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Coral Restoration: Comparisons in Space, Time, Impacts, and Costs

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CORAL RESTORATION: COMPARISONS IN SPACE, TIME, IMPACTS, AND COSTS

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ABSTRACT

Seventy-five percent of coral reefs globally face crisis due to anthropogenic disturbances, prompting heightened global coral restoration initiatives to preserve these vital ecosystems. Various regions employ diverse active coral restoration methodologies, including coral gardening, transplantation, micro-fragmentation, artificial reefs, and sexual propagation. Of these methods, coral gardening stands out as one of the most common and highly successful methods, alongside widespread transplantation practices. Restoration efforts predominantly focus on acroporids due to their relatively rapid growth and asexual fragmentation; however, a diverse range of coral species, including large, slow-growing varieties, is also employed in these endeavors. Costs vary significantly, ranging from $10,000 to $50,000,000 USD per hectare, contingent on restoration methods and locations. Coral restoration does not address the whole problem though, achieving optimal coral survivorship post-restoration involves integrating ecological processes, coral density, and arrangement. In a Bonaire study, I utilized established coral reef monitoring sites, creating 100 m^2 quadrats to assess *A. cervicornis* and *A. palmata* population density and vigor. Among 11 long-term monitoring sites, three were active restoration sites (1,796 acroporid coral outplants), three were adjacent control reefs, and five were regional control reefs. No acroporids were found at 10 m depth, but at 5 m depth, 13 acroporids were recorded at four survey sites, with five from outplant sites. The two sites with the highest acroporid densities were Calabas (restoration site) and Karpata (control site). The efficacy of coral restoration remains inconclusive based on this study of Bonaire's long-term monitored reefs.

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ABSTRACT

Coral reefs, vital ecosystems supporting 30% of global marine biodiversity, face unprecedented degradation from climate change, pollution, coral disease, bleaching events, overfishing, and coastal development. In response, a surge in global coral restoration initiatives strives to meet the urgent challenge of preserving these ecosystems and their indispensable services. This review aims to examine diverse coral reef restoration methods employed worldwide, delving into their costs and successes. Examined methodologies encompass coral gardening, transplantation, micro-fragmentation, artificial reefs, and sexual propagation. Notably, coral gardening emerges as a widely adopted and successful method, while transplantation also enjoys widespread use. The restoration landscape encompasses a diverse array of species, with acroporids being the most common due to their rapid growth and asexual fragmentation. Achieving optimal post-restoration coral survivorship mandates the integration of ecological processes, coral density, and arrangement into these restoration methods. However, significant challenges, including herbivory, corallivory, nutrient deficiencies, and diseases, must also be systematically addressed to enhance the overall effectiveness of coral restoration initiatives. Despite criticisms, the field of coral restoration continually evolves, refining methods and enhancing cost-effectiveness to address the coral reef crisis on a global scale.

INTRODUCTION

Coral reefs are essential, intricate ecosystems that support a diverse array of marine species. Despite only occupying a mere 0.17% of the world's oceans, these tropical ecosystems harbor an astonishing 30% of global marine biodiversity (Omori, 2019; Sheppard, 2018; Ladd et al., 2016). Estimates of the multitude of plant and animal species within reefs vary widely, ranging from 600,000 to over 9 million species worldwide (Sheppard, 2018). Coral Reefs are primarily found in tropical regions due to their dependence on light and preference for shallow waters (Sheppard, 2018; Cavasos, 2019). There are three classic reef types: fringing reefs, which develop along the edges of islands; barrier reefs, which form when an island subsides while reef growth keeps pace with sea level; and atolls, which encircle entirely subsided islands and surround a central lagoon (Sheppard, 2018).

Coral reefs function as a critical habitat for a variety of marine life while concurrently providing a concentrated food source for both marine organisms and human populations (Omori, 2019; Sheppard, 2018). Coral reefs contribute substantially to the nutritional needs of millions of people in tropical countries, supplying essential protein (Sheppard, 2018). Approximately 10-13% of the global fisheries catch is derived from coral reef fisheries, which supply a variety of vertebrate and invertebrate species to millions of people (Cavasos, 2019). Beyond food production, coral reefs deliver a spectrum of other invaluable "ecosystem services." They function as secure nurseries for fish to grow to market size,act as natural breakwaters for coastal protection, and serve as a magnet for tourism (Omori, 2019). Reefs-based tourism attracts a global audience including boaters, sport fishermen, snorkelers, and scuba divers, significantly contributing to the total revenue of foreign earnings of numerous countries (Omari, 2019; Sheppard, 2018). In certain nations, such as Seychelles and the Maldives, reef-based tourism constitutes one to two-thirds of foreign earnings (Sheppard, 2018). The Caribbean Sea's reefs are valued at an estimated \$3.1-\$4.6 billion annually whereas the economic value attributed to all coral reefs reaches around \$30 billion annually (Wagner, 2022; Omari, 2019).

Nevertheless, the invaluable services provided by coral reefs are under threat due to a global decline in corals (Ladd, 2016). These essential ecosystems find themselves in crisis due to an array of unprecedented changes in the global environment, including escalating CO_2 levels, pollution, declining water quality, over-exploitation, habitat destruction, the encroachment of invasive species, and the overarching influence of climate change (Omori, 2019; Zhang et al., 2023; Cavasos, 2019). These anthropogenic disturbances increase thermal stress events and coral bleaching, causing 75% of the world's reefs to be classified as threatened (Caruso et al., 2021; Omori, 2019).

Coral reefs are particularly susceptible to climate change due to their limited thermal tolerance range, compounding by existing near their upper thermal limits in tropical regions (Jones & Berkelmans, 2010; Zhang et al., 2023). The repercussions of substantial ocean warming include both immediate and long-term degradation of reef ecosystems, leading to substantial losses of coral cover on a global scale (Caruso et al., 2021). Over the past three decades, coral reefs have undergone rapid deterioration, with global coral cover declining by about 20% (Zhang et al., 2023). Indo-Pacific reefs have witnessed a nearly 50% reduction in coral abundance over the past four decades, while Caribbean reefs have experienced a staggering 80% loss (Ladd et al., 2016; Ladd et al., 2020).

In 2006, A*cropora cervicornis* (staghorn coral) and *Acropora palmata* (elkhorn coral) became the first marine invertebrates designated as 'threatened' under the US Endangered Species Act (Cavasos, 2019). Subsequently, an additional 20 coral species, 15 in the Indo-Pacific and 5 in the Caribbean, have been added to this list, underscoring the escalating nature of this crisis (Cavasos, 2019). The mounting evidence suggesting the potential global extinction of corals within a few decades poses a critical threat to the natural diversity, functionality, and ecosystem services crucial for food security, coastal protection, and biodiversity in tropical reefs. (Cavasos, 2019; Caruso et al. 2021, Ladd 2016).

Worldwide coral reef degradation has reached a critical point, prompting concern that coral reefs may not recover naturally without human intervention (Young et al., 2012). Until recently, marine conservation efforts have favored passive habitat protection reliant on natural recovery mechanisms, but the unprecedented scale of reef decline has necessitated a shift in management priorities (Bostrom-Einarsson et al., 2020, Vardi et al. 2021). In the past two decades, there has been a growing emphasis and rapid expansion of coral reef restoration, especially in the Caribbean, where thousands of coral nurseries now exist (Knoester et al., 2023; Wagner, 2022). Coral restoration represents an active approach to enhance coral populations, combat local reef degradation, and magnify existing conservation efforts for greater reef resilience against global stressors (Knoester et al., 2023). While early coral restoration initiatives concentrated on transplanting corals from healthy to disturbed areas, contemporary restoration practices have evolved significantly, incorporating various methods (Ladd, 2020). These methods encompass both land-based (ex-situ) and ocean nurseries (in-situ), primarily involving the outplanting of nursery-raised corals (Wagner, 2022; Ladd, 2020). Coral restoration strategies can also include asexual or sexual reproduction, transplanting coral fragments (live portions separated from the colony), or making artificial reefs (Wagner, 2022; Omori, 2019).

Coral restoration stands as a complex challenge that has generated skepticism within the scientific community regarding its efficacy. Despite widespread popularity and significant investment of time and resources, numerous restoration initiatives face criticism for their perceived oversight of the fundamental ecology of coral reefs (Ladd, 2020). Skeptics argue that current capabilities of coral restoration is inadequate for addressing the scale at which reefs are degrading, that reef ecosystems are too complex, and that the corals do not survive the transplantation and relocation well (Hein et al., 2017; Precht et al., 2005). Small-scale coral restoration efforts have been likened to "treating cancer with a band-aid." While it is acknowledged that such endeavors alone cannot fix the large-scale threats of climate change and ocean acidification, they can tackle local problems and raise awareness of the issues (Krumholz et al., 2010). Despite notable progress, significant knowledge gaps persist within coral restoration practices, particularly regarding the factors influencing restoration success, such as optimal density and arrangement for outplanting restored corals (Ladd, 2016). The lack of scientific evidence on outcomes and benefits, coupled with the absence of standardized measurable protocols for evaluating success indicators, remains a critical concern (Hein et al., 2017). Furthermore, the limited duration of most studies contributes to the challenge of active restoration, as published research documenting transplant survival for five years or more is rare in the scientific literature. The mean monitoring duration is under two years, with the majority of coral restoration projects monitoring for one year or less (Garrison & Ward, 2015; Hein et al., 2017). One year is insufficient for assessing the effectiveness of coral restoration, given the year-to-year variations in growth observed in scleractinian (Hein et al., 2017). Addressing these challenges is imperative for advancing understanding of coral restoration and ensuring success for reef ecosystems.

CORALS USED IN RESTORATION

Restoration most commonly uses fast-growing branching corals, such as *A. palmata* and *A. cervicornis* (Wagner, 2022). A substantial proportion of Caribbean-focused coral restoration studies, as indicated by Bostrom-Einarsson et al. (2020), predominantly utilized fast-growing branching acroporid corals, with 59% of case studies featuring these species. Furthermore, 72% of these studies emphasized the use of multiple coral species in their restoration projects (Bostrom-Einarsson et al., 2020). The primary coral species used in coral restoration include *A. cervicornis, Pocillopora damicornis, Stylophora pistillata, A. palmata,* and *Porites cylindrica* (Bostrom-Einarsson et al., 2020).

Acropora, a globally distributed genus and known for breaking branches off during storms, offers naturally occurring fragments that can be collected and re-stablized on the reef (Krumholz et al., 2010). These species of acroporids play a pivotal role in reef development, island formation, fisheries habitats, and coastal buffering in the Caribbean (Young et al., 2012). The fast growth rates, natural fragmentation for asexual reproduction, rapid wound healing, and elevated survivorship make *A. cervicornis* and *A. palmata* well-suited for restocking projects (Wagner, 2022; Cavasos, 2019). Fragments of faster-growing species, essential for habitat complexity and usable space for fish and invertebrates, exhibit detectable changes in growth rates within a few years; a 5 cm fragment of *A. cervicornis* can produce 5 to 10 times the biomass in less than one year (Krumholz et al., 2010; Page & Vaughan, 2014).

While *Porites porites* (finger coral) has been incorporated into restoration initiatives due to its rapid growth and natural fragmentation, it is less frequently used compared to *Acropora* (Wagner, 2022). Slow-growing, massive stony corals, such as *Orbicella, Montastraea, Diploria,* or *Siderastrea*, have often been overlooked in past reef-restoration activities, despite being significant reef builders resilient to climate change in the Indo-pacific and Caribbean (Page et al., 2018; Krumholz et al., 2010). Recently, there has been a gradual shift towards incorporating massive corals in restoration, exemplified by experiments in the Florida Keys involving *Orbicella 8aveolate* (Mountainous star coral) (Wagner, 2022). This experimentation assesses the success of fragmenting massive corals and explores the feasibility of "coral reskinning," where fully grown micro-fragments are planted to merge into larger colonies, establishing faster than several smaller colonies (Wagner 2022, Page & Vaughan 2014). Notably, massive corals exhibit higher resilience to high-temperature stress compared to *A. cervicornis*, demonstrating an ability to withstand local stressors; they have formed inshore old-growth reefs that receive higher anthropogenic stress, nutrients, and sedimentation compared to offshore locations (Page et al., 2018; Wagner et al., 2010). Although massive corals, like *O. faveolate,* are increasingly present in coral nurseries due to their heightened outplant survivorship and growth, their slow growth rate restricts their utilization in restoration (Wagner, 2022).

CORAL RESTORATION THROUGHOUT THE WORLD

Coral restoration is carried out in 52 countries, with most projects located in the USA, Philippines, Thailand, and Indonesia (Fig 1) (Bostrom-Einarsson et al., 2018). Coral restoration goals vary between geographic areas, programs, and projects (Goergen et al., 2020). Restoration efforts also use different species of coral depending on the region; *Acropora* spp. Are used in the Caribbean and Japan, *Pocillopora* spp. Is used in Costa Rica and the Pacific, *Madracis miriaster* in Curacao, *Montipota digiata* in the Philippines, and *Stylophora pistillata* in the Red Sea (Lizcano-Sandoval, 2018). While efforts in the USA or Australia are better described, there is a scarcity of documentation carried out by practitioners in the Caribbean and Eastern Tropical Pacific (Bayraktarov et al., 2020). The global challenges faced by coral reef ecosystems necessitate comprehensive and region-specific restoration strategies.

Figure 1: Location of coral restoration case studies used in Bostrom-Einarsson et al., 2018. *Note.* Reprinted from “Coral restoration in a changing world – A global synthesis of methods and techniques,” by Bostrom-Einarsson et al., 2018, *National Environmental Science Program*, p. 15.

<u>Caribbean</u>

In the Caribbean, the shallow-water coral cover has undergone a substantial decline, plummeting from approximately 50% to 10% over the past three to four decades (Meester et al., 2015). Historically, *A. cervicornis* played a pivotal role in constructing coral structures within the region's shallow water reefs, dominating intermediate depths (5-25 m) on fore reefs, while *A. palmata* thrived on the reef crest and shallowest depths (0-5m depth; Cavasos, 2019; Meester et al., 2015). However, both acroporid species have witnessed significant declines in regional abundance since the 1970s, primarily attributed to white band disease (Cavasos, 2019; Aronson & Precht, 2001; Ware et al., 2020).

Bonaire, renowned for having the best reeds in the Caribbean, has maintained a coral cover around 46% since 2003, surpassing other Eastern Caribbean reefs, which have not exceeded 35% coral cover (Meester et al., 2015; Steneck et al., 2019). Conversely, Florida's *A. cervicornis* populations, consisting mainly of small, scattered colonies less than 50 cm in diameter, have witnessed reductions leading to loss of habitat complexity and biodiversity (Ware et al., 2020). Some populations resembling past thickets of *A. cervicornis* exist in Honduras, the Dominican Republic, and Belize (Ware et al., 2020).

Orbicella 10aveolate also significantly contributes to reef construction in the Caribbean but, like acroporids, has experienced declines due to repeated bleaching and disease events (Zhang et al., 2023). Experimental initiatives in the Florida Keys have assessed the success of fragmenting massive corals and the feasibility of "coral reskinning," leading to its increased presence in coral nurseries (Page & Vaughan, 2014; Wagner, 2022). Despite previous findings suggesting that adult *O. 10aveolate* populations near the shore in the Florida Keys exhibit the ability to sustain high coral cover and recover faster from bleaching compared to offshore counterparts, it is still unclear whether the observed variation in thermal resistance among these populations is heritable (Zhang et al., 2023).

The decline in Caribbean coral cover has spurred restoration initiatives, including the annual transplantation of predominantly nursery-grown *A. cervicornis* onto reefs in southeast Florida and throughout the Caribbean (Cavasos, 2019). Coral gardening is a preferred technique in many Caribbean restoration programs, including Rescue A Reef, Coral Restoration Foundation, and Reef Renewal Bonaire (Cavasos, 2019; Wagner, 2022). Figure 2 illustrates coral nursery and restoration sites throughout the Caribbean (Goergen et al., 2020).

Research suggests that restocking *A. cervicornis* on depleted reefs can contribute to the long-term recovery of wild populations and their genetic diversity (Cavasos, 2019). Outplanted corals in Florida and the Caribbean report varying success rates, with some documented survival ranging from 63%-95%, while early projects monitoring small outplanted coral cohorts recorded survivorship between 0% and 89% (Wagner, 2022; Meester et al., 2015; Bostrom-Einarsson et al., 2020). Long-term survivorship data, although limited, highlight Puerto Rico as a standout in the Caribbean, boasting the oldest-known program with sustained success for over seven years (Wagner, 2022). Culebra, Puerto Rico, documented the survival of *A. cervicornis* corals that were outplanted in 2003 and remained alive in 2016 (Lirman & Schopmeyer, 2016). In comparison, the Dominican Republic reported *A. cervicornis* outplants surviving for more than 7 years, while Belize reported a 6-year survival period (Lirman & Schopmeyer, 2016).

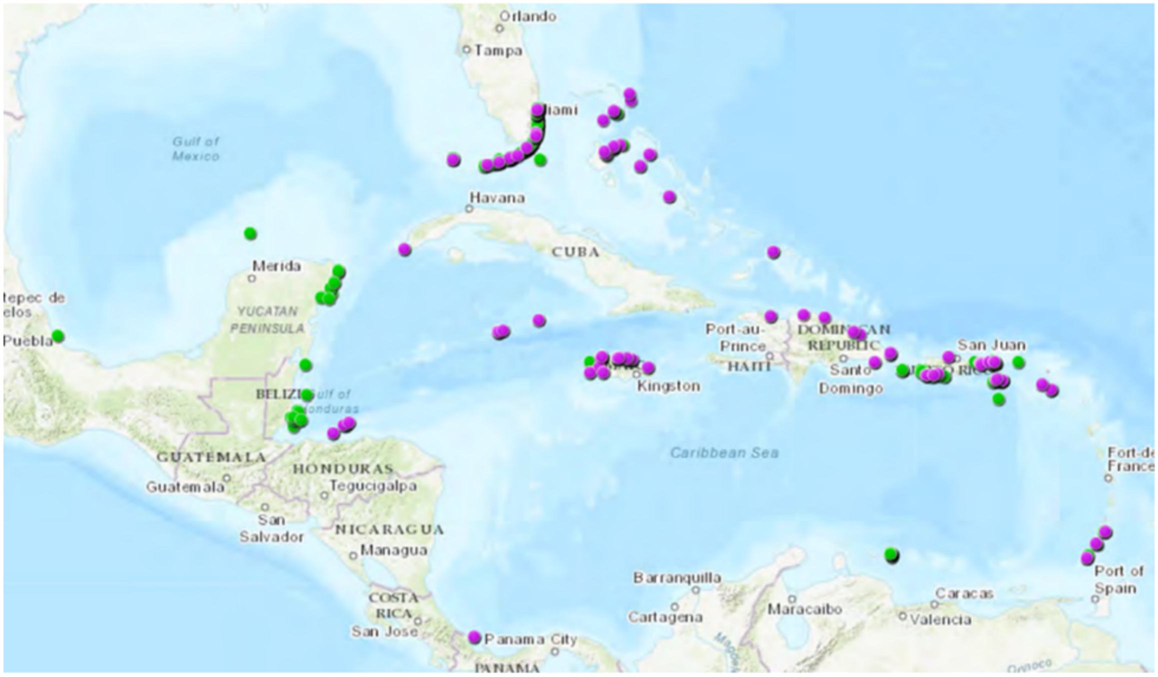

Figure 2: Map of Caribbean coral nursery sites (purple) and restoration sites (green). *Note*. Reprinted from "Coral reef restoration monitoring guide: Methods to evaluate restoration success from local to ecosystem scales," by Goergen et al., 2020, *NOAA Technical Memorandum*, p. 20.

Indo-Pacific

The coral reef ecosystems of the Eastern Tropical Pacific have deteriorated drastically in the last three decades (Ishida-Castañeda et al., 2020). The Indo-Pacific region , which constitutes approximately 75% of the world's coral reefs, serves as the epicenter for global marine diversity across various taxonomic groups (Bruno & Selig, 2007). Within the Tropical Eastern Pacific, coral reefs exhibit low species richness and high dominance of branching corals; the primary reef builder in the region is the genus *Pocillopora*, has faced considerable impact from El Nino events in previous years (Ishida-Castañeda et al., 2020). Despite variations in coral cover across subregions, a substantial overall decline has been observed since the 1960s in the Indo-Pacific (Bruno & Selig, 2007).

In Australia, where the largest coral reef is located, coral restoration research and adoption has been sporadic (McLeod et al., 2022). In the Cairns-Port Douglas region, The Coral Nurture Program has embraced coral gardening at six reefs, incorporating an innovative attachment device known as the Coralclip for the targeted outplanting of coral fragments and larval settlement devices (McLeod et al., 2022). In 2016, initial larval-based restoration techniques were initiated on Heron Island reef patches, funded by the Great Barrier Reef Foundation; this initiative focused on settling *Acropora* larvae, with higher larval-density treatments showing an increased settlement and recruitment density (McLeod et al., 2022). Additionally, a pilot capacity-building project, Boats4Corals, undertook the redesign of equipment and technology to enable boats and crews from the reef tourism industry to engage in larval restoration operations; this involved the utilization of modified larval culture pools directly on reefs, showcasing a novel approach to coral restoration (McLeod et al., 2022).

Turning attention to the coral reef ecosystems of the United States, approximately 85% are concentrated in the Hawaiian Archipelago, specifically in the Main Hawaiian Islands and Northwestern Hawaiian Islands, with an estimated economic value of $10 billion USD (Konh & Parry, 2019). Notable coral species in the Hawaiian Islands include *Montipora capitata, Montipora 13aveolate13, Montipora patula, Porites compressa, Porites lobata, Pocillopora meandrina, Pavona varians*, and *Pocillopora grandis/eydouxi* (Konh & Parry, 2019). The unique challenges confronted by Hawaiian reefs, including high wave action, slow coral growth, a substantial proportion of endemic species, and local environmental conditions such as elevated sea surface temperatures and ocean acidification, are further compounded by anthropogenic impacts like tourism, overfishing, terrestrial runoff, and coastal pollution (Forsman et al., 2018; Konh & Parry, 2019). These factors have hindered coral recovery and presented significant threats to corals in the region for the past six decades (Konh & Parry, 2019). While there are currently no ESA-listed corals in the main Hawaiian Islands, it is imperative to note that corals are protected under both state and federal law, underscoring the urgent need for conservation efforts (Forsman et al., 2018).

In the realm of reef restoration in Hawai‘i, reef restoration is gaining momentum, particularly focusing on transplantation, invasive algae removal, and the mitigation of anthropogenic threats (Forsman et al., 2018). In April 2018, a new reef rehabilitation platform constructed from Fiberglass Reinforced Plastics was designed and deployed off the south shore of O’ahu, Hawai’i (Konh & Parry, 2019). This platform adopts a gardening concept to establish a conducive nursery environment for non-fragmented corals, differentiating itself from fragmentation nurseries in other regions by prioritizing the immediate replanting of older corals, thus avoiding extended recovery times associated with fragmented corals (Konh & Parry, 2019). Notable reef restoration initiatives in Hawai‘I include a significant project led by the State of Hawai‘i Division of Aquatic Resources Coral Restoration Nursery; This ex-situ nursery utilizes the micro-fragmentation and fusion method, demonstrating significantly improved rates of covering large modules with coral tissue compared to previous methods (Forsman et al., 2018). Future efforts include large-scale reef substrate restoration, collaborative initiatives targeting rare coral species populations, and projects such as ‘Seeding the Reef’ involving larvae and colonies (Forsman et al., 2018).

Shifting the focus to Southeast Asia, restoration initiatives in Indonesia employ artificial reefs to remediate areas impacted by dynamite and cyanide fishing (Bostrom-Einarsson et al., 2020). In the Philippines, coral gardening is a prevalent practice, responding to a 2004 Global Coral Reef Monitoring Network study revealing an annual steady decline in coral cover of 3–5% (Shaish et al., 2007). Noteworthy success has been achieved in Bolinao, the Philippines, where a cost-effective nursery, established with common materials, attained over 85% survivorship after one year (Shaish et al., 2007).

Indian Ocean

Throughout history, small island nations like the Maldives have traditionally relied on thriving coral reefs for sustenance, coastal protection, and essential resources for construction or trade (Dehnert et al., 2021). Situated in the Indian Ocean, the Maldives owes its existence to 26 coral atolls; however, this invaluable asset has recently experienced substantial deterioration due to recurrent coral mass bleaching events and various regional stressors, including pollution, diseases, and predator outbreaks (Dehnert et al., 2021). While coral restoration is not widely implemented in the Maldives, the predominant method involves the application of metal frames, known as spiders, as artificial reefs (Dehnert et al., 2021). Although not widely implemented, there is an urgent need for active interventions, particularly effective coral restoration, to preserve the integrity of the “high-value” resort reefs. Coral gardening, with its scalability, potential for local tourism opportunities, and ability to intervene before disturbances like constructions and land reclamation occur, emerges as a suitable approach (Hein et al.

2021). However, a significant lack of validated information on the best restoration practices specific to the Maldives remains evident.

The repercussions of the 1998 mass coral bleaching event reverberated across Indian Ocean reefs, leading to a substantial decline in live coral cover in the Seychelles Archipelago, particularly affecting the inner granitic islands of the Seychelles Plateau (Frias-Torre et al., 2023; Montoya-Maya et al., 2016). Natural coral recovery in Seychelles has been notably slow since 1998, so responding to this challenge, a large-scale coral reef restoration project unfolded in Seychelles between 2014 and 2015 (Frias-Torre et al., 2023) This project, executed as a boutique restoration tailored to the specific needs of a hotel resort, adhered to science-based principles of ecological restoration and aimed to expedite coral recovery while aligning with the ecological restoration framework (Frias-Torre et al., 2023). This dual approach of involving the private sector in broader ecosystem restoration and implementing boutique restoration projects underscores Seychelles' multifaceted efforts to address environmental challenges and promote sustainable practices (Frias-Torre et al., 2023).

Despite slow post-bleaching recovery in the inner granitic islands of Seychelles, active restoration efforts were initiated to assist natural recovery. Between November 2011 and June 2014, a total of 24,431 nursery-grown coral colonies from various species were transplanted to a degraded reef at Cousin Island Special Reserve (Montoya-Maya et al., 2016). Coral recruitment was assessed over 14 months, revealing that six months after transplantation, the total spat density at the transplanted site surpassed that of healthy and degraded sites (Montoya-Maya et al., 2016). The study concludes that enhanced natural coral settlement resulting from coral transplantation holds promise for the success and long-term sustainability of large-scale coral reef restoration, particularly in projects aiding the recovery of naturally degraded reefs in the Seychelles (Montoya-Maya et al., 2016).

CORAL RESTORATION METHODS

Reef Gardening

Coral gardening, also referred to as coral aquaculture of coral farming, is a prevalent method in coral restoration, which involves the careful extraction of tissue and skeleton portions from healthy wild coral colonies to establish nursery stock (Cavasos, 2019; Bostrom-Einarsson et al., 2018). This two-step strategy begins by creating a sheltered nursery either in their natural habitat or a controlled environment, fostering a robust pool of farmed coral and meticulously tending to the coral recruits (Epstein et al., 2001). Subsequently, the nursery stock, consisting of nurtured and pruned small fragments, is transplanted onto degraded reefs in the second step of coral gardening (Cavasos, 2019; Wagner, 2022; Epstein et al., 2001). The intermediate nursery stage protects the coral fragments from damaging conditions during their most vulnerable stages, enhancing their post-transplantation survival upon outplanting onto the reefs (Bostrom-Einarsson et al., 2020). During the nursery stage, when the coral fragments have grown to a sufficient size, they undergo either further fragmentation, expanding the pool of available coral fragments for transplantation without the need for additional collection from natural reefs, or they are directly transplanted onto degraded reefs (Bostrom-Einarsson et al., 2020; Wagner, 2022). Acroporids are typically used in the coral gardening strategy because branching corals are fast growing, have rapid healing capacity, and provide four potential types of coral material for nursery purposes: small colonies, branch fragments, nubbins, and larvae (Bostrom-Einarsson et al., 2018; Epstein et al., 2001). Small colonies tend to grow the fastest out of the four coral material types while nubbins, which are minute fragments, tend to have the lowest growth rate (Epstein et al., 2001).

Coral nurseries can be either in-situ or ex-situ, depending on local conditions, and the success of coral gardens heavily rely on site selection for in-situ nurseries (Bostrom-Einarsson et al., 2018; Wagner, 2022). In-situ nurseries are placed in sheltered environments with favorable conditions, while ex-situ nurseries employ tanks or aquaria with controlled environments (Bostrom-Einarsson et al., 2018). Essential factors in selecting nursery locations include proximity to wild coral populations, depth, water movement, bottom characteristics, area size, adjacent habitat, competitors, human activities, accessibility, number of nurseries, and permits (Wagner, 2022). Various structures, including those on the substratum, mid-water structures, PVC trees, and fixed tables, are utilized for coral nurseries, with many located in Marine Protected Areas (MPAs) to shield the growing corals from stressors, such as fishing, dredging, snorkeling, and scuba activities (Bostrom-Einarsson et al., 2018; Wagner, 2022). Beyond restoration, coral gardening provides habitat for marine life, enhances genetic diversity, supports experimental research, and offers unique volunteering opportunities (Wagner, 2022).

Coral gardening equipment, considered cost-effective and readily available, includes ropes, PVC, metal frames, cinder block platforms, Reef Balls, and floating structures (Wagner, 2022; Young et al., 2012). Among these, metal frames are common for their availability, affordability, and resistance to storm damage, with documented survivorship ranging from 63%-95% (Young et al., 2012). Mid-water nurseries with suspended fragments report increased survival rates of 86%-97.5% in the first year

(Young et al., 2012). Other data reports an overall average of 66% survivorship with the coral gardening method (Bostrom-Einarsson et al., 2020).

Despite reported high survivorship in the short-term, some studies indicate a decline after long-term outplanting. Ware et al. (2020) in the Florida Keys showed survivorship ranging from 0%-35% after five years and 0%-10% after seven years. Long-term monitoring beyond one or two years is limited (Ware et al., 2020). Challenges associated with coral gardening include susceptibility to storm damage, predation by corallivorous snails, fireworms, and damselfish, along with temperature variations, poor water quality, competition by algae and space competitors, cost, and the need for sustained maintenance (Wagner, 2022; Young et al., 2012; Hein et al., 2021).

Direct Transplantation

One of the earliest and widely practiced methods of coral restoration involves direct transplantation of coral fragments, which involves relocating coral fragments from a donor to a recipient reef (Bostrom-Einarsson et al., 2020; Wagner 2022). This technique bypasses the intermediate nursery phase from the coral gardening method and is particularly employed to rescue corals facing potential destruction or disturbance due to planned construction activities, such as dredging, port and marina expansion, and beach renourishment (Bostrom-Einarsson et al., 2020; Wagner 2022). Branching varieties are the most commonly transplanted corals, having an average survival rate of 64% (Bostrom-Einarsson et al., 2020).

The success of direct transplantation depends on factors such as the size and health of the coral fragments, method of transportation and attachment, and the environmental conditions during transplantation (Bostrom-Einarsson et al., 2018).

Although overall studies report an average survival rate of 64%, there is limited data on the longer-term effect of direct transplantation on coral reef communities, beyond survival and growth (Bostrom-Einarsson et al., 2018). Directly transplanted coral fragments often exhibit lower growth rates, survivorship, and fecundity compared to wild-established colonies, primarily attributed to the energy required for healing and attachment to the substrate (Omori, 2019). Smaller fragments, in particular, face challenges such as smothering by algae or loss during transplantation, and have lower survival compared to larger fragments (Omori, 2019).

The collection of coral fragments involves equipment such as wire cutters, underwater scissors, chisels, or hammers (Omori, 2019). Using 'corals of opportunity,' fragments broken off by storms or other causes, is also possible with this method (Omori, 2019). Other equipment utilized in direct transplantation include Reef Balls, cement pucks, concrete rosettes, cement, underwater epoxy, plastic cable ties, metal wire, nails, and bolts (Young et al., 2012). Proper attachment methods, such as cable tie, underwater epoxy, or metal stakes, contribute to the success of directly transplanted coral fragments, whereas other methods of attachment include directly wedging them into crevices or placing loosely onto the substrate (Bostrom-Einarsson et al., 2018; Young et al., 2012).

While direct transplantation is a simple and cost-effective method that can involve local community participation, it can be considered less sustainable due to its emphasis on the ongoing harvest of coral fragments (Omori, 2019; Wagner, 2022). Nevertheless, it proves effective in salvaging corals that would otherwise face loss.

Micro-fragmentation

Similar to coral gardening, micro-fragmentation involves the utilization of coral fragments, typically in $1cm^2$ sizes (Wagner, 2022). This method, introduced in the late 2010s, has become the preferred choice for propagating branching corals over the years (Goergen et al., 2020; Page & Vaughan, 2014). The process entails extracting a small portion of a branching or massive coral, affixing it to disks or 'plugs,' and then placing it into nurseries for growth (Wagner, 2022; Bostrom-Einarsson et al., 2020). Micro-fragmentation significantly augments the number of clones from a single colony in a short time frame, enabling the outplanting of coral arrays that can fuse together and 'reskin' the substrate (Goergen et al., 2020; Page & Vaughan, 2014). Fragmentation of acroporids is accomplished by separating branches from a donor colony, where many massive coral species exhibit clonal reproduction needed for micro fragmentation but have been overlooked due to their slow growth and thick skeleton in many projects (Page & Vaughan, 2014). However, recent advancements in micro-fragmentation techniques have led to more inclusion of massive and encrusting coral species (Wagner, 2022).

Most restoration efforts use a diamond band saw to cut thick coral skeleton into the necessary fragment sizes and place them on plugs, disks, or tiles (Page & Vaughan, 2014; Bostrom-Einarsson et al., 2020). Otherwise, the equipment and process for micro-fragmentation align with those of coral gardening. Within a week, fragments grow over cut edges, and after 2-6 weeks, the corals begin producing new polyps (Page & Vaughan, 2014). These fragments are grown to approximately $6cm^2$ before outplanting (Page & Vaughan, 2018). After approximately 12 months, the fragments can be either micro-fragmented again or outplanted onto degraded reefs, similar to the coral gardening approach (Bostrom-Einarsson et al., 2020). Research has shown high survivability and rapid growth of micro-fragmented corals once outplanted (Bostrom-Einarsson et al., 2020). In the Florida Keys, three micro-fragmented coral species, *M. cavernosa, O. 23aveolate, and D. clivosa*, exhibited a 99.5% survival rate within 3-10 months post-cutting (Page & Vaughan, 2014). However, limited studies report results of micro-fragmentation outside controlled aquarium environments (Bostrom-Einarsson et al., 2018).

Coral Larvae

Larval-based restoration, also referred to as larval restoration, larval enhancement, sexual propagation, and larval reseeding, entails seeding damaged reef areas with larvae to artificially increase larval settlement and recruitment (McLeod et al., 2022). This method can utilize settled spat obtained from controlled crosses or large-scale wild spawning events for subsequent outplanting (Caruso et al., 2021). The collection of coral larvae is integral to enhancing genetic diversity amidst growing environmental stressors and integrating coral larvae restoration with coral gardening amplifies the impact on genetic diversity (Bostrom-Einarsson et al., 2020; Baria et al., 2012). The utilization of sexually reared corals in restoration efforts remains largely experimental and appears to be more costly compared to asexual propagation techniques (Guest et al., 2010).

Sexual reproduction provides opportunities for selective breeding, enhancing genotypic diversity and adaptive potential (Caruso et al., 2021). These larvae can be cultured either in-situ within floating enclosures on reefs or ex-situ in laboratory or aquaculture facilities to optimize production and retain them for restoration, preventing their dispersal in currents away from the targeted reef areas (McLeod et al., 2022). Upon reaching competency for settlement, larvae are released onto reefs using various methods, such as containment under mesh sheets or tents for small-scale manipulative experiments, larval clouds onto damaged reef areas, or deployment through underwater robotic vehicles near reef surfaces (McLeod et al., 2022). Globally implemented, larval-based restoration leverages the coral's ability to generate millions of offspring at predictable times, reducing early life stage mortality and minimizing losses from larval dispersal (McLeod et al., 2022). Despite the scalability advantage of sexual reproduction, its limited application stems from its expense, time-consuming nature, and restricted success (Caruso et al., 2021; Young et al., 2012).

In reef restoration, two experimental methods for establishing settled coral larvae are being explored: settling larvae on purpose-made substrates for nursery rearing before transplantation and introducing competent larvae directly to degraded reef areas (Guest et al., 2010). The latter approach is less likely to contribute to rehabilitation due to high post-settlement mortality, while the former, when coupled with nursery or hatchery rearing, holds potential (Guest et al., 2010).

Suzuki et al. (2020) have introduced an innovative coral restoration method known as the coral larvae cradle, designed to enhance larval supply and post-settlement outcomes in sexually assisted coral restoration projects. The cradle, shaped like a funnel and positioned vertically in the water column, is covered with nylon mesh and features a wide opening over seabed corals that narrows into a closed $9m^2$ cylinder at the surface (Suzuki et al., 2020; Wagner, 2022). The primary objective is to reduce initial coral mortality by ensuring high fertilization rates, larval survival, and settlement on artificial substrates (Suzuki et al., 2020). Specifically designed for Scleractinians, especially broadcast spawning Acropora species, this entirely in situ process eliminates the need for land facilities (Suzuki et al., 2020). With impressive success rates, Suzuki et al. (2020) achieved a 99.1% survival rate after 4 days, establishing the larval cradle as a method capable of acquiring several million coral larvae without requiring land facilities, suitable for large-scale coral restoration endeavors (Wagner, 2022)..

In reef restoration efforts, utilizing coral larvae reared outside their natural habitat has shown promising outcomes in both the Caribbean and the Pacific (Baria et al., 2012). Successes in the Caribbean include the successful rearing and outplanting of *A. palmata* larvae onto wild reefs, where they spawned concurrently with wild colonies (Baria et al., 2012). The ability of larvae-raised corals to reproduce similarly to wild corals supports the use of this method for aiding the natural recovery of depleted coral populations (Lirman & Schopmeyer, 2016). Furthermore, *A. palmata* reared from larvae exhibited spawning behavior just four years after being placed on reefs in Curacao, indicating the potential for accelerated reproductive activity in restored populations (Lirman & Schopmeyer, 2016).

While the cultivation of larvae presents advantages such as higher genetic diversity and minimal reef damage, it is essential to acknowledge that the process is labor-intensive and requires specialized facilities (Guest et al., 2010). Different approaches are needed for Atlantic and Indo-Pacific reefs due to variations in reproductive strategies and despite having promising results, more research is needed before sexual propagation methods are widely applied in restoration efforts (Guest et al., 2010).

Artificial Reefs

Artificial reefs are intentionally placed substratum structures on the seabed to establish potential habitats for corals (Bostrom-Einarsson et al., 2018; Wagner, 2022). These structures not only offer habitats for coral and marine species but are also designed for shoreline protection, erosion prevention, and mitigation of human activity impacts (Young et al., 2012; Burt et al., 2009). Typically serving as the foundation for coral transplantations, artificial reefs exhibit a 66% survival rate (Bostrom-Einarsson et al., 2020). Various structures, such as Reef Balls, EcoReefs, and BioRock, contribute to this effort.

Reef Balls, hollow structures crafted from textured concrete, mimic eroded massive coral heads and are increasingly used worldwide for fisheries enhancement (Ortiz-Prosper et al., 2001; Wagner, 2022). Their shape and texture promote coral growth and provide refuge for marine life. Studies have shown that Reef Balls, left bare for four months, become habitats for sponges, tunicates, feather dusters, barnacles, and various fish species (Wagner, 2022). They are also appropriate structures to plant several massive coral colonies; Hudson et al. (1989) constructed a small patch reef using 23 hollow concrete domes embedded with 32 hard corals, with a decade post-submersion coral survivorship rate of 87.5% (Ortiz-Prosper et al., 2001). Even though these concrete structures can recruit coral colonies, transplantation methodologies allow selection of particular coral species, which can minimize the time needed for effective establishment (Ortiz-Prosper et al., 2001).

EcoReefs, snowflake-shaped unglazed ceramic stoneware (chemically inert and non-toxic), emulate branching corals, which creates a spatially complex habitat (Wagner, 2022; Moore & Erdmann, 2008). EcoReefs are designed to meet the ecological needs of reef fish and coral, while being economically feasible to scale up restoration projects (Moore & Erdmann, 2008).

Biorock, or electric reef technology, consists of permeable, porous, self-repairing structures that stimulate settlement, growth, and survival for marine life (Goreau & Prong, 2017; Nugroho et al., 2023). Low voltage initiates electrolysis of seawater, creating a calcareous substrate for coral settlements (Wagner, 2022). Studies indicate an increased growth rate of acroporids on Biorock compared to control conditions (Nugroho et al., 2023). Biorock is cheaper than concrete or rock seawalls and breakwaters and is a cost-effective technology for protecting eroding coasts (Moore & Erdmann, 2008).

To assess the success of artificial reefs as complete ecosystems, long-term comparisons between artificial reefs and natural reefs in similar environmental conditions are necessary (Bostrom-Einarsson et al., 2018). A five-year study comparing artificial reefs and adjacent natural reefs showed a 70% and 63% similarity in scleractinian and octocoral community composition, respectively, by the study's conclusion (Hannes and Floyd, 2009). Additionally, different substrate materials, such as Gabbro rocks, have been found to increase coral recruits, suggesting their preferential use over concrete and sandstone (Hannes & Floyd, 2009, Burt et al., 2009). Since most artificial reef structures are made from concrete, they might be contributing to the very problem they are trying to mitigate; concrete production is responsible for 5-7% of global carbon emissions (Bostrom-Einarsson et al., 2020).

COSTS OF CORAL RESTORATION

Coral reef restoration is a viable yet potentially expensive endeavor, contingent upon the selected site and employed techniques (Bayraktarov et al., 2019). Table 1 provides an overview of various techniques and their associated costs per hectare (Bayraktarov et al., 2019). Notably, restoration projects have been predominantly of short duration and limited spatial scale; 60% of initiatives are monitored for less than 18 months, with a median size of 100 m^2 for restored reefs (Bostrom-Einarsson et al., 2020; Bayraktarov et al., 2019).

Outplanting coral, a costly venture in both asexual and sexual restoration projects, imposes limitations on spatial scale, typically confined to smaller projects under 1 hectare (Vardi et al., 2012; Chamberland et al., 2017). Operational costs encompass man-hours and operational expenses, labor costs vary across countries, and transplantation costs fluctuate depending on the restoration method employed (Omori, 2019). Costs can range significantly, from approximately $10,000 USD to $50,000,000 USD per hectare, and determining whether these costs are reasonable depends on the economic benefits of the local coral reefs (Omori, 2019). Coral reefs offer varying economic benefits in different regions of the world; Hawaiian coral reefs are estimated to be worth $9.7 billion USD, Philippine reefs at around $6 million annually, and the Caribbean Sea's reefs valued between $3.1-$4.6 billion annually (Omori, 2019; Wagner, 2022).

Published costs for farming one fragment containing at least one sexual propagule range from $5.3 to $163 USD, while asexual propagation techniques are less expensive, with one fragment ranging from $0.15 to $13.2 USD (Omori, 2019). Vardi et al. (2012) estimated that outplanting 3,000 *A. palmata* each year in the Caribbean for 5 years, matching the nursery output and costs as *A. cervicornis,* would result in about 30% coral cover and cost about $3 million US. The estimated cost of a matured colony is higher, with a single 2.5-year-old *A. millepora* grown from a sexual propagule costing at least $61 USD (Omori, 2019). Improving early survival rates during the nursery phase can reduce costs, as demonstrated by producing 6-month-old colonies of *A. valida* for approximately $11.2 USD (Omori, 2019).

The most cost-effective coral restoration projects utilize direct transplantation, costing $11,717 USD per hectare in developing countries, while the least cost-effective involves a combination of substrate stabilization and transplantation, estimating about $2,879,773 USD in developing countries (Bayraktarov et al., 2016). A study done by Doropoulos et al. (2019) compared costs of slick harvesting and direct transplantation; slick harvesting, a method of harvesting, developing, and releasing wild coral spawn, has a median cost of $55 USD per colony after 4 years, while transplantation cost $206 USD. Maintenance costs also vary, with keeping one fragment in land-based nurseries for 2.5 years costing $325 USD, of which 79% is covering operational costs of the nursery, while one coral fragment raised on an in-situ reef for the same duration costs only $13 USD (Omori, 2019).

In the Philippines, direct seeding of coral larvae incurs a production cost of $1,654 USD, or $20.94 USD per each of the 79 surviving colonies after 35 months (Omori, 2019). Estimates of the average monthly cost for maintaining 1,000 corals in a single raceway is about $200, covering food, cleaning, pumps, substrates, and other supplies needed for pest mitigation and water quality monitoring (Koch, 2022).

Unfortunately, this estimate does not include the cost associated with the raceway, fragmentation equipment, or staff salary.

Approximately 30% of total restoration project costs account for artificial substrates with settled corals and manual securing using binding materials such as cable ties and epoxy (Chamberland et al., 2017). In contrast, larval rearing and larval settlement typically account for less than 50% of costs (Chamberland et al., 2017). Utilizing proxies and indicator meters may help reduce costs and the time required for monitoring, addressing the knowledge gaps for long-term coral restoration effects (Bostrom-Einarsson et al., 2020). Due to its cost, outplanting should be regarded as a short-term solution within a limited geographical region, rather than a comprehensive solution for this basin-wide problem (Vardi et al., 2012).

Table 1: Cost effectiveness of different outplanting techniques used in coral restoration, showing the median, maximum, and minimum costs.

		Restoration Cost (2010 US$/ha)		
Restoration Technique	n	*Median (±SD)*	*Minimum*	*Maximum*
1. Direct transplantation	20	218,305 (±2,339,609)	9,198	8,382,653
2. Larval enhancement	6	523,162 (±1,878,894)	6,262	4,333,826
3. Coral gardening (overall)	3	351,661 (±136,601)	130,000	379,139
a. Collection & nursery phase	5	28,075 (±20,472)	9,262	56,150
b. Transplantation phase	2	761,864 (±1,033,831)	30,835	1,492,893
4. Enhancing artificial substrates with an electrical field	0			
5. Substrate addition (artificial reef)	10	3,341,754 (±44,100,144)	14,076	142,667,803
6. Substrate stabilization	8	370,986 (±9,040,923)	91,044	26,100,000

Note. Reprinted from "Motivations, success, and cost of coral reef restoration," by Bayraktarov et al., 2019, *Restoration Ecology, Vol. 27,* p 986

CONCLUSION

The surge in popularity of coral restoration has prompted the establishment of nurseries across every tropical ocean basin, with coral gardening and transplantation emerging as the predominant methods employed (Wagner, 2022; Bostrum-Einarsson et al., 2020). Coral gardening has garnered preference as the primary approach for coral reproduction and reef restoration in the Caribbean, Indo-Pacific, and Indian Ocean regions (Lirman & Schopmeyer, 2016; Wagner, 2022). Currently, coral restoration initiatives are active in 52 countries, with concentrated efforts in the USA, Philippines, Thailand, and Indonesia (Bostrom-Einarsson et al., 2018). However, the survivorship and costs associated with each coral restoration method exhibit significant variation, influenced by factors such as materials, labor costs, and geographical locations.

Despite limited data on the long-term survivorship of outplanted corals, Puerto Rico stands as an exemplar with the oldest known gardening program in the Caribbean, demonstrating success over seven years (Lirman & Schopmeyer, 2016; Wagner, 2022). In Florida, staghorn corals have thrived for over five years post-outplantation, and elkhorn colonies raised from larvae have successfully spawned after four years. The cultivation of corals in nurseries replicating natural growth patterns underscores the supportive role of coral gardening in facilitating the natural recovery of coral populations (Lirman & Schopmeyer, 2016).

While current coral restoration studies predominantly focus on the survival and growth of outplanted corals, limited attention has been devoted to examining the broader impact of restoration on the reef community and essential ecological processes. Recognizing the significance of reef diversity as an indicator of ecosystem health, it is essential to consider ecological processes during the site selection for coral nurseries (Ladd et al., 2019). Successful locations emphasize existing coral cover, available clean substrate, and water depth, but a holistic approach must integrate the ecological processes crucial for a functional and sustainable coral reef (Ladd et al., 2018).

To ensure the success of restoration and conservation efforts for threatened coral species and reefs, it is crucial to address the factors contributing to their decline, including trophic ecology and nutrient cycling (Garrison & Ward, 2015; Ladd et al., 2016; Ladd et al., 2018). The integration of fundamental ecological processes into restoration methodologies is paramount for enhancing the success of coral restoration initiatives and fortifying the functional aspects of reef ecosystems. Key processes such as predation, herbivory, and nutrient cycling play pivotal roles in coral growth and population recovery, necessitating effective management and leveraging of these processes to propel the field forward (Ladd et al., 2018). To enhance the effectiveness of comprehensive evaluations, researchers should incorporate key indicators such as coral diversity, herbivore biomass and diversity, benthic cover, recruitment, coral health, and structural complexity (Hein et al., 2017). This approach enables a more thorough characterization of the efficacy of coral restoration efforts in promoting reef resilience (Hein et al., 2017).

While the pursuit of a universal coral restoration method is desirable, this work emphasizes the absence of a perfect or superior method. Restoration projects must, therefore, consider multiple components. Ongoing advancements in techniques and knowledge are shaping the field of coral restoration, yet human intervention remains pivotal at this stage to safeguard and preserve coral reefs and their ocean environment.

CHAPTER 2: EFFICACY OF CORAL RESTORATION IN BONAIRE: COMPARISONS IN SPACE, TIME, IMPACTS, AND COSTS

ABSTRACT

Globally, 75% of coral reefs are in crisis due to a multitude of anthropogenic disturbances stressing coral reefs such as coral bleaching and disease and increased macroalgae prevents their recovery. This has stimulated increased active coral restoration projects worldwide seeking to address the decline of reef corals to protect the coral and reef ecosystems. Active restoration (i.e., the planting of corals on reefs) is a forty-year-old practice in which organizations propagate coral fragments to outplant on the reef. Reef Renewal Bonaire outplanted 1,266 *Acropora palmata* and 530 *Acropora cervicornis* on reefs in Bonaire from 2012 to 2021. I used previously established coral reef monitoring sites and replicated coral reef monitoring transects there to create 100 m² quadrats to quantify the population density and vigor of *A. cervicornis* and *A. palmata* at the 11, 20-year long-term monitoring sites. My surveys were conducted at 5 and 10 m depths. Three of the monitored reefs were active restoration sites that had received 1,796 acroporid coral outplants, and three were adjacent control reefs, with the remaining five monitored reefs being regional control reefs. I found no acroporids at 10 m depth at any monitored reef site. However, at 5 m depth, 13 acroporids were recorded at four survey sites but only five from outplant sites. The two sites with the highest acroporid densities were Calabas (restoration site) and Karpata (control site). Of the monitored reefs, Calabas and Oil Slick were the only two restoration reef sites having acroporid corals at 5 m. Calabas was the only southern site where acroporids were recorded whereas acroporids were recorded at the three

northernmost sites (i.e., No-Dive Reserve, Karpata, and Oil Slick). The efficacy of coral restoration could not be confirmed based on this study of Bonaire's long-term monitored reefs.

INTRODUCTION

Coral reefs are among the most important ecosystems in the world, as they hold more than 30% of marine biodiversity (Ladd et al., 2019). However, these diverse ecosystems are in a crisis due to a multitude of anthropogenic disturbances, including pollution, overfishing, declining water quality, and climate change. These disturbances have caused 75% of the world's reefs to be threatened, and with the increasing frequency and severity of tropical storms and bleaching events, the natural recovery of these coral reefs is constrained between disturbance events (Omori, 2019; Vardi et al., 2021). Perhaps most concerning, since the 1970s, the Caribbean has seen a dramatic loss of 80% coral cover, with degraded reefs experiencing positive feedback loops (i.e., a decrease in coral cover increases macroalgae cover, therefore decreasing coral productivity), locking reefs into their degraded state (Ladd et al., 2019).

Despite their importance, coral reef ecosystems challenge management with their complexity. The rapid scale of coral reef decline has caused global management priorities to shift and prioritize active restoration (i.e., a direct approach in which programs devote time, money, and resources to increase the abundance of corals on reefs) instead of passive interventions (i.e., processes that depend on natural recovery, like marine protected areas) (Vardi et al., 2021). These active restoration methods around the world seek to address the decline of coral reefs and protect their ecosystems. Over the past 30 years, coral restoration has relied on coral transplantation (i.e., moving and securing coral fragments onto reef substrata) as its most common technique (Hein et al., 2017). Most restoration projects using coral transplantation follow the coral gardening concept, where coral fragments are grown floating in mid-water nurseries until suitable size for ouplanting onto the reef (Hein et al., 2017). This process of coral transplantation aims to increase coral cover, develop large spawning hubs, and maintain genetic and species diversity within the reef (Omori, 2019).

Despite a global increase in restoration interest, there has been much skepticism within the scientific community about coral restoration and its effectiveness. Although restoration and outplanting efforts have increased over the years, solely replanting corals will not stop the global drivers of coral decline. Previous research has criticized restoration, claiming it is inadequate to address the scale at which reefs are degrading (Hein et al., 2017). Many argue that reef ecosystems are too complex and that the corals survive transplantation and relocation poorly (Precht et al., 2005). Additionally, the lack of scientific evidence on the outcomes and benefits of coral restoration has been criticized: There are no standard measurable protocols for evaluating the indicators of success and the effectiveness of coral reef restoration (Hein et al., 2017).

In order to increase restoration success and support reef ecosystem functions, fundamental ecological processes need to be incorporated into coral restoration efforts. Ladd et al. (2018) surveyed 116 scientific papers on coral restoration to find that projects primarily focused only on the growth and survivorship of coral, while less than 20% of the papers incorporated any aspect of ecological processes. Managing and harnessing key ecological processes (e.g. predation, herbivory, and nutrient cycling) is an essential next step in advancing the field of coral restoration, as these processes facilitate coral growth and the recovery of coral populations (Ladd et al., 2018). Herbivory is one critical process that supports ecosystem function, suppresses macroalgae cover, increases coral growth and recruitment, and facilitates the recovery of coral populations after a disturbance (Mumby & Steneck, 2008; Ladd et al., 2018). Hein et al. (2017) determined that comprehensive evaluations should require indicators, including coral diversity, herbivore biomass and diversity, benthic cover, recruitment, coral health, and structural complexity. These indicators would allow researchers to better characterize the effectiveness of coral restoration for reef resilience.

Hein et al.'s (2017) review found that coral restoration efforts have primarily focused on evaluating short-term biological responses of transplanted coral fragments. In these evaluations, coral transplant growth and survival are the most commonly assessed variables (Hein et al., 2017). The mean duration of monitoring in all 83 coral transplantation studies within Hein et al.'s (2017) assessment was under two years, with a majority monitoring for one year or less. (Hein et al., 2017). Although these timeframes are reasonable for evaluating transplantation feasibility, they are unreasonable for evaluating the success and usefulness for coral reef restoration or re-establishing coral reef ecosystems. One year is insufficient time to determine the effectiveness of coral restoration. It is even made more complicated given the complexity of the many life history characteristics of scleractinian corals and year-to-year variations in the growth of transplanted corals (Hein et al., 2017). Similarly, Bostrom-Einarsson et al. (2020) determined that 60% of restoration case studies reported only the first eighteen months of coral restoration monitoring results. In addition to a small temporal scale, most restoration projects were small in size, with 100 m^2 as the median size of a restored reef (Bostrom-Einarsson et al., 2020).

Restoration most commonly uses fast-growing branching corals, such as *A. palmata* and *A. cervicornis*. In fact, 59% of the Caribbean case studies on coral restoration from Bostrom-Einarsson et al. (2020) used fast-growing branching acroporid corals. They had a reported average survival rate of 66% for the year. However, when examined more closely in Omari's (2019) research, directly transplanted coral fragments have lower growth rates and survivorship than wild established colonies of similar size. This difference is likely caused by the physiological stresses of outplanted corals that forces them to allocate more energy into healing and attaching to substrates than growth (Omori, 2019).

Before the 1980s *A. cervicornis* was widespread in the coastal zones throughout the Caribbean, covering up to 70% of the shallow reef bottom in Bonaire (Meester et al., 2015). During this period, thickly branched *A. palmata* dominated the reef crest and shallowest depths (0-5 m depth), and the thinner-branched *A. cervicornis* dominated intermediate depths (5-25 m) on fore reefs (i.e., a portion of the reef facing open sea) (Meester et al., 2015). In the 1980s, white band disease, a bacterial disease specific to the genus *Acropora*, caused a significant decline in abundance throughout the Caribbean (Aronson & Precht, 2001). Population density of both acroporid species severely declined in Bonaire, causing changes in reef composition that have shifted coral dominance from *Acropora* to head corals (e.g. *Orbicella spp.*) (van Duyl, 1985, Relles, 2012). Today, shallow fore reefs are barren with scattered mound corals or covered in rubble and sand where acroporids were once abundant. Fast-growing acroporids are significant contributors to reef accretion (i.e., growth and gradual buildup of the reef), therefore with the decline in acroporids, these fore-reef habitats are at risk for drowning (i.e., when a reef cannot accrete vertically at the same rate as sea level rise) (Bakker et al., 2019).

Bonaire's coral restoration started in 2012 with an organization called Reef Renewal Bonaire (RRB). RRB uses propagation by fragmentation in combination with coral nurseries to grow endangered species of acroporid coral and outplant them onto the reefs surrounding Bonaire and Klein Bonaire. RRB grows both acroporid species in mid-water nurseries; the coral fragments are suspended on PVC structures floating in the water column (Reef Renewal Bonaire, 2021) (Fig 5). As of 2021, RRB had 17 restoration sites spanning a cumulative restored area of 7,890 m^2, where they reported 40,954 total corals outplanted (Reef Renewal Bonaire, 2021). This scale of restoration can be costly; according to Vardi et al. (2012), planting 3,000 *A. palmata* colonies per year for five years would cost three million dollars.

Compared to most Caribbean reefs the percent coral cover in Bonaire is relatively high, hovering around 46% since 2003. This compares to coral cover in the Eastern Caribbean that averaged 17.6%, with none of the coral reefs of the Eastern Caribbean exceeding 35% (Steneck et al., 2019). Since 2003, hurricanes and bleaching events have decreased coral cover, however, Bonaire's reefs have recovered from these events. Despite this high coral cover, *Acropora* spp. have yet to be quantified on any of the 307 transects from Steneck et al. (2019) throughout studies from 2004 to 2017. Acroporid depth ranges should include 10 m, which is the depth these transects were placed; therefore, my experiment sought to quantify any of the eleven long-term *Acropora* at Steneck et al. (2019) monitored sites in Bonaire.

In this study, I apply a BACI design (i.e., Before and After Controlled Impact) protocol, with three study sites classified as restoration reefs and three classified as adjacent control reefs. Restoration reefs had active restoration efforts happening on the reef while control reefs had no restoration but were alongside the three restoration reefs. Densities of acroporids were compared between restored and control reefs and relative past acroporid abundances (based on the acroporids found on the replicated 10 m monitoring transects).

METHODS

Survey Area

The distribution and abundance of *A. palmata* and *A. cervicornis* were quantified with visual surveys via SCUBA at eleven coral reef monitoring sites in Bonaire during March of 2023 (Fig 3). These eleven sites are involved with long-term monitoring from the last 20 years conducted by University of Maine and STINAPA. Listed from north to south, the eleven sites are No-Dive Reserve, Karpata, Oil Slick, Barcadera, Reef Scientifico, Forest, Front Porch, Calabas, 18th Palm, Windsock, and Bachelor's Beach. Of these reefs, Reef Scientifico, Calabas, and 18th Palm are fish protected areas (FPAs) and the no-dive reserve prohibits recreational diving.

The study sites are located on the west and leeward side of Bonaire, a Dutch Caribbean island, with the exception of study site Forest, located on Klein Bonaire, a small island off the west coast. Of these survey sites, three reefs had active restoration efforts for two species of coral (*A. palmata* and *A. cervicornis)*: Oil Slick, Calabas, and Bachelor's Beach. Sites next to restoration reefs without restoration efforts were classified as adjacent control reefs, which include Barcadera, 18th palm, and Windsock. Using BACI design, *Acropora* abundance comparisons between control reefs and restoration reefs were placed in the context of identical surveys at all eleven monitored reefs sites.

Survey Methods

At each survey site, quadrat sampling was conducted to quantify acroporid abundances. At 10 m depth, four replicate 10 m line transects were placed onto the reef parallel to shore. These previously defined transects were placed on top of permanent plates on the reef when present. Using these transects for position, 10 m by 10 m (100 m^2)

quadrats were used to visually survey for acroporids. To create the quadrats, kick cycle counts were used to estimate 5 m (i.e., however many kicks to reach 5 m measured using the transect), which was added on either side perpendicular to the transects to create a 100 m^2 quadrat (Fig 4). All *A. palmata* and *A. cervicornis* within the quadrats were recorded along with depth found, species, size in cm, and percent alive. *Acropora* observations were recorded on mylar sheets secured to a PVC cylinder worn on the wrist of each diver. If time permitted after visual surveys at 10 m depth, additional 5 m depth quadrats were conducted. There were no established transect lines at 5 m, so the kick cycle count estimate was used to create each quadrat. Four 10 m depth quadrats were surveyed at every site, but the number of quadrats surveyed at 5 m ranged from zero (e.g. Windsock and Forest) to five (e.g. No-Dive Reserve) and varied due to conditions and time availability.

Data Analysis

Abundance of *A. palmata* and *A. cervicornis* was calculated using density (number of acroporids per 100 m^2) and compared across all eleven reefs. Average size of living acroporids in cm was calculated by multiplying size recorded and percentage alive of each coral, then averaging all coral sizes at each site together.

Figure 3: The island of Bonaire and the locations of the 11 long-term monitored sites. * Indicates restoration reefs. ** indicates control reefs.

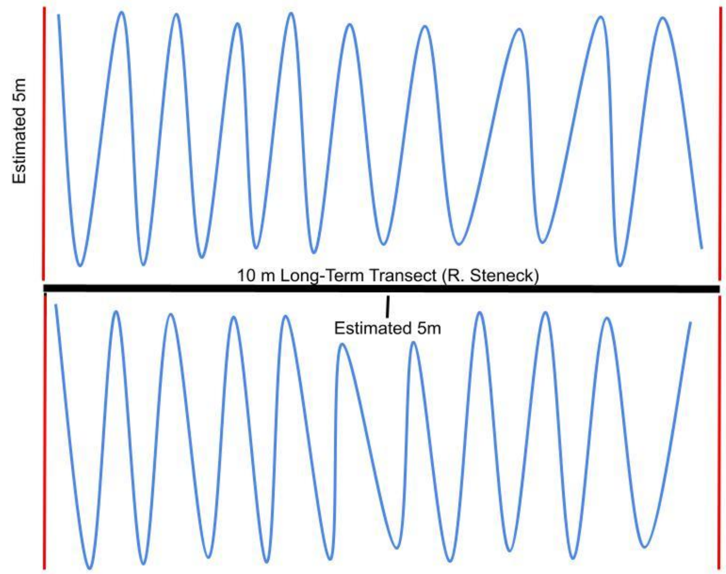

Figure 4: Sketch of visual survey over established transects. 5m was estimated using kick cycles over the transect, then that 5 m estimate is used to create a 10 m by 10 m quadrat. The black line represents the 10 m established transect, red lines represent the estimated 5 m on either side of the transect, and the blue lines represent the visual survey path taken within the quadrat.

Figure 5: Reef Renewal Bonaire's PVC trees to propagate *A. cervicornis*. Photos taken by Robert Steneck (2015).

RESULTS

Patterns Among Monitored Sites

No acroporids were found at any of our 10 m monitored stations.

A total of 13 corals were found over the eleven reefs and 72 quadrats at a depth of 5 m (Table 2). Acroporids were found at four of the eleven monitored reefs (Table 2). No data was found at Front Porch and Windsock reef due to time constraints for 5 m quadrats (Fig 6). Acroporids were found at two out of the three restoration reefs and two nonrestoration and noncontrol reefs (Fig 6). Oil Slick and Calabas were restoration reefs that *Acropora* coral was found on, while Bachelor's Beach, the other restoration site, had no *Acropora* found.

A. palmata were found at Karpata (four individuals at this nonrestoration reef) and Oil Slick (one individual at this restoration reef) (Table 2). *A. cervicornis* were found at No-Dive Reserve (two individuals, nonrestoration reef), Oil Slick (one individual, restoration reef), and Calabas (five individuals, restoration reef) (Table 2, Fig 8). Seven of the 13 total acroporids observed were recorded at restoration reefs of which 1,796 corals have been outplanted at those reefs.

Table 2. Summary of all outplanted acroporids for restoration and the number recorded at each monitored reef. Yellow highlights restoration sites where a total of seven corals were recorded out of 1,796 outplanted at those sites.

Sites	Total A. cervicornis outplanted for restoration	Total A. palmata outplanted for restoration	# *A. cervicornis* at 10m	# *A. palmata* at 10m	# *A. cervicornis* at 5m	# *A. palmata* at 5m	Total acroporids observed	Total area surveyed (m2)
No-Dive Reserve	0	0	0	0	2	0	2	900
Karpata	0	0	0	0	0	4	4	700
Oil Slick *	0	816	0	0	1	1	2	700
Barcadera **	0	0	0	0	0	0	0	700
Reef Scientifico	0	0	0	0	0	0	0	700
Forest	0	0	0	0	0	0	0	700
Front Porch	0	0	0	0	ND	ND	0	400
Calabas *	375	400	0	0	5	0	5	800
18th Palm **	0	0	0	0	0	0	0	600
Windsock **	0	0	0	0	ND	ND	0	400
Bachelor's Beach *	155	50	0	0	0	0	0	600
ISLAND WIDE TOTALS	530	1266	0	0	8	5	13	7200

Average density of *A. palmata* and *A. cervicornis* across all sites was 0.41/100m² (Table 3). Above average densities of acroporids per 100 m² were recorded for Karpata, Oil Slick, and Calabas. Karpata had the highest density of 1.33/100m^{2,} and below average densities were recorded for No-Dive Reserve of 0.4/100m^2 (Fig 6, Table 3). *A. cervicornis* was found at more sites compared to *A. palmata*, with *A. cervicornis* found at three reefs while *A. palmata* was only found at two (Fig 6). More *Acropora* individuals were found in the northern sites compared to the southern sites (Fig 6).

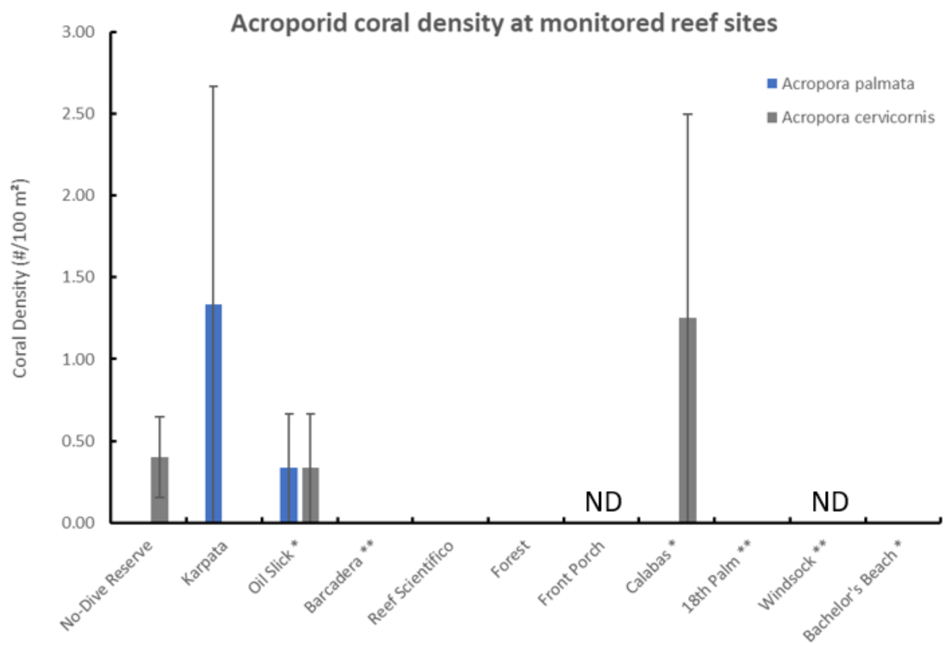

Figure 6: The density of *A. palmata and A. cervicornis* ($\pm SE$) found across eleven monitored coral reefs in Bonaire at 5 m depth. * Indicates restoration reefs. ** indicates control reefs. ND indicated no data.

Seven sites (i.e., Barcadera, Reef Scientifico, Forest, Front Porch, 18th Palm, Windsock, and Bachelor's Beach) averaged zero acroporids (Table 3). Oil Slick had the lowest above zero average of 0.67/100 m² (Table 3). Island averages of *A. cervicornis* were 0.89/site at 5m and were higher than *A. palmata* island averages of 0.5/site at 5m (Table 3).

No-Dive Reserve had the largest average size of living *A. cervicornis* at 55 cm, meanwhile Calabas had the smallest average size of living *A. cervicornis* at 17.1 cm (Table 3). Oil Slick has the smallest average size of living *A. palmata* at 4 cm, while Karpata had the largest at 23.4 cm (Table 3). Alive *A. palmata* sizes were less than half the size of alive *A. cervicornis* sizes (Table 3).

Table 3: Total area surveyed at 5m and 10 m, *A. palmata* and *A. cervicornis* abundances and density, and average size of living corals found at each site.

Site	Total area surveyed at 10m (m^2)	Total area surveyed at 5m (m^2)	# *A. cervicornis* present at 10m	# *A. palmata* present at 10m	# of *A. cervicornis* present at 5m	# of *A. palmata* present at 5m	*Density of A. cervicornis* at 5m / 100 m^2	Density of *A. palmata* at 5m / 100 m^2	Acroporids/100 m^2 (Ave per unit area)	Average size of living A. cervicornis (cm)	Average size of living A. palmata (cm)
No-Dive Reserve	400	500	0	0	2	0	0.40	0.00	0.40	55	0
Karpata	400	300	0	0	0	4	0.00	1.33	1.33	0	23.4
Oil Slick *	400	300	0	0	1	1	0.33	0.33	0.67	24.5	4
Barcadera **	400	300	0	0	0	0	0	0	0	0	0
Reef Scientifico	400	300	0	0	0	0	0	0	0	0	0
Forest	400	300	0	0	0	0	0	0	0	0	0
Front Porch	400	0	0	0						0	0
Calabas *	400	400	0	0	5	0	1.25	0.00	1.25	17.1	0
18th Palm **	400	200	0	0	0	0	0	0	0	0	0
Windsock **	400	0	0	0						0	0
Bachelor's Beach *	400	200	0	0	0	0	0	0	0	0	0
Totals & Areas	4400	2800	Ave #/site	Ave #/site	Ave #/site	Ave #/site	Ave density	Ave density	Ave density	Ave size	Ave size
Island Averages			0.00	0.00	0.89	0.56	0.22	0.19	0.41	32.20	13.69
Island Standard errors			0.00	0.00	0.56	0.44	0.14	0.15	0.19	11.60	9.69

* Indicates restoration reefs. ** indicates control reefs. Blank cells indicate areas of no data.

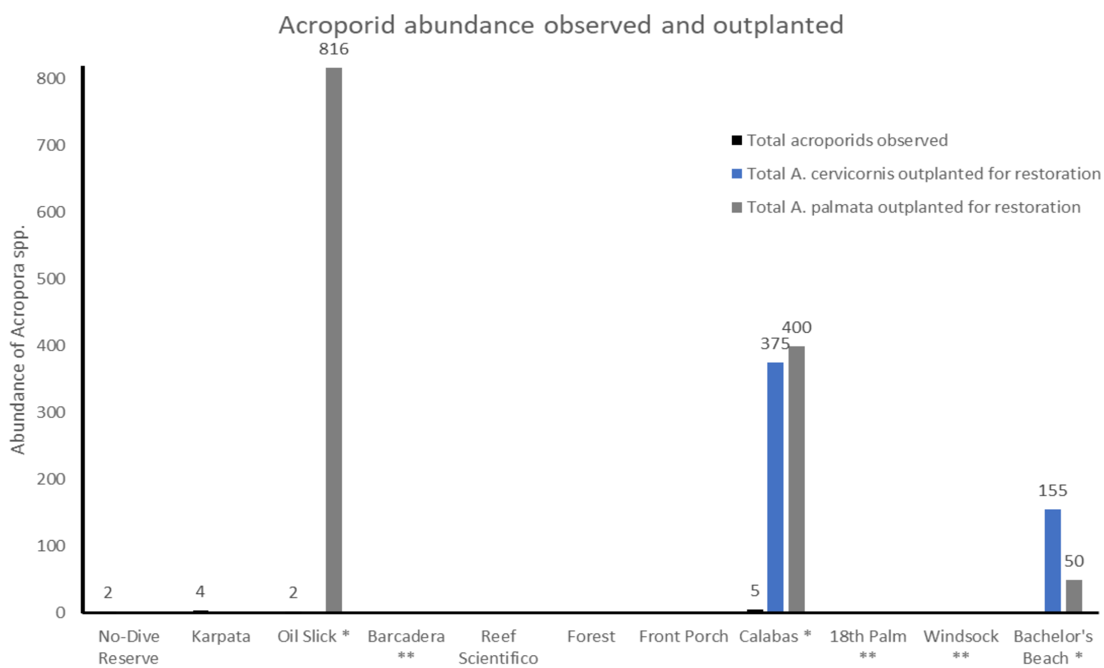

Figure 7: Total number of *Acropora spp.* found at each site compared to the total *A. cervicornis* and *A. palmata* outplanted from 2012 to 2021 (Reef Renewal Bonaire).
* Indicates restoration reefs. ** indicates control reefs.

When comparing *Acropora* individuals observed to Reef Renewal Bonaire's 2021 number of outplanted coral at each site there is no significant pattern (Fig 7). Oil Slick has the most outplanted coral, 816 *A. palmata*, while the next highest is Calabas with 775 total *Acropora* outplanted: 375 *A. cervicornis* and 400 *A. palmata* (Fig 7, Table 2). Bachelor's Beach has the least number of outplanted corals with 205 total: 155 *A. cervicornis* and 50 *A. palmata* (Fig 7, Table 2). The site with the most acroporids was Calabas, where I observed five *A. cervicornis*, but the overall range of acroporids observed was not large: zero to five (Fig 7).

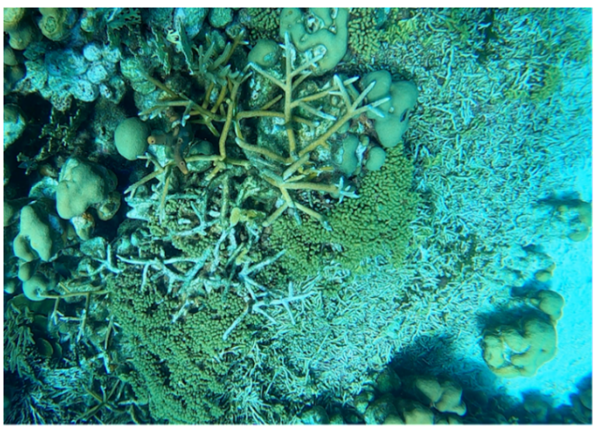

Figure 8: Photographs of *A. cervicornis* (left photo) (found at No-Dive Reserve reef, nonrestoration site) and *A. palmata* (right photo) (found at Karpata reef, nonrestoration site) found while surveying.

DISCUSSION

I found no evidence that coral restoration is increasing the acroporid densities at monitored reefs. The three northernmost study sites all had acroporids (eight in total), while the only southern study site with acroporids was Calabas. Northern sites and nonrestoration sites had similar acroporid abundances to restoration sites; therefore there was no consistent pattern of enhanced acroporids at restoration sites compared to nonrestoration sites (Fig 7). The density of outplanted coral is a critical factor in coral reef restoration success, and moderate densities (three corals/m) of *A. cervicornis* maximizes habitat production and minimizes the spread of disease and mortality (Ladd et al., 2018). The densities of acroporids on any of the reefs are not higher than 1.33/100 m^2, less than 2 per 100 m^2 quadrat, which is greatly lower than a moderate density of acroporids (Fig 6, Table 3). The standard error bars in Figure 4 overlap suggesting there are no statistically significant differences. Of the sampled sites, Karpata had the highest density of acroporids, while No-Dive Reserve had the lowest: both are nonrestoration, northern reefs, further suggesting that there is no correspondence between acroporid abundance and coral restoration.

So far, coral restoration has not proven to be effective at increasing the abundances of acroporids at restoration sites. This suggests a low "bang for the buck" but what does restoration cost? RRB is a non-profit, non-governmental organization that relies on donations for revenue. In 2021, RRB's total revenue from 2021 was 199,627 USD with a gross profit of 161,911 USD (Reef Renewal Bonaire, 2021). Using their revenue and the amount of coral's outplanted, I calculated that the cost per outplanted coral is $26.87.

Although not as costly as Vardi et al.'s calculations for coral restoration (3 million dollars for 15,000 coral outplanted), RRB is putting a lot of money into coral restoration efforts.

Because coral restoration has not been proven to be effective in this study, maybe restoration will be more effective enhancing ecological processes affecting reefs, as suggested by Ladd et al. (2018). Managing herbivory, predation, and macroalgae, along with coral restoration, would hopefully increase restoration success as well as increase coral recruitment and coral cover.

Although I found few acroporids over all study sites, the outplants could have been transplanted shallower than my 5 m quadrats. At many study sites, I visually observed multiple large *A. palmata* within 1 m of the surface while swimming back towards shore. At 18th Palm, I observed a large thicket of *A. cervicornis* very shallow and close to shore. For subsequent studies, shallower sampling, such as at 2.5 m could be informative.

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