restoration is a rapidly changing field, and future projects may be radically different from techniques described in this review. While past projects have been relatively small, isolated and disconnected, future projects will have to be industrial in scale as well as highly coordinated and connected. Of the techniques described in this review, few have demonstrated potential to be scaled up beyond a hectare of restored coral reef. The most scalable methods appear to be techniques that use sexually derived propagules as a source of corals. A comprehensive review of novel techniques and ideas are explored in sub-project SI.1c *Identification of intervention strategies*, of the Reef Restoration and Adaptation Program, so will not be detailed further here. However, large-scale coral restoration projects often rely on techniques developed throughout the relatively small-scale methods explored in this review, particularly during deployment.

A current challenge associated with selecting the most appropriate method for restoration projects, is a paucity of information on the true costs of projects. Cost-benefit analyses of different methods could be based on final numbers of breeding corals at restored sites, and should include the true cost of deployment, even if volunteer labour is used, to facilitate accurate comparison between methods. The cost data used in this report is based on currently available estimates from published literature and provides a useful starting point to estimate costs. However, ongoing work endeavours to improve these estimates.

Environmental considerations

Throughout the efforts of compiling this review, it has become evident that while coral restoration projects aim to solve the problem of habitat loss on coral reefs, they may inadvertently contribute to the very problem they are trying to mitigate. For example, most artificial reefs and structures are made from concrete, and ten percent of studies which attach corals to the substrate used cement and concrete. The production of cement is responsible for five to seven percent of global carbon emissions, mainly due to CO_2 emissions during the calcination process of limestone, from combustion of fuels in the kiln, as well as from power generation (Worrell et al., 2001). While the proportional contribution of coral restoration to the overall carbon footprint of concrete is negligible, one cannot fail to see the irony in using a technique for restoration which directly contributes to climate change. Further, a substantial number of projects (~60 percent) use plastics to attach coral fragments to the substrate, primarily in the form of epoxy putty or cable ties. While there are marine grade versions of both materials, they are likely to break down in the shallow, warm and high-UV environments of corals reefs, potentially contributing to the growing problem of microplastics in the marine environment. Finally, there is growing anecdotal evidence of failed coral restoration projects, where frames and structures are left as garbage littering the reefs they were supposed to help. As the field of coral restoration grows and spreads to more coral reefs around the world, we urge practitioners to seek out biodegradable alternatives, source local materials, employ local people and to avoid contributing to the problems facing coral reefs in the Anthropocene.

This review presents the most comprehensive summary of active coral restoration methods used to date, and combined with the online database, provides a resource for scientists, practitioners and managers. We have described coral restoration projects throughout the tropics, with a surprising diversity of coral species and morphologies used. While few projects have reported on ecological success, there is substantial evidence of our abilities to grow corals at smaller scales. Overall, all the main techniques report similar average survival and growth of corals, so decisions on what techniques to use should be based on local conditions, cost, availability of materials and appropriateness based on stated objectives. There are ongoing refinements of techniques, with a growing focus on scaling up both spatially and temporally. Coral restoration methods and projects could be a component of resilience-based management, along with water quality and fisheries management. However, one of the biggest drivers of coral reef decline is climate change. While many projects address this by propagating presumed heat-tolerant corals (i.e. those that survived recent bleaching events), ultimately coral restoration is not a replacement for meaningful action on climate change.

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APPENDIX A – RRAP DOCUMENT MAP

Reef Restoration and Adaptation Program

Reef Restoration and Adaptation Program, a partnership

Reef Restoration and Adaptation Program

GBRrestoration.org

Ian McLeod
Ian.mcleod@jcu.edu.au

Reef Restoration and Adaptation Program, a partnership:

Australian Government
Australian Institute of Marine Science
Australian Government Great Barrier Reef Marine Park Authority
Great Barrier Reef Foundation
CSIRO
QUT
James Cook University Australia
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