A Regional Assessment of Coral Restoration within the Western Atlantic with a Case Study of Barbados

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Abstract

Coral reefs accommodate as much as a quarter of all known marine species and generate billions of US$ per year through ecosystem services such as coastal protection, fishery and tourism. Unfortunately, the hard coral global coverage is rapidly declining. Especially the coral reefs of the Caribbean Sea have experienced a severe deteriorating in the last four decades. Many projects have sought to restore or rehabilitate the reefs, often with varied success of coral growth and survival. This paper investigates previous research on coral restoration within the Caribbean region to establish which methods are most commonly used and of those, which have produced the best hard coral survival rate. The information is gathered through a regional meta-analysis of studies with set criteria that have quantified the effects of assorted coral restoration methodological approaches. In addition, the study monitored a recent coral restoration initiative in Barbados succeeding the outplanting of *ex situ* grown coral micro-fragments and their further development on the local coral reefs. The meta-analysis showed increased survival rate in fragments outplanted following a nursery phase, which is coherent with the findings of the case study of this thesis. Further survival was absent in the case study.

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I would like to show my appreciation to my supervisor, Professor David Smith for his continuous patience and guidance throughout my research and writing. For always making me feel encouraged. To Emma C. Revill and Corinne B. Whitby for always having my back. Susan Mahon, as the manager of Coral Restoration Alliance, Barbados (CORALL), for letting me be a part of the project. To Zoë Lisk for answering my endless amount of questions. Nikki Hassell and Tracy Phillips for welcoming me onto the project and for their continuous help along the way. To Renata Goodridge and Hazel Oxenford for spending time and energy on me and giving me guidance when I needed it the most. To Lisa Carne and Austin Bowden-Kerby for making time to answer my questions and giving me advice on different approaches to coral restoration. To Holly Drew and Nadeshinie Amanda Parasram for friendly support and advice. To Hayley Delemere and Sarah Beach for being incredible friends, always ready to listen. To my Barbadian family, Rosita, Sheika, Tameika, “Uncle” and Simron Browne for welcoming me into their home and lives.

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Abbreviations

AP	*Acropora palmata*
CZMU	Coastal zone management unit
CORALL	Coral Reef Restoration Alliance Barbados
COO	Corals of opportunity
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
GDP	Gross national product
GESAMP	Group of Experts on the Scientific Aspects of Marine Environmental Protection
GS	Google Scholar
HC	Hard coral
ID	Identification
ICZM	Integrated coastal zone management
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
MOTE	Marine Observation
MPA	Marine Protected Area
MDL	method detection limit
nMDS	non-metric Multidimensional Scaling
POM	Particulate organic matter
PP	*Porites porites*
PS	*Pseudodiploria strigosa*
RC	Rock
RB	Rubble
SD	Sand
SE	Standard error
STD	Standard seawater solution
IDB	The Inter-American Development Bank
RRN	The Reef Resilience Network
TSS	Total suspended solids

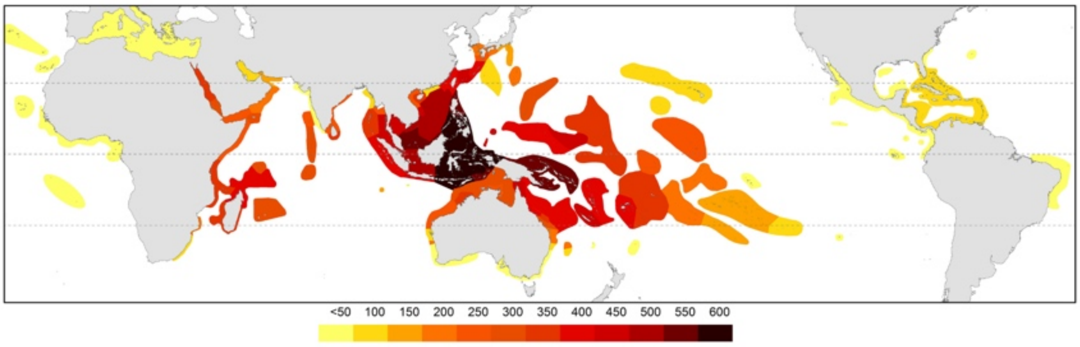
UNCLOS United Nations transcribed the first Conference in the Law of the Sea

WoS Web of Science

Chapter 1: Introduction

The three-dimensional underwater structures constructed by numerous entwined polyps of reef-building corals (Scleractinian) connected by an ever-expanding skeleton and assisted by the symbionts make up what we refer to as our coral reefs. The coral polyps comprise a symbiotic relationship with dinoflagellates commonly referred to as algae. The symbiont inhabits the gastrodermal layer of its hosts (Muscatine and Cernichiari 1969, Muller-Parker et al. 2015) in exchange for a large proportion of the nutrients produced by the algae through photosynthesis. The coral polyps are highly dependent on this symbiosis as they themselves are only capable of obtaining between 10-20 % of their daily nutrient requirements (Muscatine 1990). The symbiosis requires an environment with a steady supply of light penetrating the water column (Falkowski et al. 1984). Studies estimate that the world's coral reefs cover < 0.1 % of the ocean surface (Reaka-Kudla 1997, Spalding and Grenfell 1997) generate an important ecological system that supports a quarter of all known marine species (Mulhall 2009). This makes coral reefs the most species rich per unit compared to any other marine environment (Knowlton et al. 2010). The coral reefs produce an inimitable foundation for sustainable marine life and consequently function as an incomparable source of food and income for associated stakeholders. Reef related fishery has a global value of US$6.8 billion per year with a regional value in the Caribbean of US$395 million (Burke 2011). Furthermore, in the updated version (Costanza et al. 2014) of the 1997 study that then estimated the annual value of global coral reefs at US$375 billion (Costanza et al. 1997), this figure was increased, although the coral reef biome area was estimated to have declined by -34/ha from 1997 – 2011, from ~ US$8000/ha/yr. in 1997 to ~ US$352000/ha/yr. in 2011. The substantial change in value was partly due to the additional studies since the first paper had been published which highlighted the importance of coral reefs in regard to coastal erosion, storm protection and recreation. Additionally, coral reefs have, in recent decades, proved beneficial to the pharmaceutical industry by being a prominent source of useful substances (Glaser and Mayer 2009). Through adequate funding and research, the marine systems, coral reefs included, may prove significant in future medical manufacturing in relation to diseases such as cancer, AIDS and general inflammation (Bruckner 2002, Erwin et al. 2010, Leal et al. 2013), by providing useful ingredients.

Unfortunately, coral reefs are rapidly declining worldwide mainly due to climate change and anthropogenic activity such as unsustainable fisheries, increased marine pollution and coastal development. Almost 19 % of the original global coral reefs have been lost (Normile 2009) and a further 15 % could be lost within the next couple of decades (Wilkinson 2008). Coral polyps require a specific range of salinity, temperature, light and nutrient levels, conditions which place coral colonies largely within a restricted environmentally conditioned band within the 30th north and south longitude of the Earth's equatorial plane. Studies have shown a further congregated



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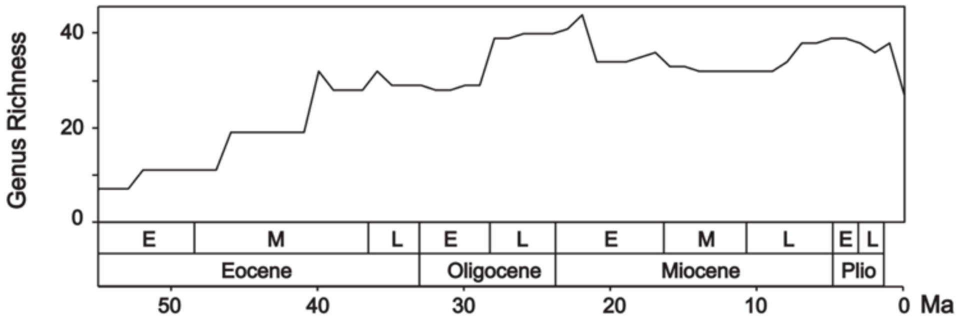
range of species abundance across regions (Fig. 1.1) with the Indo-Pacific region having the highest level of abundance and the Caribbean Basin the lowest (Plaisance et al. 2011, Veron et al. 2015).

1.1: Caribbean and Western Atlantic Basin Reefs

In a study intending to uncover the species diversity of Caribbean reefs during the Late Cenozoic era, a combination of chronostratigraphic methods, including biostratigraphy, paleomagnetic and strontium-isotope analyses, were completed (Budd 2000). Identification was established by gathering standardised morphological characters founded on morphometric and molecular data (Forsman et al. 2009) to revise species diversity from earlier periods. The gathered data (Budd 2000) suggest an increase in genus richness since the early Eocene, with two recessions until the present time (Fig. 1.2). Results show that the coral species richness has remained relatively steady over time. Further, patterns suggest that the Caribbean was not limited in population dispersal by a predestined amount of species (Aronson and Precht 1997), but rather there was a biogeography limitation (Rosen 1988). A rise of the Isthmus of Panama created a



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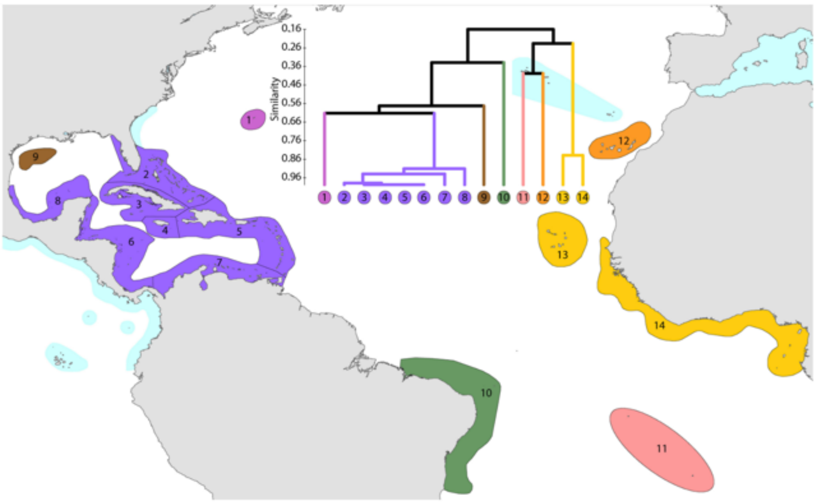
complete marine barrier during the late Pliocene (Muss et al. 2001), separating the Caribbean Sea from the Indo-Pacific (Briggs 2005). A diversity centre developed in the southern part of the Caribbean Sea (Fig. 1.3), with branches of species reaching the waters towards North



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America in the north, and south towards South America. A study on regionally isolated populations (Fig. 1.4) of the hard-coral species *Acropora palmata* (Elkhorn Coral) confirms this trend by analysing genotypic arrangements (Baums et al. 2005). Coral colonies were sampled and genotyped from 44 reefs throughout the Caribbean. The study showed a

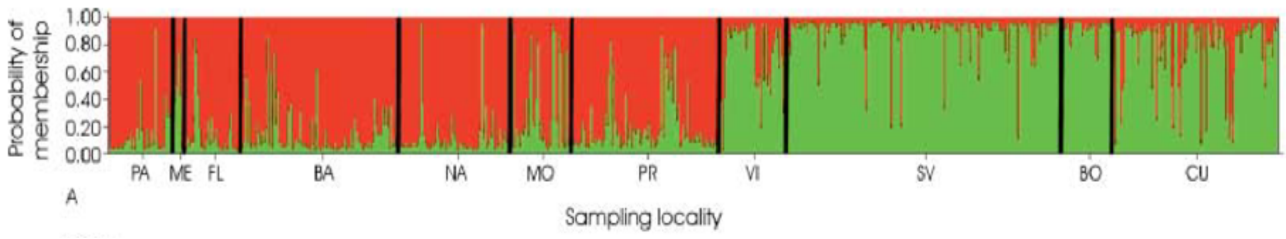
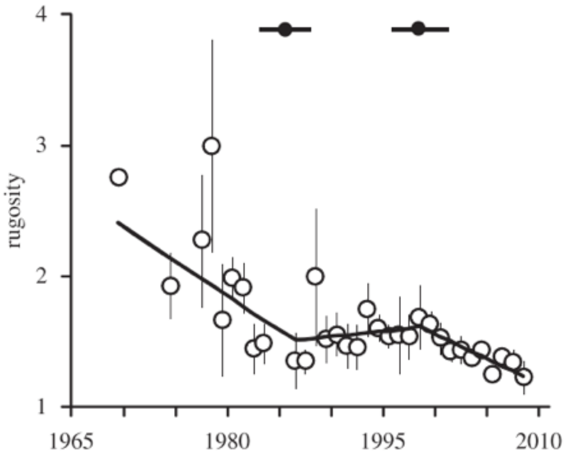


Fig. 1.4: Geographical presentation showing two genetically differentiated clusters of the *Acropora palmata* population located in the Caribbean. The western branch include Panama (PA), Mexico (ME), Florida (FL), the Bahamas (BA), Navassa (NA), Mona Island (MO) and Puerto Rico (PR) and the eastern the US Virigin Islands (VI), St Vincent and the Grenadines (SVD), Bonaire (BO) and Curacao (CU) (Baums et al. 2005).

split population of two main differentiated clusters distinguishing between the southern and northern areas. Although this study shows a variation between significant areas, it also suggests no contemporary input to the gene pool from other areas. These studies support the general results and imply a consistent but restricted genetic pool with a high level of genetic similarity in the Caribbean region. In another study investigating samples from the hard coral species *Acropora cervicornis*, collected from 160 individuals in 22 populations across 9 regions (Vollmer and Palumbi 2007), the limited genetic flow over a greater as well as a smaller spatial scale within the Caribbean was confirmed. This may explain the drastic decline in live coral coverage during the 1970s – 1980s within the Caribbean and western Atlantic Basin, a drop of almost 80 %. Studies show that fitness and heterozygosity are strongly corelated; a limitation in the genetic pool would likely limit the ability to adapt to the various changes during the anthropocene (Reed and Frankham 2003). For the decline of live coral coverage throughout the Caribbean, evidence points to an unfortunate array of events. A substantial part of the decline was largely caused by a combination of four incidents; 1) increased pollution within the area, 2) rise in nutrients and pathogens causing an increase in White Band Disease (WBD) and decrease of the black sea urchin (*Diadema antillarum*), 3) higher levels of fishing and 4) extreme weather events. First, an increase in water pollution (Aronson and Precht 2001, Gardner et al. 2003, Hoegh-Guldberg et al. 2007) caused a rise in the occurrence of WBD which lead to an 80 - 90 % decline in some coral species. In 2008, the decline placed the two species of branching acroporids *A. palmata* and *A. cervicornis* on the International Union for Conservation of Nature (IUCN) (Kline and Vollmer 2011) as critically endangered. Concurrently, there was a pathogen induced (Lessios et al. 1984) decrease in the numbers of the black sea urchin (*Diadema antillarum*). This combined with the long-term exploitation of herbivorous fishes in the region

(Hughes, 1994) meant that algae could thrive and eventually this caused a phase shift to macro-algae dominance. This is a state that not only causes the smothering of current coral colonies which eventually die-off, but also may limit coral recruitment leading to a stable state of algae dominance (Mumby 2009a). To introduce all these events, in 1980, an extremely powerful Cape Verde category 5 hurricane passed through the Caribbean, wiping large areas of hard corals. And again, the two most affected species were *A. palmata* and *A. cervicornis*, meaning that the more robust bouldering species had a chance to expand. Thus, these events combined not only to lead to the reduction of coral coverage, but also affected the population composition causing a change in reef morphology and eventually a degradation of reef complexity (Gladfelter 1982, Weil 2004). One systematic review of 464 records from 200 reefs across the Caribbean suggests a flattening of the reefs during the period 1969 – 2008 (Alvarez-Filip et al. 2009). A rugosity index was calculated from average complexity per site per year at three different depths (< 6 m, 6 – 20 m, and > 20 m) of five subregions within the Caribbean (Central America, South America, Lesser Antilles, Greater Antilles, and North Atlantic).

In a study considering change in community composition between lagoonal and offshore settings prior to the 1960s, it was shown that although revealed within both settings, the rate at which it occurred was much greater in the near shore environment (Cramer et al. 2012). Comparison of the terrestrial and aquatic timeline of the area suggests the difference in rate of change could be explained by 1) the effects of the hurricanes and 2) the increase in regional population (Guzmán et al. 2006) causing elevated levels of sedimentation, nutrients and pollutants in near shore waters. In relation, a former paleoecological analysis of the reef framework (Lewis 1984) surrounding Barbados had consistently displayed declines in *A. palmata* already following the European colonisation. By the early 1900s, a decline in coral community and structure was well underway (Pandolfi et al. 2003), suggesting a link to the changes on land. To further highlight the relationship, data were collected from coral reefs along the coast of Jamaica from the early 1950s till the mid-1990s (Hughes, 1994). During the first two decades, the results showed relatively health with a high level of coral cover. However, in the years to follow, the coral coverage rapidly declined from almost 80 % to between 0 % and 20 %. (Fig. 1.5).



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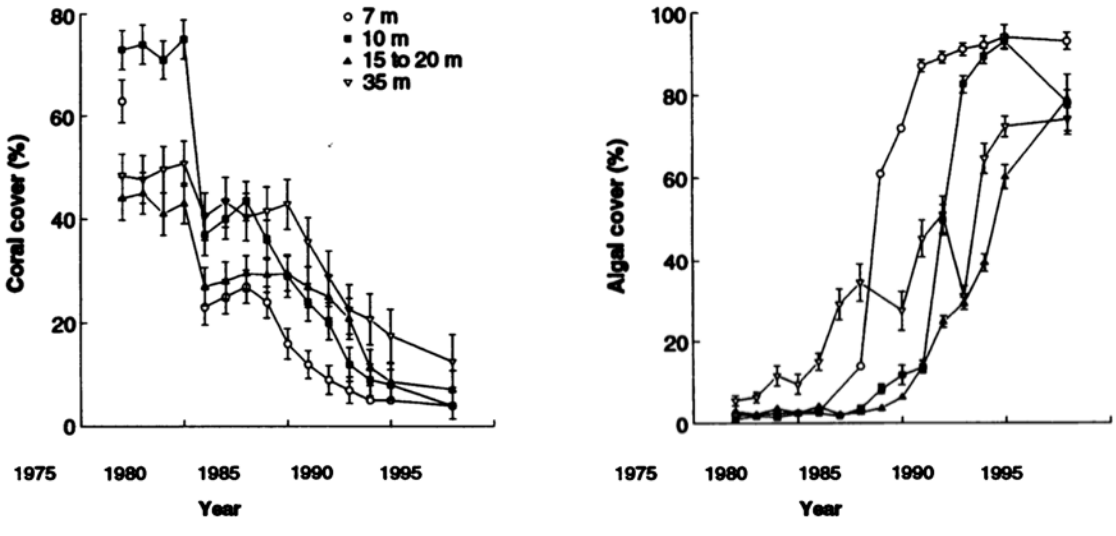


Fig. 1.6: Data from Jamaica, showing the decline of coral coverage during the period from 1975 to 1995. The same period shows an increase in algae cover (courtesy of Hughes, 1994).

Meta-analyses (Gardner et al., 2003; Côté et al., 2005; Gardner et al., 2005) of data collected from 1980 – 2001 concur that there has been a general 80 % reduction in coral coverage throughout the Caribbean region (Fig. 1.6). One study focused on the impact of hurricanes within the region by contrasting decline of coral coverage loss at impacted and non-impacted sites (Gardner et al. 2005). The immediate decline of coral coverage at impacted areas was ~ 17 % by the following year. This decline occurred at a faster rate in subsequent years in non-impacted areas, although the long-term decline of impacted and non-impacted sites was similar throughout the 1980s. During the 1990s the non-impacted areas declined at a faster rate than the sites impacted by hurricanes (Gardner et al. 2005). These results concur significant stressors impacting the coral coverage, or interestingly, as argued in the paper, perhaps the affected areas experience reduced fishery and generally reduced anthropogenic pressure due to the impact the hurricanes caused on land and in local communities.

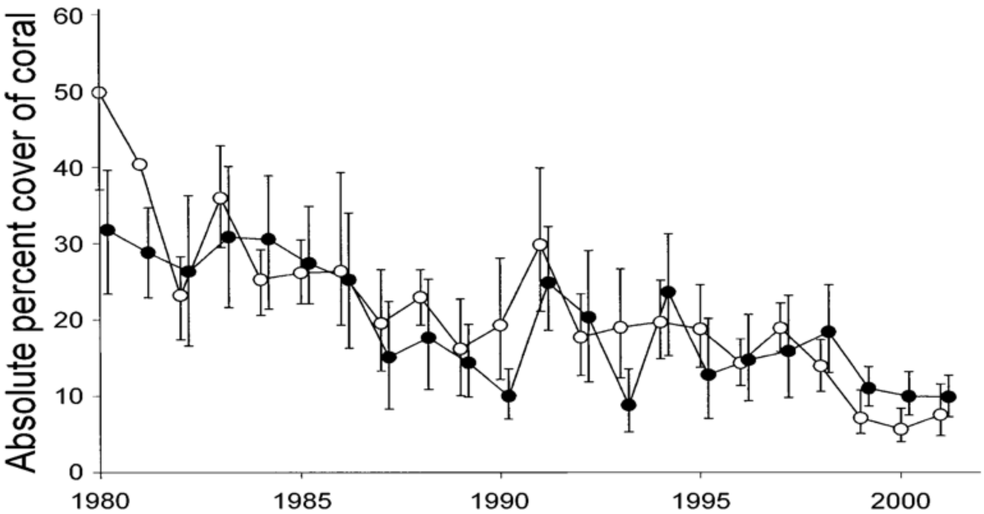


Fig. 1.7: The total percentage of coral cover across the Caribbean Basin during the years from 1980 – 2001, extracted by meta-analysis from 63 separate studies (Gardner et al. 2005). The solid circles represent sites impacted by hurricane, whereas open circles represent non-impacted sites.

Due to the topography of the Caribbean Basin, the commercially exploited fish population is often associated with rocky or coralline ground within the Caribbean, thus most common fishing methods exclude trawling and nets (Munro 1983). There are approximately 180 coral reef associated species, expanding across 10 families which are considered the most economically dominant species; these include Acanthuridae, Scaridae, Holocentridae, Pomadasyidae, Mullidae, Chaetodontidae, Balistidae, Lutjanidae, Serranidae, Carangidae, and Carcharinidae. Traps, hook and line, small nets and spearfishing are, due to the ecological topography, financial mean and fish species caught, the most common method of fishing in most parts of the Caribbean. Certain methods, such as the traps, have proven to be particularly detrimental to marine life and parrotfish through their persistence of material and efficiency to catch targeted species, thus reports recommend strict regulations on these methods (Bellwood et al. 2004). Jamaica can be mentioned as an example of how the environmental conditions led to a crash in fish populations (Hardt 2009). During the period from 1920 – 1950 an increase in local population and tourism meant the demand for fish grew stronger and the use of traps caused high pressure on the local fish stock. The combination of higher demand, improvement in technology and cheaper supply, meant an increased use of chicken wire for the structure of the traps. These improvements made the traps last longer and they became more proficient at holding larger individuals, thus leading this method to become the most dominant way of fishing within the area surrounding the coastline of the island (Hardt 2009). By the 1970s, the numbers of fishing canoes had expanded to between twice and three-times sustainable levels (Munro 1983). Consequently, a change in the taxonomic arrangement of fish species was triggered. In particular, a depression in large predatory species was recorded. A decade later, a study revealed (Munro 1983) an 80 % decline in fish biomass collected through fishing traps on the fringing reefs on the north coast of Jamaica. As motorised fishing boats became more accessible, fishermen expanded to the southern side of Jamaica, subsequently repeating the modification and decline of fish composition.

In relation to the expanding coastal urbanisation, coastal waters have become more eutrophic, largely due to river-borne agricultural run-offs containing nitrogen and phosphorus (GESAMP 2001). This has led to a seafloor with areas depleted of oxygen, increased levels of nutrients and sedimentation along with other pollutants like *Escherichia coli* and heavy metal from terrestrial discharge (Fabricius 2005). Some studies show a decrease of up to 50 % in calcification (Marubini 1996, Ferrier-Pagés et al. 1999) with an increase in either dissolved inorganic nitrogen (DIN) or phosphorus (DIP). Other studies show only little effect (Kinsey and Davies 1979, Koop et al. 2001). Most studies show an increase in density of zooxanthellae in the presence of increased DIN, whereas there is no relationship with increase in DIP (Muscatine et al. 1989). When nitrogen is in excess, the zooxanthellae will uptake this to enhance growth, hence the increase in zooxanthellae density with increased DIN (Marubini and Davies 1996). The decreased coral growth trailing an increase in DIN may be explained by the amplified density of zooxanthellae, and their uptake of the available CO_2 for photosynthesis. The coral becomes deprived of the CO_2 otherwise used for calcification (Muscatine et al. 1989, Marubini and Davies 1996, Marubini and Atkinson 1999). Although these experiments show the influence of DIN and

DIP on coral health, it has to be considered that they were all conducted *ex situ* at elevated levels which the natural environment would most likely never experience. Subsequently, although these observations can indicate the effect elevated levels of nutrients may cause, they will not draw an exact picture of the *in-situ* response, where water currents, rainfall, uptake by phytoplankton, bacteria and the general food web might significantly influence the result. In contrast to DIN, an increase in particulate organic matter (POM) is thought to promote growth in both host and symbiont, which explains the omitted increase in the density of zooxanthellae (Muscatine et al. 1989, Marubini 1996). The particulate matter is often suspended in the water column, consisting of bacteria, plankton, and detritus. Fine particles can be used by a variety of benthic organisms, including coral polyps. Feeding saturation is species specific and varies from low to moderate uptake (Veron 2000). Enriched levels of nutrients may originate from agricultural outruns and, additionally, poorly locally managed runoff from golf courses and sewage discharge. For Latin America and the Caribbean Sea, almost 86 % of sewage water is discharged untreated (Burke and Maidens 2004, Anon. 2006) (Fig. 1.8). Adding to the increased levels of nutrients, human pathogens, industrial chemicals, oils and greases (GESAMP 2001), all presenting further stressors to the surrounding coastal coral reefs, are all included in the runoff and discharge (Patterson et al. 2002, Sabdono 2009, van Dam et al. 2011). Although little field research has examined the relationship between sewage water and coral health, one paper from 2015 (Wear and Thurber 2015) looked at eight observational studies conducted on coral reefs which were exposed to a considerable volume of sewage

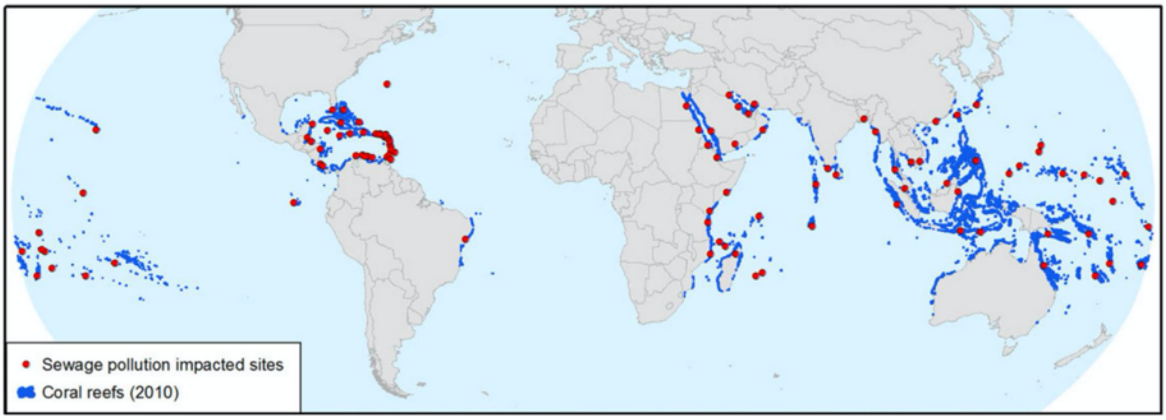


Fig. 1.8: Global distribution of coral reefs (blue) with documented coastal sewage pollution (red). The map displays the high occurrence of sewage outrun within the Caribbean Basin (Courtesy of Wear and Thurber, 2015).

discharge. The paper compared neighbouring reefs of similar environmental conditions with only little or no known exposure to sewage water. In seven out of eight cases the coral reefs that experienced sewage water showed signs of negative impact. One reef showed no correlation. Although the author warns of a study design without before/after control, it is also suggested that the result may inspire the hypothesis of a correlation between sewage outrun and coral reef health.

It is estimated that ~ 36 % of the Caribbean coral reefs are located less than 2 km from developed landmasses (Burke and Maidens 2004). Apart from the increasing risk of exposure to terrestrial runoff, the close proximity between coral reefs and coastal development presents a varied array of threats to coral health. Poorly managed coastal construction may impact the coral reefs directly through dredging, land reclamation, mining, destruction of coastal habitats, combined with further runoff from the construction site itself. But even more so there may be an indirect impact that follows the removal of mangroves and sea grass along the coastline (Burke 2011). These trees and plants, with their massive root systems, preserve and filtrate the sedimentation arriving into coastal water from upstream water bodies. When these habitats are removed, the sedimentation is often released directly onto the coral reef where it may cause harm by smothering, limiting sunlight and by carrying pathogens and harmful chemicals (Burke 2011). It is estimated that 29 % of coral reefs within the Caribbean region can be considered in a high-risk threat category (Fig. 1.9). These threats include the sedimentation from upland deforestation, elevated nutrient levels from poorly managed agriculture, coastal development, pollution and overfishing (Burke 2011). These considerations do not include the impact of ongoing climate change and what effects it may bring to the area through an increased frequency and severity of weather patterns, such as hurricanes, elevated temperatures, ocean acidification and the rise of sea levels. In the past 60 years, coral restoration has been on its way but often with less efficient results due to a combination of limited financial support and an interlinked complexity of the matter requiring skilled communication and collaboration. A recent study (Hein et al. 2019) extracted data from face-to-face interviews conducted transnationally across four well established coral restoration projects. The interviews obtained information on five main subjects: 1) demographics, 2) experience with coral reefs, 3) benefits and limitations of coral restoration effort, 4) financial aspects, and 5) overall opinions on the coral restoration projects. In total, 120 participating projects.

Fig. 1.9: The estimated level of threat via coastal development throughout the Caribbean. High level threat areas are marked with red and low-level threats with blue. The level of threats was calculated by the distance from coral reef to developments, ports, harbours, airports and tourist areas in combination with population density, population growth, and visitor numbers. Marine protected areas (MPAs) were counter calculated as a mitigating factor (courtesy of Burke and Maidens, 2004).

The interviews obtained information on five main subjects: 1) demographics, 2) experience with coral reefs, 3) benefits and limitations of coral restoration effort, 4) financial aspects, and 5) overall opinions on the coral restoration projects. In total, 120 participants responded to the developed questionnaire, and 11 mutual subjects became evident; the echoing beneficial themes related to: 1) socio-cultural benefits, 2) ecological benefits, 3) project appreciation, 4) positive experience and 5) economic benefits. Whereas the six recurring themes were: 1) technical limitations, 2) management limitations, 3) ecological limitations, 4) restoration limitations, 5) staff limitations and lastly 6) legislative limitations. The study concluded that while coral restoration experiences present many challenges and outcomes often appear inadequate, these projects, with the correct management, improved local involvement and funding may present a powerful tool to support coral reefs from negative impact.

1.2: Coral Conservation

The decline of important species throughout the Caribbean has instigated raised aspiration to conserve and enhance rectification (Linton and Warner 2003). A particular effort is often aimed at taxa such as *Acropora spp*., *Dendrogyra spp*., *Orbicella spp*., all categorised as "endangered" in the US Endangered Species Act. The effort is directed through increased regional propagation and restoration projects (GESAMP 2001), while drafting species specific recovery plans (e.g. *Acropora palmata*, *Acropora cervicornis*) (National Marine Fisheries Service 2015). In areas of low anthropogenic pressure, a damaged coral reef is capable of retrieval through its own resilience (O'Leary et al. 2017), a supplemental testimony of the importance of environmental regulatory management when aspiring to successful restoration. However, if the damage to the reef is more severe, an operational approach may be necessary. Ecological restoration is the act of assisting a damaged or degraded ecosystem in its recovery. A comparison of biodiversity and complexity versus biomass and productivity is advised to establish the most beneficial practice of restoration (Edwards and Gomez 2007).

Coral restoration can generally be considered to include two means of tactics: a passive and an active restoration course (Rinkevich 2014). In the early years of coral restoration, these methods were often kept separate, but recently the necessity for combination has become increasingly more acknowledged (Rinkevich 2008, Shokry and Ammar 2009, Hughes et al. 2017). Passive management will include the improvement of any disorder of the environment that originally contributed to the damage, e.g. exploitive fishing, discharge from sewage systems and agriculture, or coastal development (Edwards et al. 2010). The active method is the physical assistance, e.g. coral reef transplantation (Edwards and Gomez 2007). Full recovery of the reef to pre-damage conditions is not often achieved by restoration. Restoration simply brings the reef to a rehabilitated state and in some circumstances, has led to a replacement system (alternate ecosystem) (Edwards and Gomez 2007, Lirman and Schopmeyer 2016).

Passive Restoration; Marine Management and Legislation

The success of coral restoration is predisposed by many variables of both abiotic and biotic characteristics often introduced indirectly or directly by anthropogenic behaviour. But recently, greater attention has been given to the less utilised section of coral restoration (Anthony et al.

2015, Thia-Eng and Bonga 2018), the passive part that includes governmental support and management through legislation and enforcement. Present marine management and conservation research encourages an elevated attentiveness to regulating and monitoring multiple aspects that may affect the marine environment (Levine et al. 2015, Weijerman et al. 2016, Anthony et al. 2017) and not focusing on, but solely embracing the contemporary topic of climate change (Bellwood et al. 2004, Jackson et al. 2014). The argument is that by focusing excessively on climate change, and the often-associated coral bleaching, governments and marine managers forget local conditions, which could be improved more easily. Either that, or they are simply left overwhelmed by the problem (Mumby et al. 2017). Due to the challenges in achieving a successful result solely from active coral reef restoration, projects may be launched as an integral part of a larger coastal management framework.

From 1956 – 1958 the United Nations transcribed the first Conference in the Law of the Sea (UNCLOS I), restructured later in 1973 (UNCLOS III). A demand to construct present international regulations rose with the technological development of fishing fleets that allowed them to travel further ashore than the previous “Freedom of the Seas”, which, in general, allowed nations to work within a three nautical miles radius of their country. Due to the many nations participating, the convention was only reinforced almost 20 years later, in 1994. The convention includes more than 160 nations plus the European Union. Most countries surrounding the Caribbean Sea have signed and agreed to the Convention. One exception is the United States of America, which only signed the agreement but does not approve due to objections in regard to Part XI, relating to agreements regarding the continental shelf e.g. drilling and exploitation of resources. Another exception is Venezuela which has neither accepted nor signed the Convention or any agreement. Additionally, Colombia signed the Convention, but not the agreement.

Further, in 1992, a meeting was called by the United Nations. The agenda was to corporately structure a transcript advising governments how to lead their individual country to a sustainable 21st century. The agenda, referred to as Agenda 21, consists of four sections. Section one covers the social and economic dimensions by addressing poverty, consumption, human health and how to integrate the environment and development in decision-making. The second section aspires to conserve and manage resources for development by directing awareness to deforestation, sustainable agriculture, the protection of the seas, sensible management of toxic waste and maintaining high quality of water resources. Lastly, the third and fourth sections address equality between groups of human populations, and how to implement the suggestions of the previous three sections. Chapter 17 focuses on all seas, including any related water bodies. Section B, part 17.18 states "*Degradation of the marine environment can result from a wide range of sources. Land-based sources contribute 70 per cent of marine pollution, while maritime transport and dumping-at-sea activities contribute 10 per cent each*" (Agenda 21 1992). This quotation demonstrates the link between terrestrial planning and management with the health of marine environments. The agenda further states (part 17.22) that in accordance with UNCLOS III, states should commit themselves to preventing, reducing and controlling the degradation of marine environments in order to sustain the livelihood provided by the oceans. It is from these parameters that the idea of integrated coastal zone management (ICZM) grew. Agenda 21 suggests that the management involves gathering information from observational data, expert statements and research, whilst sharing that information with developing countries. In addition, education and training of local fishermen, scientists, technologists and managers should be promoted within the ICZM to integrate public awareness. In relation to the formulation as per Agenda 21 and with an extension derived from the Intergovernmental Panel on Climate Change (IPCC) of 1995, the ICZM is defined as a comprehensive assessment which includes a set of objectives to plan and manage coastal resources whilst taking into account traditions, cultural and historical perspectives with all the conflicting interests and uses. One paper (Suman 1998) summarises the ICZM as the "*continuous process of adaptive management of coastal areas to achieve the goals of sustainable development…*" and suggests that in the naming of the assessment lies an acceptance of varied coastal ecosystems, a coordination of management to consider and include all stakeholders, whilst also including expert evaluation for institutional implementation of the ICZM. It is a tool to combine stakeholders within the development and management of coastal areas (Post and Lundin 1996) in such a way that all benefit in terms of their interests and responsibilities whilst maintaining a sustainable relationship to socio-economic activity and the coastal environment. In order to control the threats placed upon the coral reefs, certain activities, such as fishery, tourism and coastal developments, should be measured and thoroughly attended to throughout the region (Post and Lundin 1996, Bellwood et al. 2004). In order to provide decision makers with substantial evidence of progress and challenges about where to focus the ICZM, scientific, social and economic research (Linton and Warner 2003) should be used to identify any indicators of change in state of status. The indicators are used to continuously supervise conditions for assessments of impact and requirements of management whilst measuring the efficiency of an existing ICZM. Changes in existence/non-existence, state or abundance of bioindicators and living components of human surroundings provide useful information of its current health status. By monitoring changes to the environment in relation to the qualitative and/or quantitative changes in a bioindicator, a source may be identified.

Designating a Marine Protected Area (MPA) is an example of a method for passive coral restoration where, according to IUCN "*Any area of intertidal or subtidal terrain, together with its overlying waters and associated flora, fauna, historical and cultural features … has been reserved by legislation to protect part or all of the enclosed environment*" (Kelleher 1999). Depending on enforcement and local understanding, an MPA will often request limited to no traffic and/or fishery. The idea of an MPA is that the species populating the confined zone will thrive, reproduce and "spill over" into surrounding waters. The Reef Resilience Network (RRN) divides an effective MPA management into five principles (Green et al. 2014); 1) Effective management, 2) Representation and replication, 3) Critical areas, 4) Connectivity and finally 5) Socioeconomic criteria. All of these principles represent a reduction of destructive behaviour within the protected area by limiting daily activity, improving coastal development and lessening local pollution. Whilst planning the representation and replication of an MPA, many factors should be included. RRN highlights specifically the importance of considering: 1) Biodiversity composition, 2) Biogeographic, geographic, and environmental gradients in habitat and species composition, and finally 3) Ecosystem integrity. These factors are specifically emphasised in order to conserve and represent all commercial key species throughout the trophic cascade. Critical area ecosystem functions are considered areas that naturally exhibit resilience towards environmental stressors. These are habitats well known as spawning aggregations, nursery grounds, or areas that in other senses are linked to developmental habitats for a variety of life stages. Furthermore, migration corridors can be considered as critical areas by providing a flow of larger animals between feeding and breeding grounds. Lastly, areas that provide habitats for rare or threatened species should be included in critical areas. These critical areas are highly associated with the next principle: connectivity. When composing MPAs, it should be considered whether to expand to a network of MPAs, which will provide the flow between sanctuaries of safe havens and ensure the protection of ecological and functional productivity. These networks not only ensure the dispersal of larvae recruitment and settlement between areas, but also increase ecological as well fishery "spill-over" (Stobart et al. 2005, Forcada et al. 2009, Di Lorenzo et al. 2016). MPAs, or marine reserves, may be one way of regulating and monitoring a particular site of interest, but some critical voices have argued that these marine zones do not always aid the coral coverage and health, and policymakers should consider a more wholesome approach (Aronson et al. 2006). Other studies counter argue the statement by showing an increase in coral coverage following an increase in fish population within an MPA (Fig. 1.10) (Mumby and Harborne 2010, Mellin et al. 2016). And other studies add that these MPAs will show some improvement but require an adequate long-

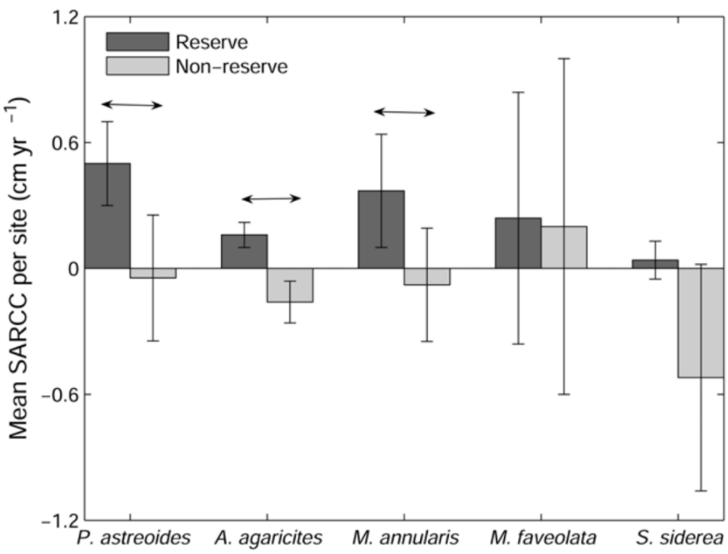


Fig. 1.10: Results of size-adjusted rate of change of cover (SARCC), inside and outside the marine reserve Exuma Cays Land and Sea Park, Bahamas from 2004-2007. Horizontal arrows shows the significant differences (one-tailed t-test P < 0.05) between inside and outside the reserve (Mumby and Harborne 2010).

term approach which, in return, will bring measurable benefits per year (Selig and Bruno 2010, Costello 2014), although positive results may fail to appear without adequate management due to illegal fishery and inadequate sizing (Edgar et al. 2014). Research may show varied data with regard to the effectiveness of MPAs, but most studies concur that 1) local management on the surrounding anthropogenic impact and an efficient administrative approach needs to be in place for an MPA to succeed. Although coral reefs may be able to recover following severe damage (Stobart et al. 2005), when situated in supporting conditions (Gilmour et al. 2013), this ability may be undermined when anthropogenic disturbance exceeds the rate of recovery (Crabbe 2010).

Active Restoration: Methods of Coral Restoration

The most commonly used method of coral restoration is the transplantation of coral fragments, a method that makes use of the natural ability of coral colonies to asexually reproduce (Highsmith 1982). The method can be split into three different categories: 1) direct transplantation, 2) an intermediate phase or 3) micro-fragmentation. All include the collecting, transporting and outplanting. The collection will often be done by harvesting coral fragments off neighbouring coral reefs using a hammer and chisel or cutters. Some studies have shown a higher survival rate when acquiring larger fragments, ideally 10 – 30 cm in size (Bowden-Kerby 2001). The harvest should not exceed > 10 % of the donating colony as studies have shown a detrimental effect above this threshold. Other methods of harvest can be either by collecting fragments from already established nurseries or by collecting fragments that have already been displaced from their colony by environmental disruption, referred to as "corals of opportunity" (Musco et al. 2017, Schopmeyer et al. 2017). Transporting the fragments may be done, over short distances, by scuba divers underwater carrying the collected corals in crates or baskets, and, over longer distances, out of water with minimal exposure to air and sun. Studies have shown reduced survivability when transportation exposes the fragments to air and sunlight for more than 2 hours (Harriott and Fisk 1987). A sturdy attachment to the underlying substrate reduces loss of fragments and further encourages self-attachment. The three most common methods of attachment are by underwater epoxy, cable ties or cement (Sujirachato et al. 2013). Studies which have investigated the different attachment methods seem to agree that there is no better method. Instead the best method differs according to species, lifeform, substrate, financial means and a hardy installation (Edwards and Gomez 2007, Edwards et al. 2010, Sujirachato et al. 2013). Further, considerations should be given to outplanting arrangements as some experiments have successfully shown a change in growth and survivability relative to neighbouring morphology and species (Boström-Einarsson et al. 2018).

Direct transplantation is the most common and oldest method of active coral restoration. The aim of the method is primarily to encourage recovery of a damaged reef. The method is cost effective as it requires no middle step for the coral fragment to recuperate. Fragments harvested from donor colonies within the area are transplanted directly onto the reef. But the method places a certain level of stress on both the donor colony and the harvested fragments, thus requiring that the conditions within the environments for harvesting have stabilised and reached an ambient point encompassing the best conditions for recovery following the meddling. Further, growth and survivability on the acquired fragments will depend on fragment size and colony health prior to the transplantation. One study that compared (Edwards and Clark 1999) the demerits and benefits of direct transplantation reached the conclusion that the harm the method caused surpassed the benefits, and the reefs would gain more from a natural recovery if environmental conditions were normalised. They added, that direct transplantation should only be performed as a last course of action.

One method is inspired from terrestrial forest management and is also referred to as silviculture. It is a method that has been developed by recognising the ecological complexity of a well-functioning habitat and how long-term productivity depends on the ability to absorb and resist stressors without the loss of function. The method has been developed not to create an artificial tree factory, but rather to maintain the complex ecosystem (Perry 1990, Rinkevich 1995). Initially, to supply the coral nursery with first generation donors, fragments are collected directly from *in situ* donor colonies, collecting corals of opportunity or by collecting propagules from adult coral colonies held in captivity. Hereafter fragments are taken from these collected donors. The method, often referred to as coral gardening, prevents stress on existing coral reefs by instead harvesting from the donor fragments maintained in coral nurseries. Th first step of coral gardening involves placing the coral fragments in an artificial habitat, often including a constructed PVC tree, tables, mesh, nets or rope on which the fragments may grow, located in sheltered areas to prevent damage. Fragments will often be preserved by removal of algae growth, or simply kept under conditions that limit the occurrence of algae and predation from fish and invertebrates. The coral nurseries act as an intermediate phase to allow smaller fragments to grow under managed conditions before being outplanted at chosen *in situ* sites. The second step includes the outplanting of fragments onto degraded natural coral reefs.

Nurseries are placed in either *in situ* water nurseries in a sheltered environment, or *ex situ* in a laboratory-based nursery. The field-based nurseries are often monitored and maintained in favourable conditions on substrate-based structures, such a cement blocks (Marubini 1996), fixed tables (Shaish et al. 2008), lines (Goergen et al. 2017), ropes or frames (Quinn and Kojis 2006), or they float mid-water on nursery trees and floaty tables (Shafir et al. 2006, Shaish et al. 2008). Fixed nurseries are kept stationary and elevated one to two meters above the seafloor, often using fixed modular trays or robes, species depending, extended between metal or PVC bars (Bowden-kerby 2014). Trays are most commonly used for the massive or encrusting coral lifeforms attached to discs (Bowden-kerby 2014). The trays are spread across a PVC frame and tied onto lines. This method is most commonly used to support heavier slow growing species such as *Pseudodiploria Spp.* The rope method is typically reserved for the lighter fast-growing species, e.g. *Acropora spp*. This method has previously been used throughout the Indo-Pacific and the Caribbean on a limited budget with promising results. Ramets are inserted by untwisting and twisting the rope for every 10-15 cm (Bowden-kerby and Carne 2012), depending on the rate of growth. Land-based nurseries place the coral fragments in tanks where conditions are monitored and adapted to provide optimum conditions to secure best coral growth and survival. Although this method uses a gentle approach by limiting direct harvesting from coral reefs, the disadvantage may be reduced genetic diversity. Some researchers (Carne et al. 2016) suggest genotyping as a mandatory component of coral restoration to maintain a healthy coral reef that can cross-fertilise at transplantation sites.

Floating nurseries can be placed anywhere from shallow water to offshore deeper waters. For both types, the location should be in a low traffic area to reduce havoc caused by human activity. A mid-water nursery can be placed so that divers can access the nursery without damaging already existing coral reefs, whilst being kept at a distance from corallivores. It is recommended that these structures are kept at >20 m above a sandy seafloor. Further, it is recommended that any restoration trays are kept to a size easy for following transport. This method can be used for a larger amount of ramets. The nursery can be attached by either sinkers (concrete blocks) or an anchor, or a combination of both for extra stability and depending on the size of the nursery. For further stabilisation, buoys can be attached along the sides and in the corners of the nursery. The same structure can be used for larger (10,000 ramets/year) or smaller nurseries (<1000 ramets/year).

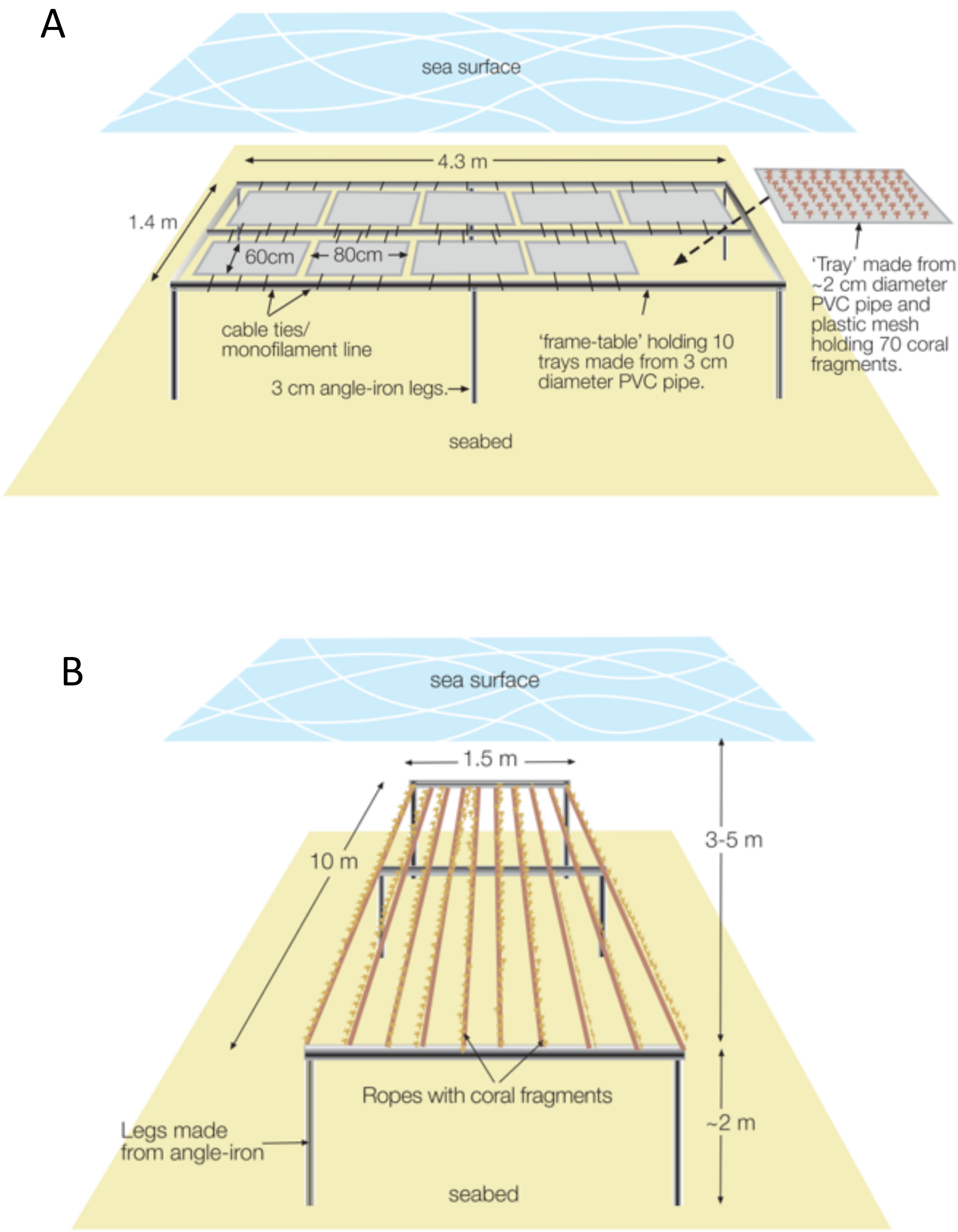


Fig. 1.11: Examples of nursery methods. **A)** Fixed table containing trays to hold <70 more dense and heavier coral fragments (Edwards et al. 2010). **B)** Table containing ropes for lighter species such as *Acropora Spp*.. The ropes are untwisted every 10 cm to hold one fragment (Edwards et al. 2010)

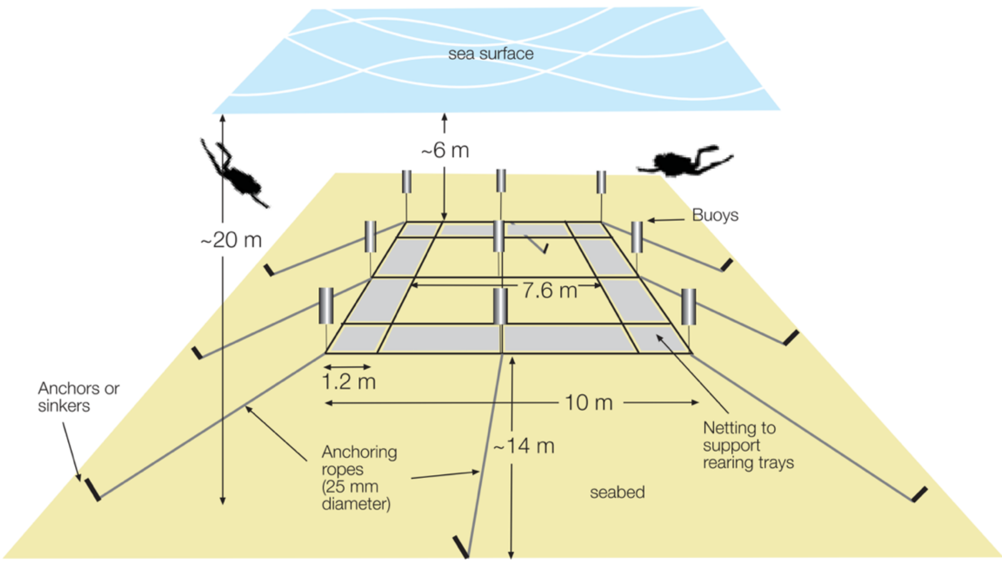


Fig. 1.12: The floaty nursery suspended mid-water by anchoring ropes. The nursery is kept mid water by the use of buoys and can be adapted according to tidal changes (Edwards et al. 2010).

The newest method in coral nurseries is Coral Tree Nurseries, designed to eliminate some of the challenges that the traditional methods experience. The fundamental approach is similar; smaller coral fragments are harvested from a “mother” generation within the nursery but hung from produced underwater trees. The trees are tied to a rocky seafloor or anchored to a sandy bottom, and the fragments are hung from the arms of the tree using wire or monofilaments. The tree is held up by a subsurface floatation device which can be adjusted for correct match to the water column (Nedimyer et al. 2011). This makes it possible for the nursery to flow with the water movement reducing the chances of damage to the nursery itself, or the fragments attached. The design of the structure also limits algae growth due to reduced surface area surrounding the fragments. This method is a long lasting, low-cost approach that has the further advantage of using materials that are readily available in most places. The method has also been proved to increase the growth rate of coral fragments due to the surrounding water flow and less energy consumed for hold fast action. Furthermore, the design resembles the floaty nurseries in being located mid water, thus reducing damage from predation and algae overgrowth when compared to the traditional methods.

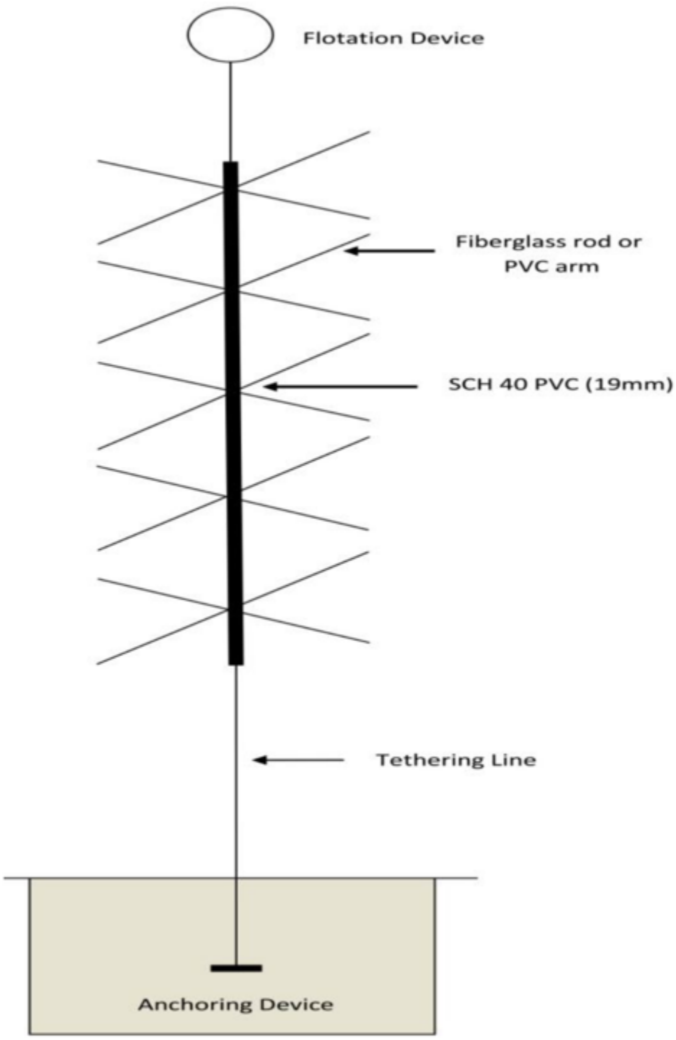


Fig. 1.13: The design of a Coral Nursery Tree with a device for anchoring to the substrate and floating device for buoyancy at the top of the tree, which makes the tree mobile to flow with water movement thus limiting damage from wave action (Nedimyer, Gaines and Roach, 2011).

Only a small section of research includes slow growing coral species. This has mainly been due to a combination of three reasons: 1) the particular lifeform associated with slow growing corals presents a more challenging structure to manipulate, since it has a planar thick skeleton, 2) the slow growth rate prevents a quick result which is often required for project which are financed short-term and 3) the focus has predominantly addressed, particularly within the Caribbean, the endangered Acroporids. But by neglecting other lifeforms, coral restoration may risk a less complex coral reef structure with reduced ecological diversity. A new technique, exploring the natural ability of familiar coral colonies to fuse, increases their collective size, and thereby the probability of reproduction and survivability (Raymundo and Maypa 2004, Forsman et al. 2015). This process occurs naturally when coral colonies are obstructed by physical conditions such as currents or storms, or by biological impact from corallivores or bioeroders (Highsmith 1982, Bowden-Kerby 2001). Micro-fragmentation focuses on encouraging slow growing coral colonies to spread across dead skeleton at a faster rate. Previous studies show that when reaching a certain size, some coral species may switch focus from growth to fecundity (Babcock 1984, Szmant 1991, Kai and Sakai 2008). In the opposite way, a larger previously reproductive colony may switch from reproduction to growth when fragmented (Highsmith 1982, Van Veghel and Kahmann 1994, Lirman 2000). It is the latter practice which micro-fragmentation exploits by splitting larger fragments into smaller sized pieces to enhance growth rate. Fragmentation has been used as a supply for the aquarium trade in particular (Delbeek, J 2001, Shafir et al. 2001), but it has increasingly been integrated as a means of coral reef rehabilitation. Fragments of colonies are collected and brought to a land-based laboratory where they are split into smaller pieces with a size of approximately 1 cm, or 1-5 polyp mouths, per square. The pieces are produced using a mechanical saw. Any excess skeleton is removed to produce a planar base. The small fragments are mounted onto cement discs, using a non-harmful glue. The fragments are thereafter brought up in tanks providing optimum conditions for the fragments to grow and encrust over an artificial substrate. In a recent study from Florida, US, (Page and Vaughan 2014, Page et al. 2018) micro-fragmentation was used to investigate the difference in growth rate, for live coral coverage of large fragments versus smaller arrays of micro-fragments outplanted onto dead coral reef substrate. Two common bouldering species, *Orbicella faveolata* and *Montastrea cavernosa*, were used for the experiment. The initial growth rate (167 days following the outplanting) of arrays of both species were significantly higher in comparison to the larger fragments. The two species are shown (Fig. 1.13) with *O. faveolata* on the left and *M. cavern* on the right. For each graph, the arrays are displayed on the left and larger fragments on the right. The results show a higher level of live coral surface area for both species of arrays, although with a level of high variability in the results. Thus, for the long-term (2.5 years following) result, the difference between the arrays and larger fragments of both species were not significant (Fig. 1.14) due to this variability.

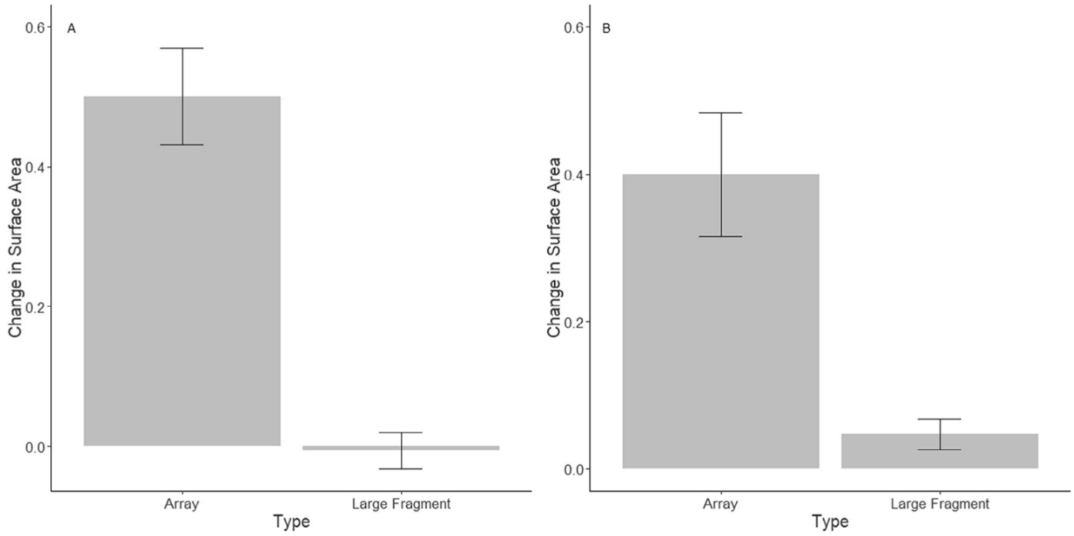


Fig. 1.14: The change in surface area (per initial cm^2) of two bouldering species, *Orbicella faveolata* **(A)** and *Montastraea cavernosa* **(B)**, over a duration of 2.5 years (courtesy of (Page et al. 2018)). The error bars represent the standard error of the mean.

The variability was most likely primarily caused by differences in predation and a possible genetic distinction. However, the study showed that the arrays of micro-fragments of the species *O. faveolata* produced almost 6.5 times more tissue than the larger fragments and the *M. cavernosa* almost twice the amount of tissue throughout the course of the experiment.

1.3: Rationale

A myriad of conditions may affect the success of coral restoration. An understanding of the environmental conditions, may that be of biological, physical, political or governmental character, prove vital to construct a suitable project adapted to particular conditions. Emphasising the challenges of a certain area could enable a more suitable approach in regard to choosing appropriate coral species and methodology. A fundamental understanding and knowledge of the processes of coral restoration, combined with an ecological assessment of the outplant area, may benefit prior to planning a coral restoration project. A pre-study of local conditions can provide valuable information when combined with a comparison between methods and results extracted from previous similar projects. Few projects have attempted coral restoration in Barbados, and even fewer studies have monitored and presented the results.

Aim:

to observe and contribute to a current coral restoration project on the west coast of Barbados in and outside of the Folkestone Marine Reserve. The thesis will explore existing literature to identify best practices within the Western Atlantic and Caribbean Sea in order to present experience and evidence based guidelines for future coral restoration projects.

Specific objectives:

- To investigate a recent coral restoration project initiated by the Barbadian government and completed by Coral Reef Alliance, Barbados (CORALL).

- To use meta analyses of projects from within the Caribbean and across the world to determine 1) whether survival rate varies across different areas within the Caribbean region, 2) which methods of coral restoration are the most commonly used, and, 3) which have proven most successful in rate of survival.

- To compare the results of the meta-analyses with the coral restoration project to explore survival rates 1) across regions, 2) lifeform and species and 4) methodologies.

Chapter 2: A Case Study on Coral Restoration, Barbados

2.1: The Island

When the Caribbean plate and the Atlantic crustal collided over one million years ago, a continuous tectonic uplift created a habitat of shallow water which encouraged coral growth. In the midst of this, the unique coral island appeared (Fig. 2.1), amongst the islands formed by volcanic eruptions, on the border between the Atlantic Ocean and the south eastern part of the Caribbean Sea (Blanchon and Eisenhauer 2001). In more modern times, following the British colonisation in 1625, the land was deforested to grow tobacco, cotton and sugar cane. Regardless of the abolition in 1834, Barbados remained a British colony until its full independence in 1966 and is still today a member of the commonwealth. The population of Barbados is currently estimated to be just short of 290,000 on an Island that is just a little

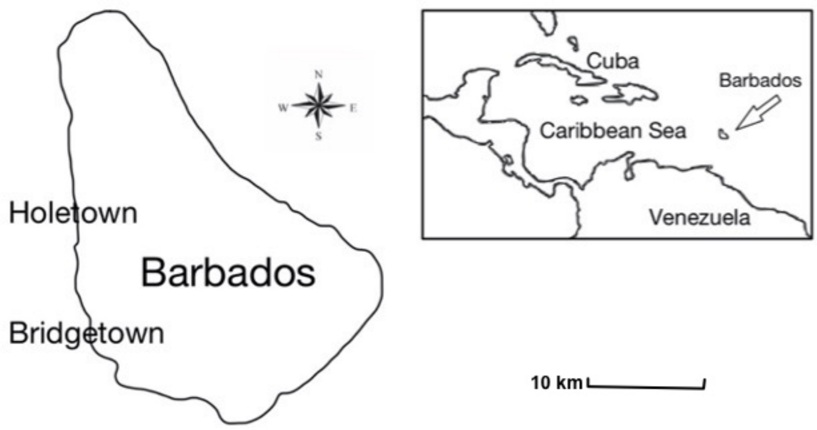


Fig. 2.1: Barbados located on the border between the Atlantic Ocean and the Caribbean Sea, north of Venezuela.

larger than 430 Km^2, with an average of 39,000 international visitors per month (World Bank 2019). In the last two decades, the number of individuals visiting Barbados has increased by more than 200,000 from 442,000 per year to 632,000 in 2016 (Fig. 2.2). The tourism industry has meant a high level of financial development for Barbados, but it has also had an effect on several environmental aspects, due to coastal development, failing wastewater management, and coastal disturbance. In 2018, the travel and tourist industry contributed almost 35 % to the gross national product (GDP) as a share of GDP with a total average of 36 ±3.31 % in the last decade (Turner 2018). Thus, tourism is one of the most important industries in the country, along with agriculture and fishery. Agriculture occupies an average of 41 % (1961 – 2019) of the total land area in Barbados (World Bank 2019). and along with forestry and fishery, it amounts to 4.28 ± 2.80 % of the annual GDP. Previously, Barbados relied predominantly on the production of sugar cane, but during the mid-1990s, many younger Barbadians realised an opportunity for a change in living standards and shifted their focus from agriculture to tourism,

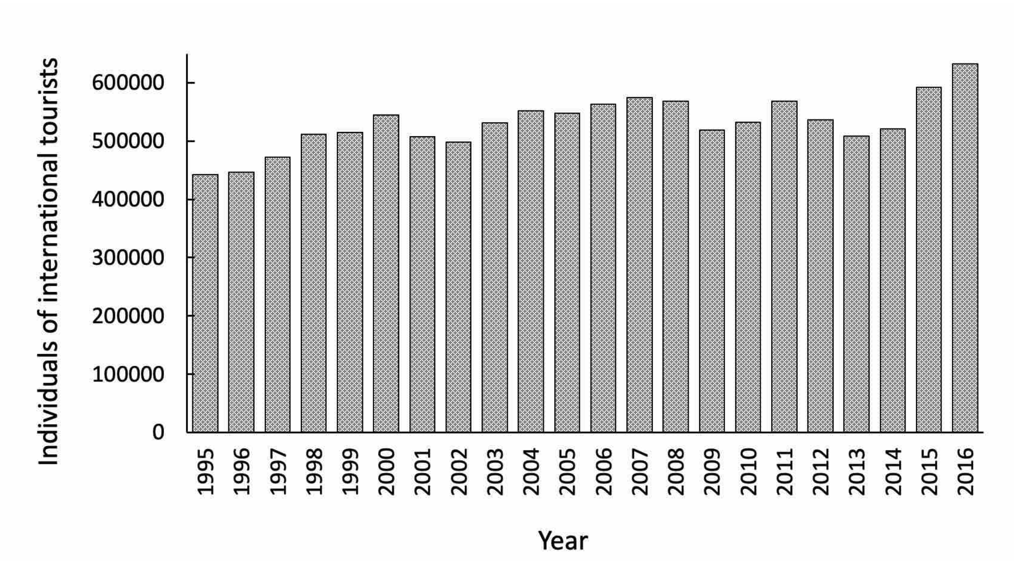


Figure 2.2: Annual numbers of international visitor since 1995, showing an increase of almost 200,000 people per year from first recordings in 1995 to 2016 (World Bank 2019).

which meant a massive decline in the cover of agricultural land (Fig. 2.3). Another decline can be noticed in 2009, which was caused not only by the global recession, but also a change in international trading arrangements (Braithwaite 2013). The farmers had, up to then, been able to trust an agreement that guaranteed prices above the worldwide market. But a change of agreements, initiated by the European Union, caused the expenses to increase and the net income to drop, resulting in financial losses for local farmers. Agriculture has had an impact on the coral reefs through the waste that is produced in the production of farm animals and crops. The methods by which the land is prepared and cultivated impacts, in particular when considering the coral reefs, surface water runoff (Moore et al. 2014).

A similar decline can be observed for fishery. The numbers for "Captured Fishery Production" (Fig. 2.4) show a high level of variability with peaks and troughs since 1960. However, an overall general trendline shows a decline from the second lowest recorded numbers at the beginning of the 1960s (2,113 metric tons in 1964) up to today, with one significant peak in 1988 which landed a total of 8,939 metric tons (World Bank 2019). The annual landings thereafter declined with the lowest total in 2012 with 1,371 metric tons, except for one additional peak at 3,564 metrics tons in 2008.

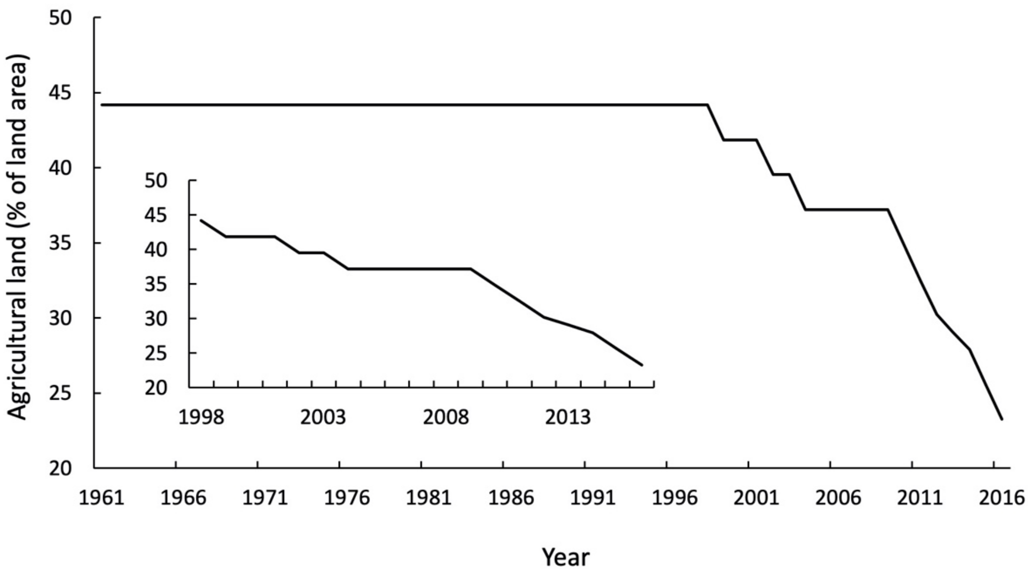


Fig. 2.3: The percentage of land used for agricultural purposes remained constant from 1961 until the first decline in 1997. Another drop occurred again in 2009. The decline caused much of the land to be left unused and still today many fields are left overgrown or have been sold for housing development. The small graph display the particual trend following 1997.

This was a time when Barbados in general experienced a financial recovery, before being affected by the global recession. The earlier peak between 1988 and 1989 could be partially due to the export of swordfish by US longliners operating in the Atlantic to support a demand from the Japanese market. Due to the convenient location of Bridgetown, many US fishermen transhipped for resupply and fuel (Weidner et al. 2001). Nowadays, the pelagic fishery still lands a large amount of fish per year and is considered to be of great economic importance. In contrast, the nearshore is still mostly unregulated and unmonitored due to being perceived as less important or influential. But although a substantial number of data exist on the numbers within the pelagic fishery, data are lacking for nearshore fishery (Gill et al. 2017). The limited data that do exist on the reef fishery are collected by The Barbados Fisheries Division and split into different subcategories, which further restricts the limpidity of accurate landing data. Although, it is known that the main landings of reef fishery include parrotfish (*Scaridae*) and surgeonfish (*Acanthuridae*), both species are considered important to coral health since they

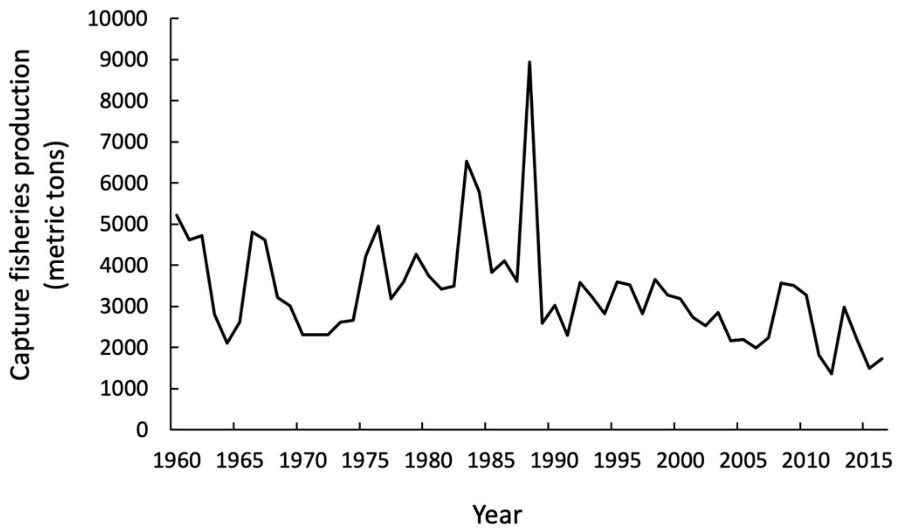


Fig. 2.4: The capture fisheries production, meaning the volume of fish catches landed for purposes such as commercial, industrial, recreational and subsistence use, measured in metric tons per year, showing a steady decrease since first recordings in 1960 up to 2016 (data from World Bank, 2019).

enhance coral growth and reproduction through their herbivorous behaviour (Mahon et al. 2007, Mumby 2009b). In examples for other fishery-dependent countries, when commercial fishery declines, prices increase which leads to a higher level of dependency on locally caught fish, often provided from reef-based systems. Thus, suggests a need for increased management of this fishing method to avert impending damage to the remaining reef system.

Throughout Barbados, the livelihood and economic foundation of the population is primarily dependent on the ability of terrestrial and marine ecosystems to provide ecosystem services. In order to monopolise these services, of course, a healthy environment is required. One of these ecosystems is made up of the coral reefs surrounding the island. The coral reefs are the keystone in maintaining a shoreline that supports a sustainable fishery and continues to attract many tourists every year (Moore et al. 2014).

2.2: CORALL

The Coral Reef Restoration Alliance (CORALL) is a non-governmental non-profit association established in Barbados in 2016. The coral rehabilitation project was initially implemented by the Barbadian government through the Ministry of Environment, Water Resources, and Drainage's Coastal Zone Management Unit under the Coastal Risk Assessment and

Management Programme. The project was financially supported by non-reimbursable technical-cooperation funding for the project through public-private partnership to Preserve Coral Reefs through the Inter-American Development Bank (IDB). A budget of US$ 1,540,950 was allocated to the project with a set schedule of 42 months for execution and 48 months for disbursement (Watson et al. 2015). The project aimed to "*address the declining health of Barbados' nearshore reef, likely to threaten the physical and economic viability of the coastal zone*" (Watson et al. 2015). The key methods for achieving were: 1) through transplanting > 60,000 units of coral fragments produced in laboratories, 2) developing a partnership with > 7 hotels through which it would be possible to 3) reach > 50,000 people "through new tourism product marketing campaigns centred on coral restoration", and lastly with 4) an additional > 10,000 tourists who were to "*participate directly in the coral care and restoration tours"* (Watson et al. 2015). The government stated that they were not willing to support the project long-term and thus, once CORALL had been running for a year, the funding was supported by additional membership fees and volunteer contributions.

The coral cultivating facility was assembled in the wet laboratory at Bellairs Research Institute, Holetown, Barbados. Through micro-fragmentation, the corals were grown and kept *ex situ* in the laboratory for later outplanting. Dr David Vaughan, then associated with the MOTE Marine Laboratory and aquarium, Florida USA, held a two-day course (2016) to educate the staff on the process of micro-fragmentation. The laboratory was assembled to contain two times six tanks to keep coral fragments which were covered by a continuous inflow of seawater from Folkestone Marine Park. Coral fragments were provided by the Barbadian Coastal Zone Management Unit (CZMU) and collected primarily from the dive site (Asta) located on the southern part of Barbados (precise location not obtainable). Fragments were collected from three hard coral species, the critically endangered elkhorn (*Acropora palmata*), and the two common species categorised as of least concern according to IUCN, finger coral (*Porites porites*) and brain coral (*Pseudodiploria strigosa*). These species were chosen in an attempt to provide aid for the endangered population of *Acroporids* while conserving the existing population of *Poritoid* and *Mussid*, thus cooperatively encouraging a diverse topography through diverse coral structure. Two coral nurses were employed to manage the laboratory and maintain an ambient environment for the produced coral fragments (ramets) until outplanting. The ramets were kept in the laboratory for a year before being outplanted.

2.3: Methods

The Sites

All fragments included in the CORALL project were collected from same reef (Asta reef) of the southern part of Barbados (Fig. 2.5). The fragments were later split between three transplantation sites (Fig. 2.5) on the west coast of the island, all representing different environments. The first site was located in the north, Port St Charles (13°15”31’N 59°38”35’W), a residential luxury marina development with a high level of marine traffic from a combination of private yachts and fishing boats. The marina is made up by two breakwaters, one of limestone and one of granite boulders. In the between the floor is dominated by a sandy floor. The coral fragments were outplanted on the limestone boulders of one of the breakwaters.

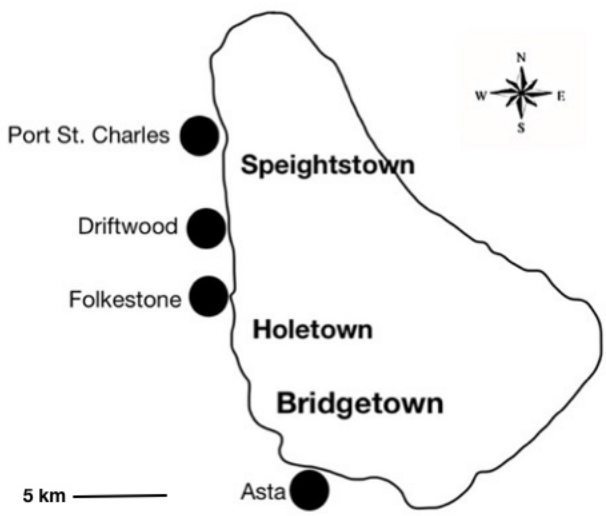


Fig. 2.5: Fragments collected from the southern part of Barbados on the Asta reef. The three outplanting sites were located along the west coast of Barbados, the most northern spot being Port St. Charles and the most southern site being Folkestone Marine Reserve, with Driftwood in between.

The area surrounds a small beach used by tourist and the local community with a small port for mooring. Due to the local current and placement of the breakwaters, there is a gentle current flow through the marina, introducing a continuous change in water. The second site, Driftwood (13°12”45’N 59°38”26’W), is located further to the south (Fig. 2.5). The site experiences a low level of traffic due to shallow reef formations. However, there is often a high level of breaking waves and some local fishermen reef fishing (net and traps) from shore. In general, there is a low level of human pressure, although some private bungalows and small exclusive hotels surround the area. The substrate is sandy with ridges of live and dead corals. The coral fragments were placed on a metal frame surrounded by boulders. The third and most southern site (13°12”18’N 59°38”15’W) was located within *The Folkstone Park and Marine Reserve*, approximately 2 km north of Holetown (Fig. 2.5). The transplantation site was within the scientific zone of the reserve, only frequented by swimmers and snorkelers; it has a no-take fishing restriction. Vehicles with special permission are allowed in at a maximum speed at 5 knots. The location experiences some high swells, depending on the season, with a substrate dominated by boulders and sand. The coral fragments were placed on graphite boulders scattered across the seafloor.

Micro-fragmentation

Fragments for micro-fragmentation were collected by the team from the CZMU, Barbados. Fragments were collected from three hard coral species *Acropora palmata* (AP), *Porites porites* (PP) and *Pseudodiploria strigosa* (PS). The donor colonies were disseminated using a hammer and chisel and coral pieces were handed over to CORALL Barbados for further processing in the

Fig. 2.6: Demonstration of microfragmentation and final ramets attached to tiles. **Left**: Larger fragments are cut into smaller ramets using a diamond saw, C-40, Gryphon Corporation. Each ramet contains 1 – 5 polyps per square piece. Any excess skeleton underneath the top layer of calcium is gently removed to achieve speedy attachment onto the artificial substrate. **Right:** Final ramets displayed on white tiles whilst being kept ex situ within the CORALL laboratory (photos courtesy of Zoë Lisk, coral nurse, CORALL).

laboratory. The original fragments were cut into smaller fragments (ramets), using a water-cooled diamond saw (C-40, Gryphon Corporation) (Fig. 2.6, left). Ramets of a size of ~ 1 cm^2 of live coral tissue were cut from the original donor fragments. Each ramet on average contained 1 – 5 polyps per piece. Any excess skeleton underneath the polyps was carefully removed, to encourage tissue to flush to the artificial base (Forsman et al. 2015). The fragments were then mounted onto white plastic tiles (4 x 4 cm) using epoxy (Attwood Epoxy Putty Stick) (Fig. 2.6, right). The same procedure was used for the branching and massive lifeforms. The initial procedure was demonstrated and supervised by Dr David Vaughan.

Laboratory Set-up and Maintenance

Each tank was afforded a continuous flow of managed transitory seawater (Fig. 2.7). The water was directly brought from the Folkestone Marine Park and Reserve located adjacent to the laboratory. The seawater was sampled each week to monitor nutrient and pH values of the water entering the tanks. An eXact iDip® 570 Marine Kit was used to investigate water samples for ammonia, calcium, nitrate, hardness and pH value. The temperature of the laboratory was kept at a constant 28 degrees Celsius (^{o}C), and the incoming saltwater was kept at natural temperature. The ramets were kept in the laboratory for a year allowing them to acclimatise and grow.

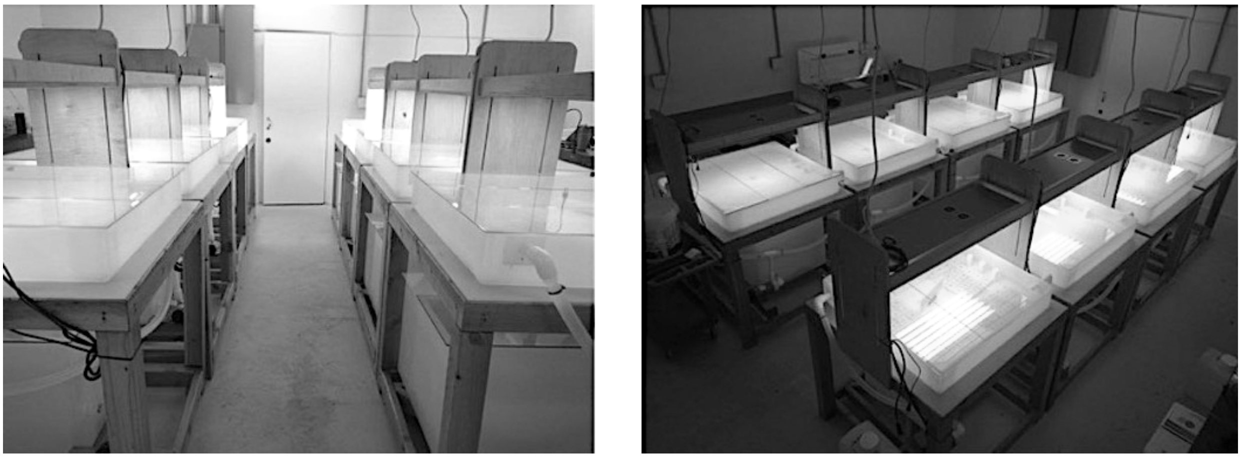


Fig. 2.7: The water tank set-up; two rows of four tanks, with a continuous flow of sea water pulled in from the Folkestone Marine Park and Reserve and supply of light (photos courtesy of CORALL).

Water Quality

A 250 mL water sample was collected from Folkestone (A), Driftwood (B) and Port St. Charles (C) to run tests for nutrient analysis. Due to limited funds, these analyses were only done once (June 2018). Analysis tested levels of ammonia, phosphate and nitrate-N. The collected water samples were also tested for total suspended solids (TSS).

Nutrient analysis

Ammonia

A 2 mL sample was pipetted from each seawater site sample, plus an additional 2 mL from Folkestone to act as a sample control (A+). In addition, a 2 mL sample of iodised water and a sample of 2 mL standard solution was added in the analysis for experimental control. A reagent (AmVer™ Diluent Reagent, Hach® Company) was added to each sample and left for 20 min to allow a colouration to fully develop. The mixed samples were compared individually to a colour disc and the result was recorded (Hach 2015).

Phosphate

5 mL of seawater was pipetted from each seawater site sample, plus an additional 5 mL from Folkestone to act as a sample control (A+), and an additional 5 mL of iodised water and a sample of 5 mL standard solution for experimental control. The samples were poured into separate tubes and a phosphate reagent (PhosVer® 3 Phosphate Reagent, Hach® Company) was added to each. The tubes were sealed with a silicone film and agitated in a centrifuge until the powder had almost dissolved. As the sample solution and phosphate reagent reacted, a blue colouration developed. For accurate use, a spectrometer was reset by using a cuvette containing iodised water. The solution from each tube was poured into separate cuvettes and successively placed in a spectrometer (HANNA HI 83200-01) for individual sample readings. The data results were recorded (Hach 2017).

Nitrate-N

15 mL was pipetted from each seawater sample plus an extra sample control from Folkestone (A+) into test tubes, with an additional 15 mL of iodised water and a sample of 15 mL standard solution. A reagent was added to each sample (NitraVer® 6 Nitrate reagent). The test tubes

were covered by silicone film, evenly distributed in a centrifuge and mixed until the reagent was fully dissolved in all tubes. Solutions were left to rest for two min. 10 mL was taken from each mixed sample and placed in new test tubes. A second reagent was added (NitriVer® 3 Nitrite reagent) and the tubes were all again evenly distributed within the centrifuge and left until the reagent had fully dissolved. The spectrometer was once again reset by a sample of iodised water, and the solutions were poured into cuvettes and successively placed in the spectrometer for value readings. All data were noted (Hach 2014).

Total Suspended Solids

To determine the level of sedimentation at each site, a water sample was tested for total suspended solids (TSS). A pre-prepared weighed standard glass-microfibre filter of a diameter of 47 mm and pore size of 1.5 μm (Hach® Company) was inserted (wrinkled side upwards) into the filtration apparatus per each water sample. The suction function was switched on and a small volume of reagent-grade water was added to infuse the filter. A sample of 100 mL seawater from Folkestone (A) was filtrated through the filter funnel. The filter was washed with 3 x 10 mL volumes of reagent-grade and 3 min suction followed for complete drainage. The filter was carefully removed from the filtering apparatus and transferred to an aluminium weighing dish. The weighing dish containing the filters was placed in an oven for one hour at 103 – 105 ^{o}C. The dish was thereafter cooled and weighed. The methods for drying, cooling and weighing were repeated until a consistent weight was accomplished. All of the above was repeated for the samples collected at Driftwood (B) and Port St. Charles (C). An additional cycle was repeated on an extra 100 mL sample from Folkestone (A+) as a sample control. To perform an experimental control, 100 mL of deionised water and another sample of 100 mL of a standard seawater solution (STD) was filtered according to the same procedure as the site samples (Jenkins 1982).

Ecological Assessment

To establish the ecological environment surrounding the three outplanting sites, an ecological assessment of each location was conducted. The method was inspired by Reef Check (Freiwald et al. 2019) but adapted to each site for maximum applicability (Fig. 2.8). Thus, each site location slightly differed in terms of the method to accommodate the variation in topography but, in general, the same procedure was applied at the sites. Three transect surveys were conducted per transect line at ~ 3 m by a team of two divers during two dives. Two 5 m wide (centred by the transect line) by 30 m sections were placed at each outplant site to survey diversity and abundance of invertebrates, fish, substrate and impact. The first survey collected data on key fish important for fishery and aquarium trade (Appendix 1). The next survey was on invertebrates and substrate (Appendix 1). Specified species of invertebrates of ecological importance or significance for food or curio trade were accounted for. The surveyor also noted any visible impact on the substrate, e.g. coral disease, bleaching, damage, predation. Due to financial limitations, these surveys were done in one dive per transect section (30 m). At Port St. Charles, the transect line was placed on the breakwater at a depth of approximately 4 m; this depth permitted each diver to work in a width of 2.5 m on each side of the transect. It was decided to place the transect directly on the breakwater and not the seafloor due to the sand in surrounding the area. Conducting the assessment on the seafloor would have skewed the result and not accurately represented the substrate immediately below the ramets. The two transect lines were placed continuously with a 5 m gap between the transect lines to ensure sample independence. At both Driftwood and Folkestone Marine Reserve Park, the topography at the sites made it difficult to collect data parallel to the shore (Fig. 2.8). The reef ridges are perpendicular to the coastline, demanding the surveyors to ascend and descend throughout

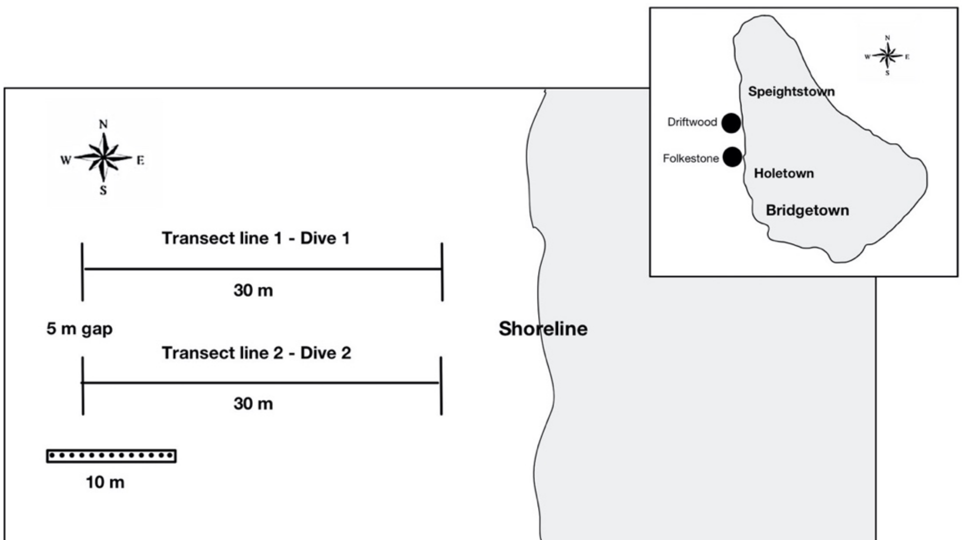


Fig. 2.8: The perpendicular placement of the 2 x 30 m transects at Folkestone and Driftwood, with 5 m between to eliminate repeat of individual counts.

the survey. This created a safety issue, and as the depth remain relatively constant until further offshore, the transects were, instead, placed outwards. The transect lines were placed with a gap of 10 m to ensure sample independence. Two surveys (2 x 30 m) were done per site once per month during November and December 2018 over a period of three days. All surveys were done between 9 am and 3 pm, by a team of two scuba divers. One day of surveys involved two dives processing two transects. One dive included 1) reeling out the transect, 2) video documentation of the full transect recording of the underlaying substrate, 3) a fish survey (Fig. 2.9), 4) invertebrate, substrate and impact survey and lastly (Fig. 2.10) 5) reeling in the transect. The duration of one transect was an hour. This process was repeated for the second transect at the same site.

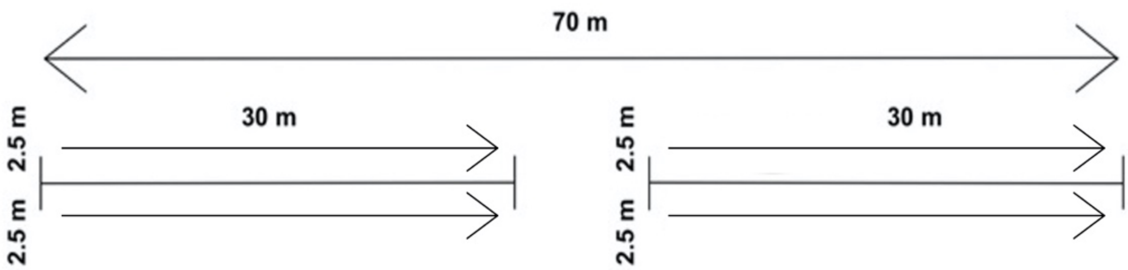


Fig. 2.9: The rectangular prism (30 m x 5 m x 5 m) area in which the fish survey was conducted along a transect. All species present were recorded to estimate the diversity and abundance. The survey was conducted by continuously swimming parallel to the transect line. No recordings were collected in the gap between the transects.

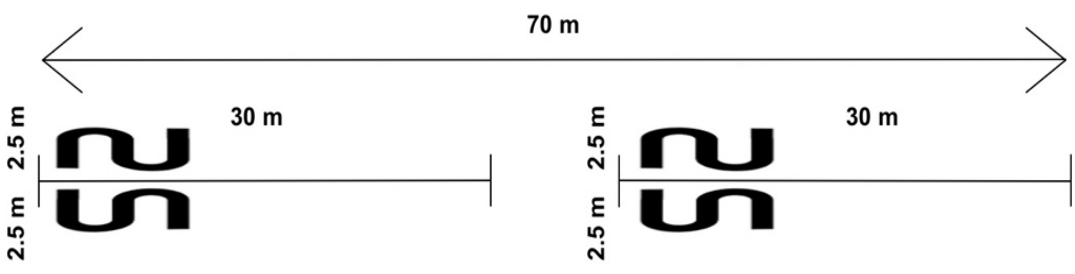


Fig. 2.10: Illustration of two 30 m sections of the 70 m belt transect, to show the area in which invertebrates were recorded. Notice for this survey each section was 30 m with a 10 m separation gap between. The survey was done in a continuously S-shaped pattern along the transect.

Transplantation

In preparation for the outplanting, all ramets were detached from the white plastic tiles (Fig. 2.11), a bolt was mounted on each tile (for attachment at sites) and the ramets were reattached with epoxy (Attwood Epoxy Putty Stick). Each individual tile was labelled with species specific colour coded plastic tape (*A. palmata*, yellow; *P. porites*, red; *P. strigosa*, green) with a handwritten (black marker) identification (ID) number for documentation at transplantation sites. ID numbers were planned to ease later identification during the follow-up monitoring. All ramets were outplanted in one day (18th May 2018). CZMU was present throughout the day for

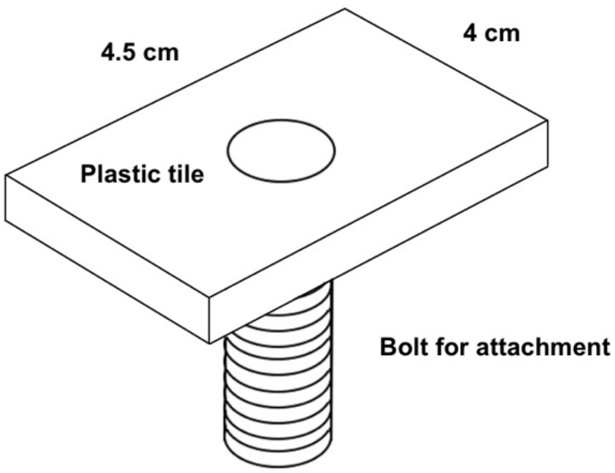


Fig. 2.11: Each plastic tile was equipped with a bolt for attachment to the reef. The tiles were made of glossy white plastic measuring approximately 4 x 4.5 cm.

coordination, assistance and observation. For each species, the size of each ramet at the time of outplanting was ~ 3 cm in diameter. A total of twenty ramets were outplanted at Port St. Charles (AP=1, PP=4, DS=15). These fragments were outplanted directly onto the boulders of limestone that make up the breakwater in front of the marina (Fig. 2.12), placing the ramets (tide dependent) ~ 3 m above the sandy seafloor and 3 m below the sea surface.

Spaces to place the outplants were found from a quick evaluation of the site. A broom brush was used to remove algae on the substrate. Holes were drilled using an underwater drill (Fig. 2.13). The holes were each filled with epoxy (Attwood Epoxy Putty Stick) and left for 12 min. to harden. The bolt underneath the tile was pushed into the epoxy.

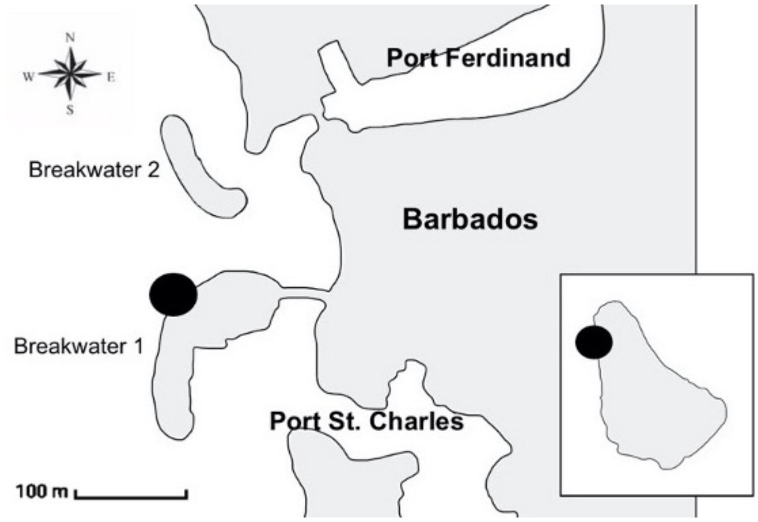


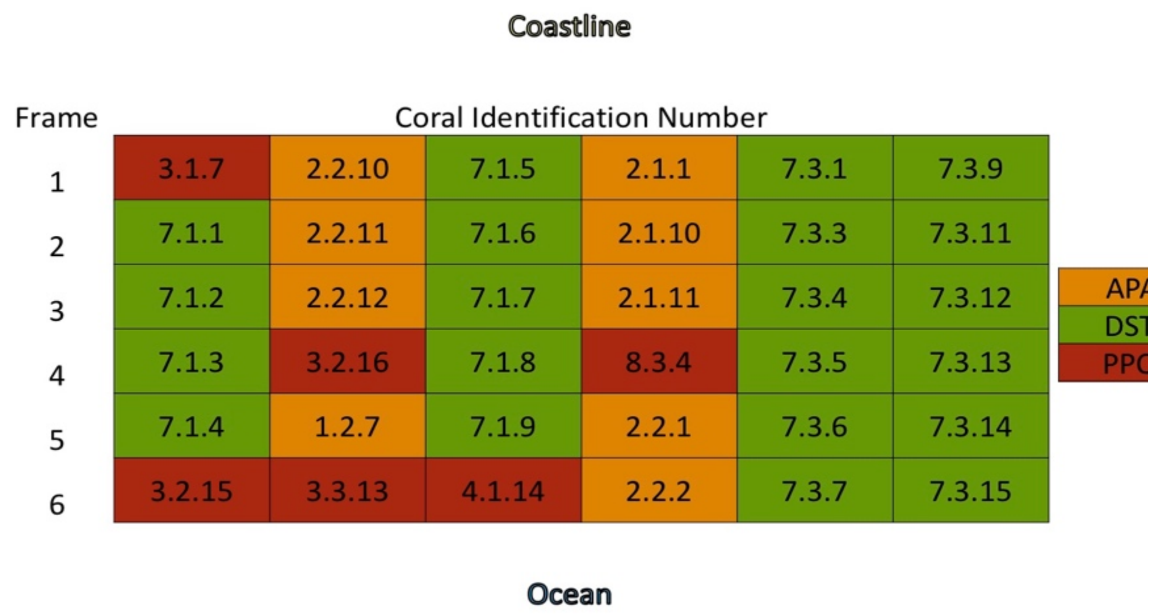
Fig. 2.12: Location of outplanting site at Port St. Charles, the most northern of the three chosen sites.

Fig 2.13: Team members of CORALL and the Barbadian CZMU at Port St. Charles. Holes were driled into limestone boulders and saturated with mouldable epoxy which was left to harden while the team moved on to prepare next place of attachment. Tiles were pressed into the epoxy when hardend satisfactorily, on average after 12 minutes (private photo).

Thirty-six ramets (AP=9, PP=6, DS=21) were outplanted at Driftwood. The fragments were mounted onto metal rails which were then attached to the permanent metal frame already present at the site. The fragments were placed according to an identification diagram for follow-up monitoring (Fig. 2.14A). The metal frame consisted of six rails, each with six ramets (Fig. 2.14B).

Twenty-two ramets (AP= 1, PP=4, DS=17) were outplanted at Folkestone Marine Reserve. A similar methodology was used as at Port St. Charles; an underwater drill was used to drill holes.

A



B

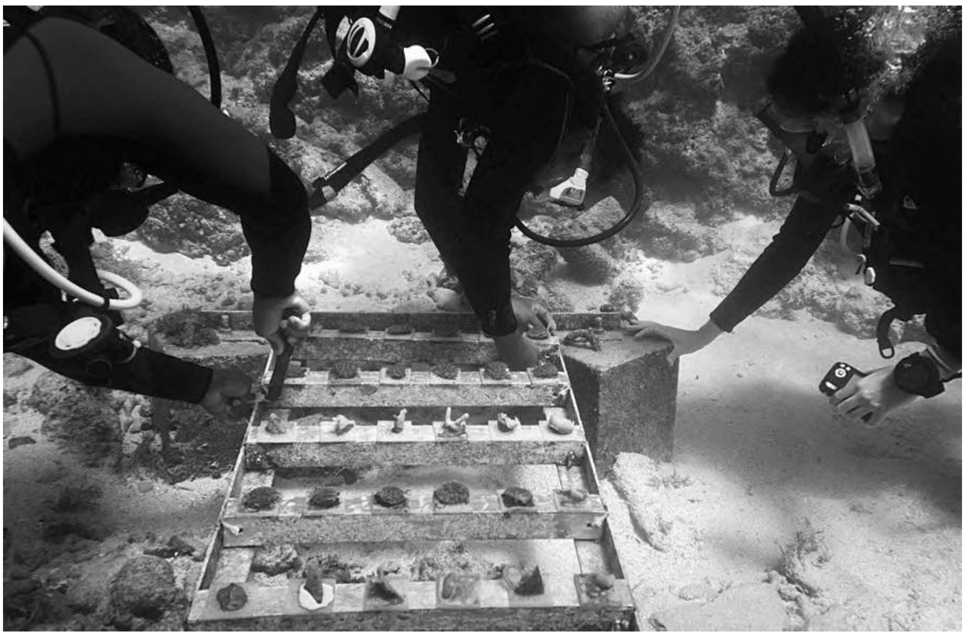


Fig. 2.14: (A) Diagram to identify coral fragments at Driftwood. *Acropora palmata*, orange (APAL), *Pseudodiploria strigosa*, green (DSTR), and *Porites porites*, red (PPOR). **(B)** Metal table at Driftwood with the fragments from 2018.

into the substrate, as here, in contrast to Port St. Charles, there were graphite boulders (Fig. 2.15), not limestone. Each tile was attached with the bolt in the epoxy for each hole and left to harden. The boulders were situated on a predominantly sandy floor and attaching the ramets onto the boulders lifted them ~ 2 m above the sedimentation and ~ 3 m below the sea surface

Fig. 2.15: Fragments of *Pseudodiploria strigosa* at Folkestone Marine Park and Reserve two months after. Plastic tape with ID numbers can be seen at each tile (private photo).

Data analysis

To test for differences in abundance of urchins per site, ANOVA tests were run with SPSS (Build 1.0.0.1347) with a Bonferroni post-hoc test. The same test was done to compare the fish abundance of parrot fish (Scaridae), grunts (Haemulidae), butterflyfish (Chaetodontidae) and surgeonfish (Acanthuridae) between sites. Further, the variation of fish community composition between the three different sites was visualised by non-metric multidimensional scaling (nMDS; after 999 permutations) based on a Bray-Curtis dissimilarity index. The test was run in R studio (Version 1.2.5033, 2009 – 2019). A test to compare the difference in abundance of substrate between sites was conducted by the use of ANOVA tests which were run with SPSS (Build 1.0.0.1347), with a Tukey post-hoc test.

2.4: Results

Water Quality

Nutrients

Ammonia

The results of this test showed a level of < 0.01 mg/L across all three sites which is a reading below the method detection limit (MDL) of 0.01 mg/L.. The result is considered non-harmful.

Phosphate

Results for the phosphate test showed a result of < 0.06 mg/L for three sites, which borders on the minimum range of detection for the test conducted on the sample. The MDL of this was 0.06, thus a result of less than.

Nitrate-N

Driftwood and Folkestone showed identical low levels of nitrate-N at 0.01 mg/L, whereas Port St. Charles showed a higher level at 0.06 mg/L.

Total Suspended Solid

Values for TSS between 10 – 25 mg/L are considered within the normal range. The results of our tests showed all sites had values within the normal range (Fig. 2.16), although Driftwood (25 mg/L) and Port St. Charles (23 mg/L) were at the high-end. Folkestone (21 mg/L) was in in the mid-range. No mean ± SE could be calculated due to the test only being done once.

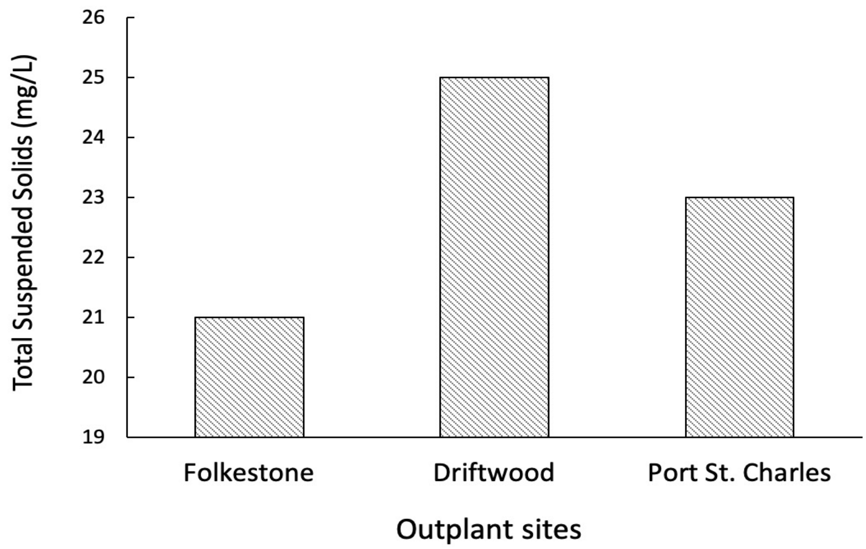


Fig. 2.16: Levels of TSS in June 2018 for each outplanting site. A range between 10 mg/L and 25 mg/L is considered the norm. All sites were in the higher end of the spectrum on the day the water samples were collected and tested.

Ecological Assessment

Fish

Port St. Charles presented a population density of 2.62 individual fish / m^3 with an average abundance of 1,967 ± 68.96 individuals per transect, and a species richness of 22. The value for species richness was echoed Folkestone (22) and 19 at Driftwood, although population varied across the sites with 1,752 ± 58.54 individual fish recorded per transect at Driftwood and 1,909 ± 62.83 at Folkestone. This made up a population density of 2.34 individual fish / m^3 and 2.55 individual fish / m^3 respectively. Three families repeatedly dominated across all the sites (Table 1), parrot fish (Scaridae), wrasse (Labridae) and damsel fish (Pomacentridae). Damsel fish are considered one group for the results section, but data were collected separately on two species in this group, the Sergeant Major (*Abudefduf saxatilis)* and the Brown Chromis (*Chromis multilineata*). The reasoning for this was that the Sergeant Major is often caught for the aquarium trade, thus, its presence or absence may act as an indication of fishery pressure. Meanwhile, the Brown Chromis occurred in vast numbers across all sites and it was decided to specify this species to prevent a skewed statistical result. When considering results of the number of individual damsel fish present at each site, deducting the Brown Chromis took the total of damsel fish at Port St. Charles to 189 individuals, Driftwood to 319 and Folkestone to

253. This still left the damsel fish as one of the dominant families at each site but resulted in a smaller value than when the species is included.

Table. 1: The three species occurring the most across all three sites.

	Port St. Charles	**Driftwood**	**Folkestone**	**Total fish per family**
Pomacentridae	1454	1090	1334	3878 ± 185.49
Scaridae	71	9	26	106 ± 32.04
Labridae	276	599	312	1147 ±154.05

A total of 23 additional species were observed across the three sites. Both Port St. Charles and Folkestone had 22 fish species with a variation of species represented between sites. Driftwood had 19 species. The sites showed a significant difference in abundance of parrot fish ($F(2,9) = 4.93$, $p = 0.036$), though the post-hoc showed the significant difference was only between Port St. Charles and Driftwood. Similarly, comparing the abundance of grunts across the sites showed a significant difference ($F(2,9) = 6.29$, $p = 0.020$), although here Port St. Charles and Driftwood showed no significant difference in the following post-hoc test. Looking at butterflyfish showed a significant difference ($F(2,9) = 11,78$ $p = 0.003$), although the post-hoc showed that there was no significant difference between Driftwood and Folkestone. When comparing data between sites for surgeon fish, there were no significant differences ($p = 0.68$) amongst the variables. The community composition revealed by a nMDS ordination (Fig. 2.17) was structured by site (Adonis, ($F(2,9) = 1.25$, $p = 0.34$). A following ANOVA showed a good level of homogeneity within the assumption ($F(2,9) = 1.08$, $p = 0.38$).

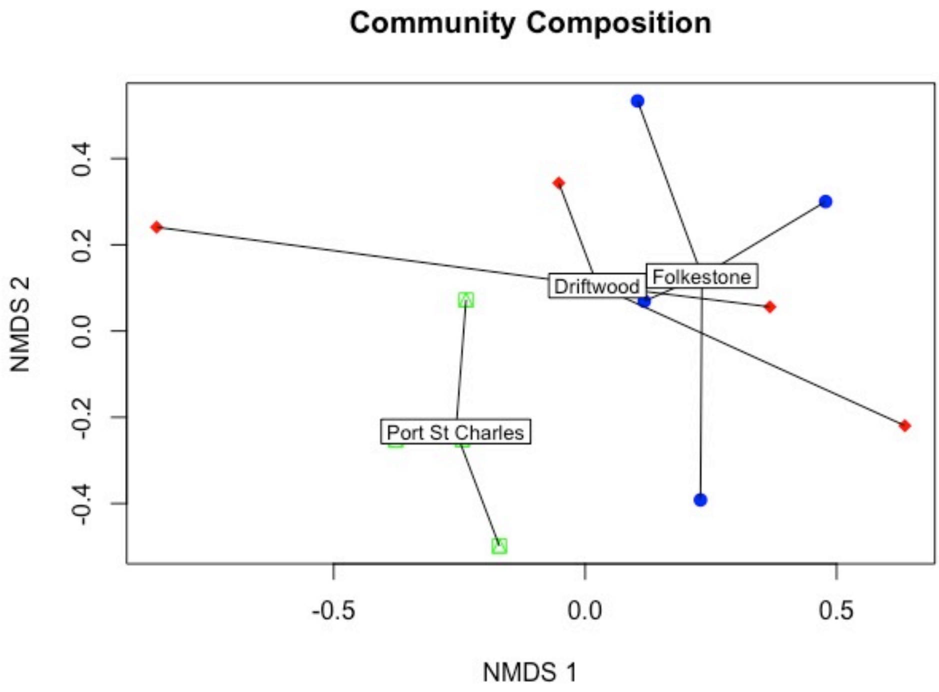


Fig. 2.17: A nMDS between the three sites: Port St. Charles (green), Driftwood (red) and Folkestone (blue), to investigate community composition. For fish species, Port St. Charles show very little similarity to the two other sites, whereas Driftwood and Folkestone had a certain level of overlapping, thus similarities.

Table 2: The Shannon-Weiner and Equitability results of the diversity at each outplanting site. Folkestone shows slightly higher diversity compared to Port St. Charles and Driftwood. Comparing the species' evenness across the three sites shows a high level of similarity, although Driftwood had a slightly higher level (0.46) of equitability.

Site	H'	Equitability
Port St. Charles	1.36	0.44
Driftwood	1.34	0.46
Folkestone	1.40	0.45

Invertebrates

Three species showed at higher level of presence at all sites, with additional species at one site. The three most common species throughout the sites were the Christmas tree worm (*Spirobranchus giganteus*), feather duster worm (*Sabellidae spp.*) and long spined sea urchin (*Diadema antillarum*). The sites showed a significant difference in abundance of long spined sea urchin (F(2,9) = 14.85, p = 0.001), although not between Port St. Charles and Folkestone. At Port Saint Charles only a total of two long spines sea urchins were spotted over November and December, one per month. Driftwood experienced a 45.65 % decline in individuals from November to December with a total of from 213, and Folkestone a 90.63 % drop from a total of 35 individuals over the two months.

Substrate

All three outplant sites commonly had either hard coral (HC), rock (RC), sand (SD) or rubble (RB) represented, although there was disparity in composition between sites due to the variation in representation of substrates (Fig. 2.18).

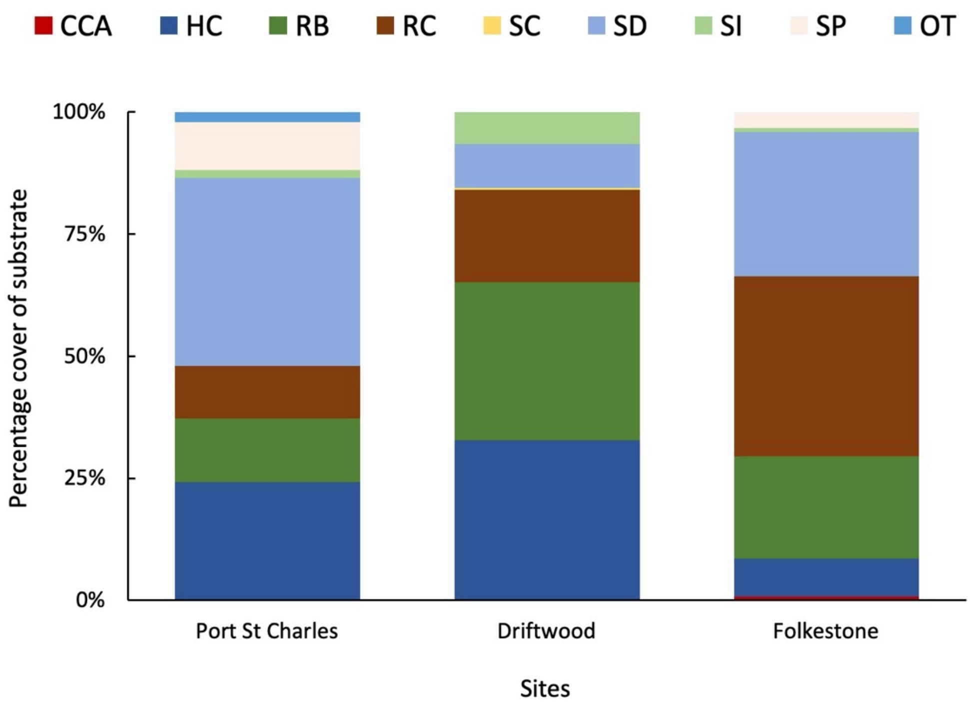


Fig. 2.18: Variation of substrate composition at each outplant site. Substrate was as follow; Calcifying Coralline Algae (CCA), Hard Coral (HC), Rubble (RB), Rock (RC), Soft Coral (SC), Sand (SD), Silt (SI), Sponge (SP) and Other (OT) which include rubbish, invertebrates and anything alike that is not possible to categorised under the specified remaining groups. All sites had HC (dark blue), RB (light grey), RC (light blue) and SD (yellow) represented, although the structure at each site varied due to the evenness of each substrate.

In a comparison of substrates between sites, only RC showed significant difference ($F(2,9) = 4.24$, $p = .05$). A Tukey test showed the difference was between Port St. Charles and Folkestone.

Outplants

When the monitoring ended in December 2018, only Driftwood had survivors representing all three species (Table 3). Folkestone had no surviving ramets, and Port St. Charles had only P. strigose left (27 % survivors). Considering the overall survival rate, in December 2018, with no discrimination between sites or species, the result was 21.04 % (± STDV 31.44 %), whereas, at the only site with survivors of all species, Driftwood had an overall survival rate of 54.23 % (STDV ± 34.32 %).

Table 3: The numbers show percentage survival out of the initial amount of ramets per species at each site over time. The original numbers for Ports St. Charles were AP = 1, PP = 4 and PS = 15. For Driftwood AP = 9, PP = 6 and PS = 21. And finally, Folkestone had AP = 1, PP = 4 and PS = 17.

			Surviving ramets (%)	
		A. palmata (AP)	*P. porites* (PP)	*P. strigosa* (PS)
Port St. Charles	November	0	0	27
	December	0	0	27
Driftwood	November	33	67	90
	December	22	50	90
Folkestone	November	0	50	6
	December	0	0	0

2.5: Discussion

A recent paper examined more than ten case studies on coral restoration. The result showed a typical survival rate of between 60 – 70 %, regardless of genera or species (Boström-Einarsson et al. 2019). The survival rate of the case-study in this thesis ranged from 0 – 90 % across the three sites between each species, with a total average of 35.90 %. Focusing solely on Driftwood, the only site that had survivors of all three species, the overall survival rate in December 2018 was 66.67 %.

The most evident difference between this site and the remaining two are the outplant substrate. Whilst the coral fragments of both Port Saint Charles and Folkestone were outplanted directly onto the substrate, the coral fragments at Driftwood were placed on a metal table elevated approximately 30 cm above the seafloor. Following the outplanting some fragments were lost, and whereas these fragments generally were lost at Port Saint Charles and Folkestone, the coral fragments at Driftwood were often trapped under the table and therefore easily found and reattached. Another benefit from placing the coral fragments on the metal table was the algae growth. At the other two sites the algae growth occurred rapidly following the outplant and the fragments exhibited signs of bleaching soon after the outplanting, with final die-off within two weeks. At Driftwood the algae growth was more sporadic and slow growing, giving the CORALL team time to adapt and implement a removal procedure, which continued throughout the remainder of the project.

Another reason for the steep decrease in coral survival at Folkestone was the presence of Scaridae. During the first week of the outplant the team witnessed bite marks, particularly on the *Porites porites*, and soon after the fragments had vanished entirely. The *Pseadudiploria strigosa* and the *Acropora palmata* did not seem to experience same level of predation. Should the study be repeated at Folkestone, perhaps possible fish predation should be included when considering the project planning. At Port Saint Charles the continuous current caused the fragments to loosen from the mounted tiles which, along with algae overgrowth, became the most common cause of fragments loss.

2.6: Recommendations

Funded projects focused on coral restoration are largely scheduled to last less than five years thus focusing predominantly on short-term results whilst limiting the opportunities of gaining long-term knowledge. Considering most published work, this appear to be a worldwide trend. A probable consequence may not only be the curtailed knowledge revenue, but also an unintended encouragement for project teams to haste through planning to get to the part that gain quick results. The information attained through confidential interviews of some of the people involved in the initial phase of CORALL left an impression of too many people and ideas were fitted into the 48 months project scheme. This meant that limited research had been done in preparation for the work in the laboratory, in the preparation prior to outplanting or the procedure for outplanting. Further, did the structure of the project consequently leave little data for reliable statistics. From an outside observation, this project could have benefitted for a considered course of direction and a well-informed management.

Below are listed some recommendation for future projects of coral restoration:

1. Coral fragments were mounted onto white smooth surfaced plastic tiles. These were initially used in the coral tanks in laboratory and later *in situ* at each outplant site. The tiles that were installed *in situ* are still remaining, without any proposition of removal. Thus, a project that was allegedly enhancing the coral reef has added to the increasing threat of plastic pollution in the aquatic environment. Further, studies show that corals prefer to settle and grow on substrates that are rough in contour and of similar colour to themselves (Salinas-de-León et al. 2011). Hence, mounting the fragments on a shiny white surface may have restricted their initial growth. Cement discs, which have already been tested and successfully used in previous projects (Bowden-kerby and Carne 2012), could have been a fitting alternative. These are easily made, at very low cost. A greater consideration for mounting methods and materials of coral fragments may not only benefit the results of the project, but also the environment.

2. The identification diagram at Driftwood worked very well as an accurate and easy method of identifying each individual fragment. In contrast, the plastic tape that was used at Port St. Charles and Folkestone Reserve began to either dissolve or become overgrown with algae within two weeks of the outplanting. Some labels showed sign of predation. A more practical labelling system, with again less plastic pollution, would be advisable for future projects. Earlier projects have shown metal labels as a sustainable system (Lirman 2000). Previously mentioned cement discs could additionally be incorporated into the labelling system by moulded in different shapes to differentiate between species. Using an impractical labelling system makes it 1) time consuming to identify coral fragments when processing photos and 2) further, create a source of error by misidentification.

3. The methods and sites exposed the coral fragments to varied biological and physiological impacts without constants. The project design further failed to include a control group. For these reasons the data were inadequate for statistical comparison.

By ignoring a statistical footing valuable information may have been left undiscovered. Furthermore, CORALL did not organise site assessments prior to the outplanting, which meant that annual or seasonal fluctuations or changes between the sites were unknown. Neither was it possible to compare changes at the sites before and after the coral restoration.

4. The species used in the project could have benefitted from a site match. A visual assessment of the sites would reveal each of the species represented at specific sites. Port Saint Charles showed an abundance of *Pseudodiploria strigosa*, Driftwood a general population of *Porites porites* and Folkestone a combination of different species with some occurrences of *Acropora palmata*. Pairing the species with habitat could potentially have advanced the outcome of the project. According to the Reef Restoration and Adaptation Program under 5 % of transplantation studies (Fig. 2.19) include slow growing species (Boström-Einarsson et al. 2019). This is possibly related to the common time restrictions as previously mentioned. Due to the development of micro-fragmentation, designed by the former senior scientist at MOTE Dr David Vaughan, it is now possible to speed up the growth of slow growing hard coral species. This is a valuable discovery, which can potentially have a great impact on future coral restoration by incorporate a greater species diversity, thus encourage a greater reef complexity. Not much research has yet been mustered on the method, thus knowledge is limited. CORALL was testing the method but failed to recognise the potential the site at Port Saint Charles presented; A site that showed an already well-established population of the slow growing species, *Pseudodiploria strigosa*, suggesting suitable conditions for survival. Planning and adapting according to the given conditions may present opportunities for greater focus on the outcome.

5. The results of the ecological assessment at Driftwood showed highest level of live coral coverage and lowest occurrence of sand, whilst water quality showed highest level of TSS. This could be due to shallow waters combined with high level of water movement, which the high level of rubbles is a testimony of but may also be due to a high level of plankton. In addition, Driftwood also exhibited the lowest level of sponge (Porifera) abundance providing new coral colonies the opportunity of spreading due to decreased

spatial competition. Further, Driftwood was the only site that showed a presence of soft coral and the statistical analysis of all three sites showed a significant difference between the two other sites at Port St. Charles and Folkestone, but none between Driftwood. Thus, this result could suggest an overlap in community composition at Driftwood of the other two sites. Considering the fish community at each site in relation to the survival rate of the outplanted coral ramets, Driftwood, with the highest survival rate, showed the lowest level of fish diversity but scored the highest evenness across the species present.

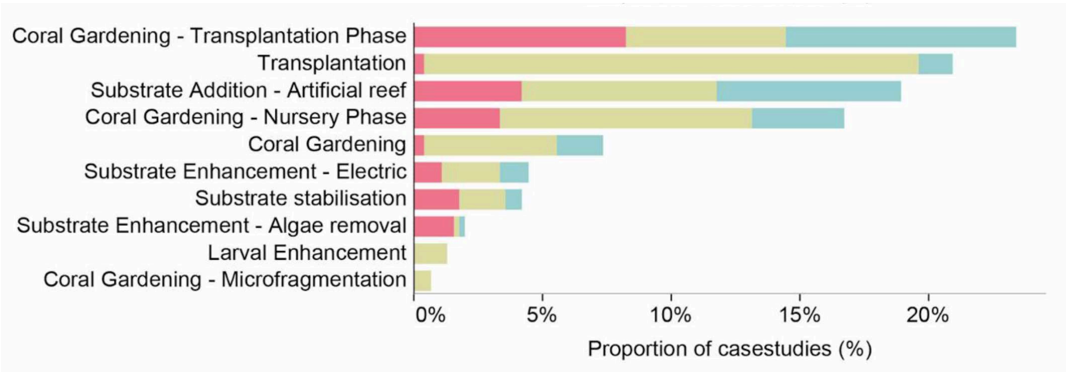


Fig. 2.19: Showing the proportion of methodologies used in global coral restoration projects, displaying micro-fragmentation as the least tested method (courtesy of Boström-Einarsson et al., 2019).

6. Micro fragmentation was an experimental part of CORALL requiring a temporarily *ex situ* laboratory to grow collected coral fragments. The fragments were treated by standard method and the ramets were kept in tanks with a constant waterflow and light source regulated by the nurses. Unfortunately, the ramets experienced a number of stressful events during their one-year period *ex situ*, causing some to bleach, ultimately a portion died off. When comparing the cost-benefit of the short-term laboratory with the outcome of the project each surviving ramets has come at a very high price when considering the limited knowledge, the project produced. In an attempt to produce a balance between cost and benefit two possible courses could be taken; 1) allow the coral nurses working the laboratory to focus on species suitable for a tank environment and 2) expand the time in the laboratory to produced generations instead of reintroduce first generations back into an environment following a stressful time in the *ex situ* environment.

Chapter 3: A meta-analysis of Coral Restoration, the Western Atlantic and Caribbean Sea

3.1: Introduction

Systematic reviews and meta-analyses were initially a tool developed for personnel working in healthcare as a quick and easy access to information collected from numerous research papers (Ganeshkumar and Gopalakrishnan 2013). The method provides wholesome evidence when addressing a specific question as it allows the researcher to expand an examination across many studies in contrast to a few. Thus, a systematic review reduces the probability of a constricted conclusion (Mikolajewicz and Komarova 2019). A systematic review is the summary of research which uses constructed and reproduceable methods when addressing a certain inquiry. The summary is based on primary studies and constructed in such a way that preconception is diminished, and random errors reduced. The meta-analysis may follow as the statistical strategy that gather the collected research into a complete estimate by reducing quantity to a single result through an effect-size (Fernandez-Duque and Valeggia 1994, Osenberg et al. 1999). The systematic review and meta-analysis combined account for six steps; 1) construct the question, 2) gathering all literature on topic, 3) gather relevant literature, 4) evaluate the quality, 5) calculate the outcome measures of each study and combined them and 6) interpret results (Ganeshkumar and Gopalakrishnan 2013). First, construct the question. The foundation of the remainder of the process. The question must be concise and clear in its structure and direction of investigation. Secondly, gather the information regarding said question to retrieve relevant sources. Arrange criteria for the acquired sources, collect through scientific databases, grey literature (e.g. thesis, non-peer-reviewed journals), internal report etc. Extract from online databases using Boolean operators AND, OR and NOT. Next step will exclude any sources that does not provide adequate or useful information, whilst gathering all relevant literature. Then an evaluation of quality; have the collected data been presented in a consistent manner? Did the studies provide the accurate data useful for this analysis? When these measures are finalised, collect the retrieved data in a comprehensive database to calculate the outcome measures. And lastly, interpret the data produced. By gathering information from several sources, the systematic review eliminates biased viewpoints with an accurate conclusion through a combined and condensed source of valuable information with a more realistic result. The limitation of a systematic review and a following meta-analysis a rise throughout the valuation steps. Perhaps certain research was overlooked? Duplication of publication? But the most common limitation is the publication bias. The work that is published often present a significant results, which undermine the opposite outcome, thus skew the reality and may produce a faulty interpretation (Ganeshkumar and Gopalakrishnan 2013, Mikolajewicz and Komarova 2019).

Numerous projects within the Western Atlantic and Caribbean basin have focused on coral conservation. Many have addressed the active part of coral conservation by examining coral rehabilitation or coral restoration through coral nurseries, coral transplants or outplanting of coral fragments (Lirman 2000, Bowden-kerby and Carne 2012, Meesters et al. 2015, Carne et al. 2016). These projects offer valuable knowledge and understanding of coral conservation and provide useful information to inspire future projects (Precht 2006, Edwards et al. 2010). This chapter examine previous research through a regional meta-analysis to 1) review methods commonly used in coral restoration and further 2) examine the heterogeneity between those methods to 3) produce a review of which of those methods that appear to produce the most sustainable result. To compare and summarise the data available published papers were scanned, and data were extracted from those papers that complied to an assortment of set criteria. Studies were selected if following information was provided, 1) year and location, 2) initial and final number of coral fragments, 3) the observed coral species, and 4) treatment method and final adhesive used. In some instances, data were not specified in numbers but rather a graph. For those instances, the software WebPlotDigitizer 4.2. was used to directly extract numbers. The data used in the meta-analysis were collected from research exclusively conducted within the Western Atlantic and Caribbean Basin (Fig. 3.1) representing 7 countries through 14 projects. The projects each investigated the success of coral growth and survival by different or similar methods over varied time intervals. Using Excel for Mac (version 16.31), the data was extracted from the selected papers and used to investigate; 1) percentage survival per species, 2) survival according to morphology, 3) survival associated with coral donor, 4) survival following nursery/no-nursery stage, 5) survival when transplanted to artificial versus natural substrate, and lastly 6) survival rendering adhesive used.

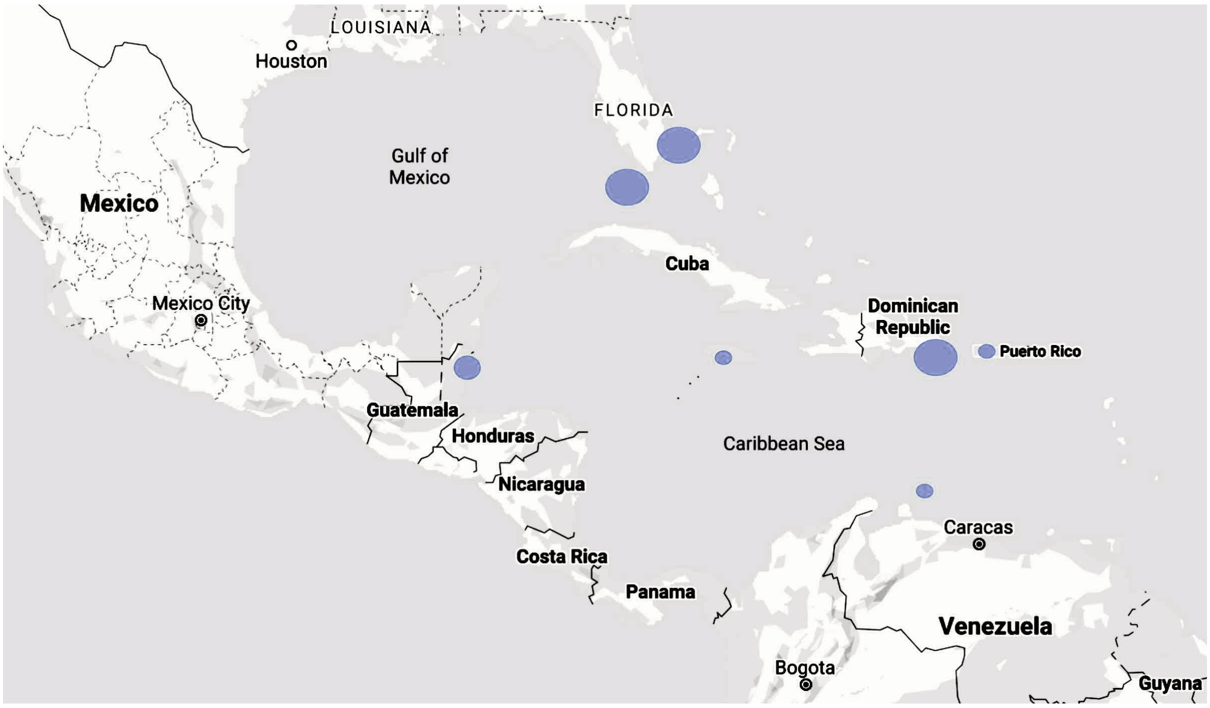


Fig. 3.1: Distribution of the regional project sites included in the meta-analysis within the Western Atlantic and Caribbean Basin. Circles show numbers at each site; small circle = 1 study, medium circle = 2 studies and largest circle = > 3 studies.

In this meta-analysis the questions investigated were; is there a common trend for coral restoration approaches throughout the Caribbean region? Which of the methodologies used are the most successful in terms of survival? A plan for exclusion and inclusion was prepared to construct a specific "reject log" (Stroup et al. 2008). Any research or data that did not meet set criteria were excluded. For this meta-analysis, data collected, and research conducted outside of the studied region was excluded. Also, research that was exclusively laboratorial based, projects that were overlapping other projects, or where the complete data lacked relevant information was discounted. Research that were included in the collection were managed within the region of the Western Atlantic or Caribbean, had a peer-reviewed published paper produced, and included data that would answer the questions stated above. Only peer-reviewed papers were included to minimising meta-bias in the analysis.

In supplement to the first edition of the systematic review another review was completed during June 2020 to include latest data from projects of 2019 and 2020. Further, for first review only Google Scholar (GS) was used as search engine, in latest review also Web of Science (WoS) was used to collect papers from. Further, for the second review data was not only extracted for the Caribbean, but also the remainder of the world. This was done to 1) investigate whether trends have changed with supplementary knowledge during the added year, and 2) to compare global trends in methodology and tools to those of the Caribbean.

3.2: Methods

To gather the available literature regarding coral restoration in the Western Atlantic and Caribbean region an electronic literature search was conducted using GS with following terms: "coral restoration" OR "coral rehabilitation" AND "Caribbean" AND "transplantation" AND "growth" AND "survival". Further was the search limited to the time interval from 1975 till 2019. The search was performed on the 22nd of February 2019. A limited number of papers (n = 4) were found at individual publishers, not mentioned through Google Scholar, they were added to the search. Each research paper was treated individually regardless whether other projects sites were bordering. Papers overlapping a previously published paper on same project were either merged or only recent data was used. All papers were evaluated unconnectedly to investigate whether a research project with the presented data accommodated set measures for inclusion. The criteria were as follow, 1) be published and peer reviewed 2) written in either English, Spanish or French 3) research conducted in the Western Atlantic or Caribbean region 4) fragments could be, at some stage, kept in a laboratory, but should predominantly be kept *in situ*, thus 5) laboratory manipulated studies were excluded. To limit duplication of data, papers offering exclusively secondary data were excluded, e.g. reviews and meta-analyses. Remaining papers (Fig. 3.2) that met the protocol were examined to extract the needed data. The initial electronical search provided 220 published papers, with an addition of four available papers found at individual publishers. These were included. The remaining papers were scanned for duplicates and/or overlapping data and eventually if they met the set criteria, were included. The papers were read in more details to determine whether useful data could be extracted. 11 papers met the criteria and were included in the review. Some papers did not include all information required but was still considered sufficient to be included, i.e. one paper did not state the year the fragments had been harvested or outplanted. The earliest papers had been published in 2001 and the latest in 2018. Coral fragments had been harvested between 1996 and 2013, and outplanted between 1997 and 2015. Most studies had been monitored for more

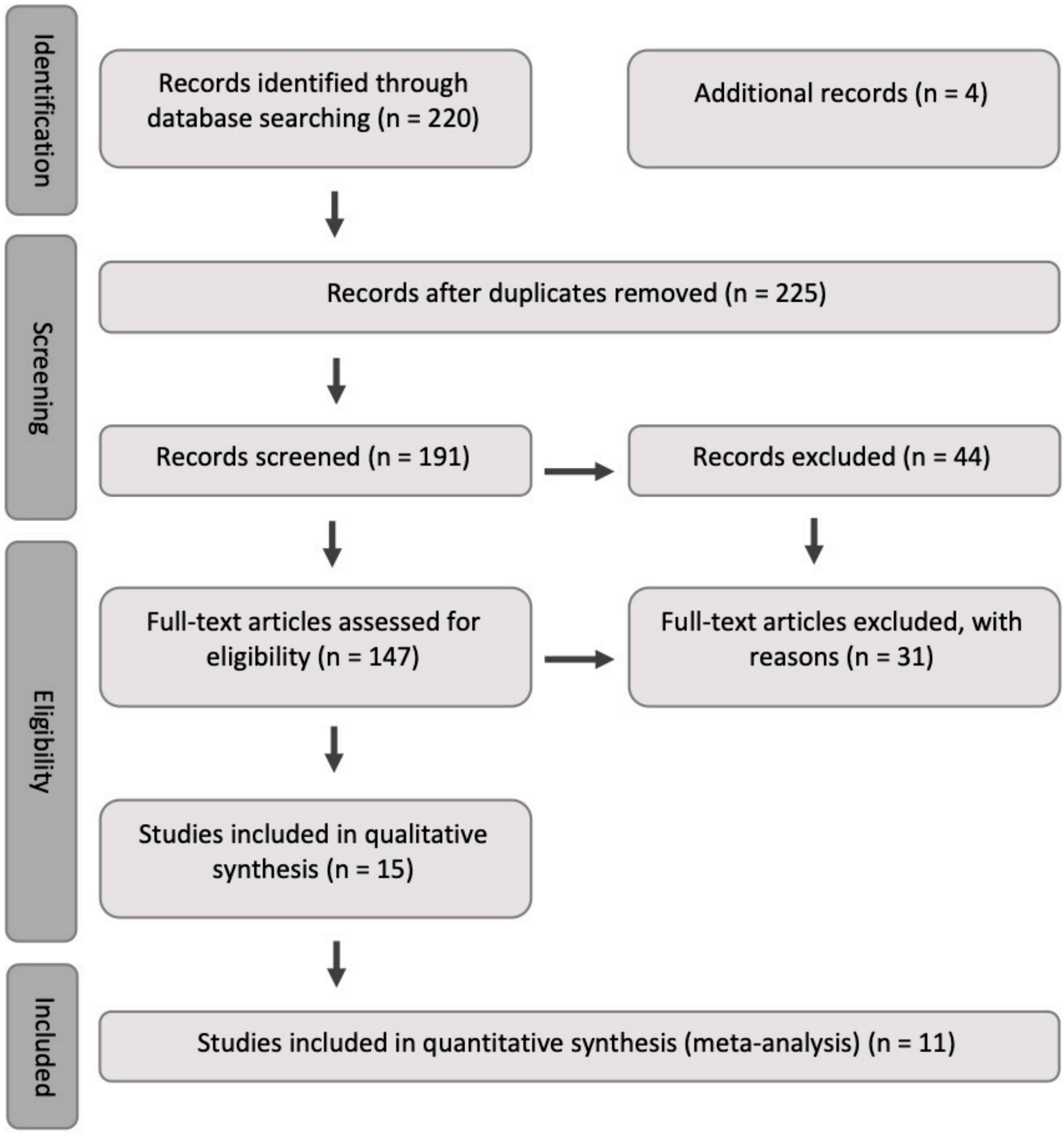


Fig. 3.2.: A study specific flowchart structured (papers listed in Appendix 2) according to a PRISMA (Preferred reporting Items for Systematic Reviews and meta-analysis) template to select the papers that meet the requirement for the meta-analysis conducted (Flow chart inspiration by (Claar et al. 2018)).

than one year but less than 5 years. Fragments taken directly from a storm affected reef with fragments freely available on the floor were merged into a category, “Corals of opportunity” (COO). The category named “Nursery” (N) are coral fragments outplanted following a period in a coral nursery, where the fragments originated from prior to the nursery was not differentiated. Lastly, the category of “Wild population” (WP) was fragments harvested directly from a wild growing donor colony.

Research for the second edition of the systematic review was collected from both GS and WoS on the 10th of June 2020. For GS following search criteria were used; “Coral Restoration” OR “Coral Rehabilitation” AND “Survival” AND “Hard Coral” covering the period from 1975 to 2020. The search results presented 219 papers (One paper less than the search for previous systematic review). For WoS following criteria were used; “Coral Restoration” OR “Coral Rehabilitation” AND “surv*” AND “Hard Coral”, also for the period 1975 to 2020. A total of 98 papers was suggested. Same procedure as first review was followed with an exclusion of duplicates and papers that did not follow set parameters (Fig. 3.3). For the first review postgraduate research had been excluded from the data, whereas in the second review any Doctor of Philosophy research was included to broaden the available data. Any other postgraduate projects were still excluded. Many research projects were excluded from the remaining region due to insufficient data or due to unrelated focus of subject; numerous projects focused on coral coverage or sexual restoration. These studies were therefore excluded. A total of thirty-four research papers were included for the second review separated into two groups, the “Caribbean” (n=25) and the “Other” (n=9), including six projects from the Indo-Pacific Ocean, one project located in the Red Sea and One in the Arabian Gulf.

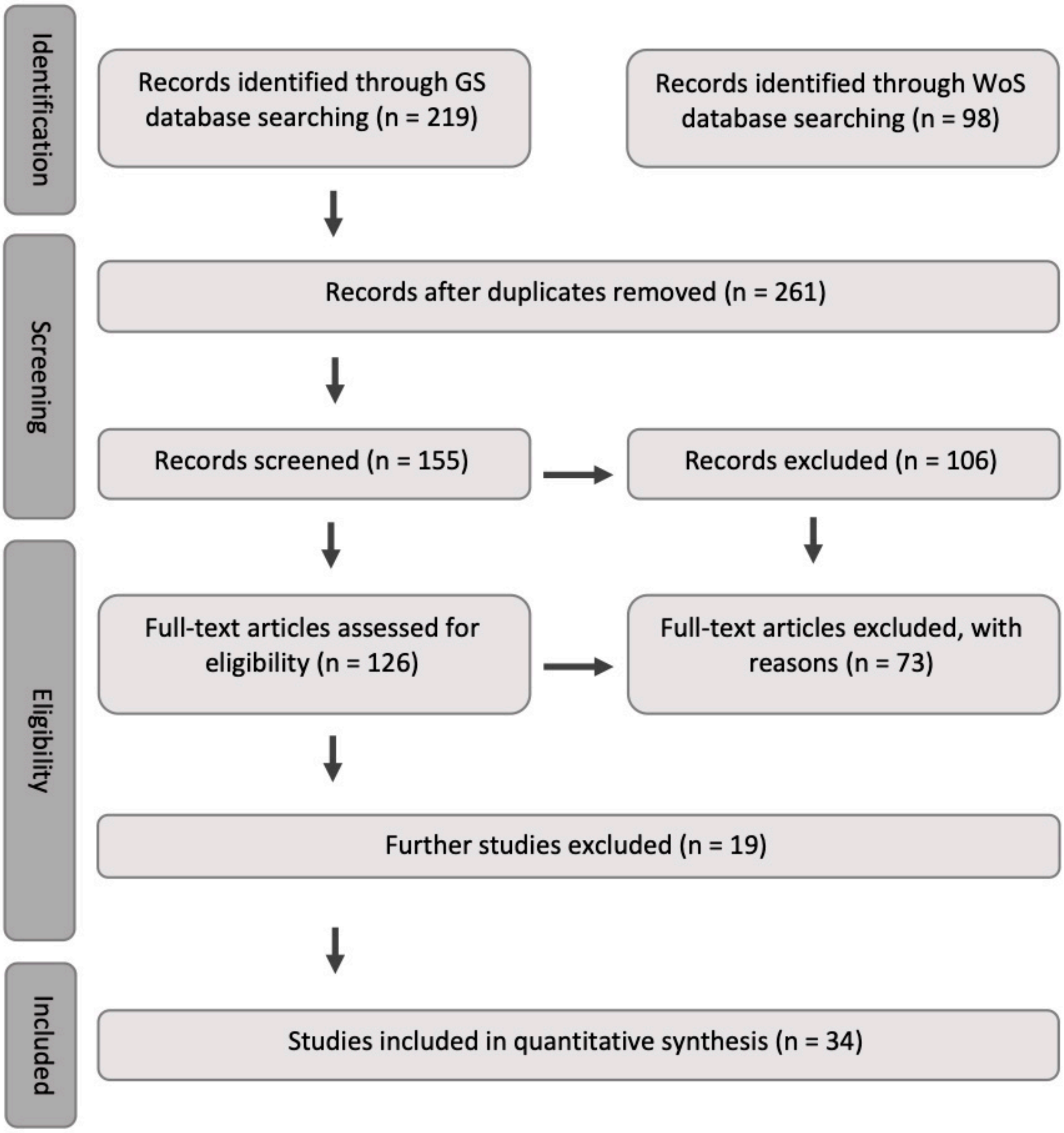


Fig. 3.3.: Showing the original search results of both search engines used for the second review (papers listed in Appendix 3), with the following screening and elimination of gathered studies (Flow chart inspiration by (Claar et al. 2018)). The included papers (n=34) all contributed with data that complied set criteria.

3.3: Results

The overall mean survival rate (Fig. 3.4) across the investigated projects were 83.5 ± 27.64 %. Bonaire had the highest survival rate of 100 %, a result of one study which exposed all the outplanted fragments of *Acropora cervicornis* (N = 180 fragments) to identical conditions. Lowest survival rate of 25 % was from Jamaica, calculated across three variables of different outplant methods, with a total of 451 ± 95.71 fragments in the study, all of the species *Acropora cervicornis*. The methods investigated were cement (18.96 %), cable ties (30.25 %) and lines (32.00 %), last 18.79 % was unaccounted for. United States of America contributed with the most studies, in total six studies were included with a total amount of 1151 fragments.

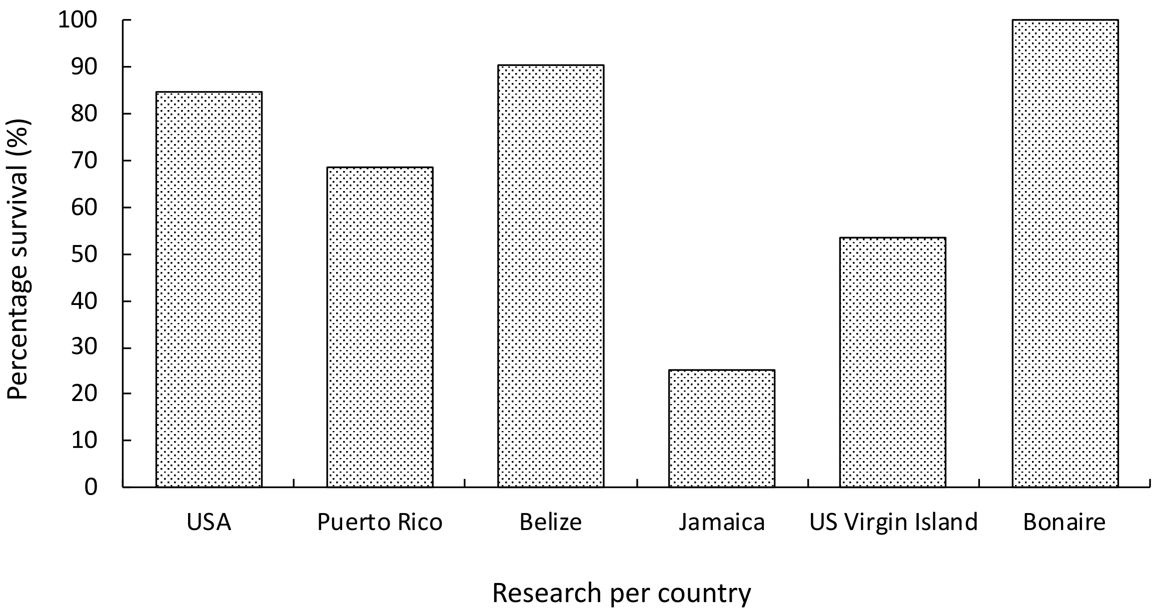


Fig. 3.4: Percentage survival per country included in the systematic review. The overall survival rate of all studies combined was 83.5 %, with the lowest score in Jamaica of 25 % (n = 113), and highest score in Bonaire of 100 % (n = 180). USA contributed with the greatest numbers of projects (n = 6), a total of 1151 coral fragments. The american studies covered both variation in the adhesive used, source of fragments although predominently harvested from a nursery or the wild one study included corals of opportunity. The studies also varied acrross coral species and covered both branching and massive morphology. Puerto Rice contributed with two projects and a total of 413 coral fragments. Both of these studies used fragments that were taken from a wild donor and outplanted onto an artificial structure, using cable ties to securre the framents. The studies only included acropoirids. Remaining countries only contributed with one study, ranging from 60 (Virgin Island) to 4168 (Belize) coral fragments per study, leading to an average of 1215 (± 1976 coral fragments) per projects. The studies only included acropoids, expect Virgin Islands which also contributed with some *Porites porites* (n = 15). Virgin Islands was also the only project to include corals of opportunity as donors and outplant directly onto substrate. The remaining countries only used wild donors and outplanted the fragments onto an artificial structure (i.e. cement discs, A-frames).

Exploring the data extracted for the second review, considering the remaining region, Tanzania contributed with ~ 76 % of the total fragments used in the group. When that project is included in the distribution of fragments disperse almost equally between the “Caribbean” and the “Other”, with the Caribbean contributing with ~45 % to the total amount of fragments used. Considering same data, but excluding data from Tanzania, the Caribbean supply with ~ 78 % of the total fragments used in global coral restoration projects. Comparing the two groups they show similar survival rate, respectively 75.76 % and 72.58 % for the “Caribbean” and the “Other”. Both groups contain one project each with a dominant proportion of the total use of fragments, which did above average.

When considering the species in their lifeform (Fig. 3.5), separated into either branching or massive, the often faster growing and more fragile branching coral had a mean survival of 83, SE ± 23.7 % whilst the slower growing massive coral had a mean of 85, SE ± 25.6 %. The specific species used included *Acropora cervicornis, Acropora palmata, Colpophyllia natans, Diploria labyrinthiformis, Pseudodiploria strigose, Dichocoenia stokesii, Meandrina meandrites, Siderastrea sidereal, Orbicella faveolata* and *Montastraea cavernosa* with an overall survival rate of 84.62 ± 20.61 %. One study from Belize contributed with more coral fragments than all of the other studies combined, a total of 4168 fragments and a survival rate 90.43 %. The studies included ten hard coral genera counting 12 species. The tree species *Montastraea cavernosa, Meandrina meandrites* and *Siderastrea siderea* showed a 100 % survival rate (Fig. 3.6), all

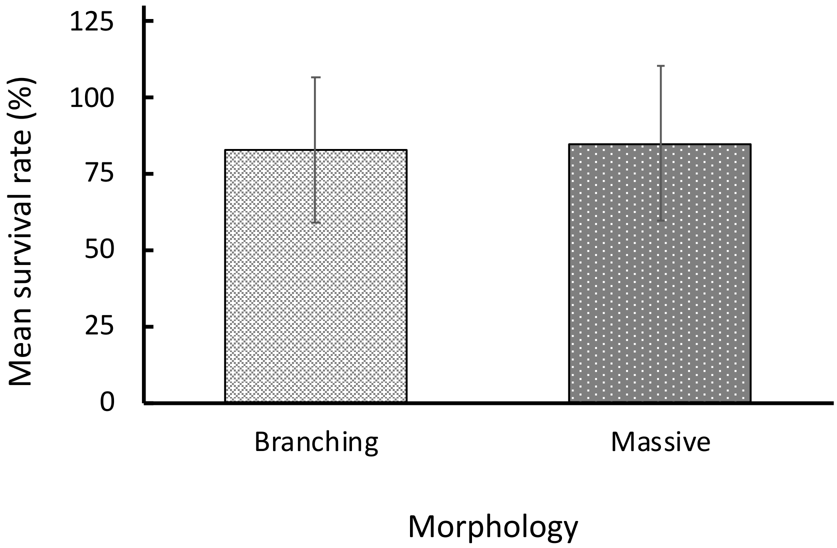


Fig 3.5: Comparing survival rate between the two lifeforms branching and massive hard coral colonies. Massive showed a higher survival rate when compared to branching colonies. The projects, including all six countries within the Caribbean region, started off with a total of 6242 coral fragments of the branching type and 181 for the massive. The branching group showed a survival rate 83 ± 24 % , and the massive group a survival rate 85 ± 26 %.

categorised as massive lifeform. The *Montastraea cavernosa* was included in a study from USA and kept under *ex situ* conditions during its nursery phase and outplanted directly onto substrate with the use of an array and attached with epoxy. In same study was used one further species of same lifeform, kept and outplanted under same conditions, the species was *Orbicella faveolata*. The survival rate in same project was 80 %, the overall rate across three studies was 73 %, ranging from 17 % - 100 %. The *Meandrina meandrites* and *Siderastrea sidereal* with 100 % survival rate were collectively included in another study from USA, collected from a wild population and reattached directly onto substrate using cement. Focusing on the genera *Acropora spp.*, here including all species represented within the Caribbean region, the hybrid species, *A. prolifera*, has the highest survival rate (91 %), whilst *A. palmata* has the lowest (76 %). Two common species, *Pseudodiploria strigose* and *Porites porites* show respectively a slightly higher (86 %) and slightly lower (73 %) than average survival success (Fig. 3.5). The species that scored the lowest survival rate was the *Colpophyllia natans*, which data was collected from another North American study containing 8 fragments with 5 surviving.

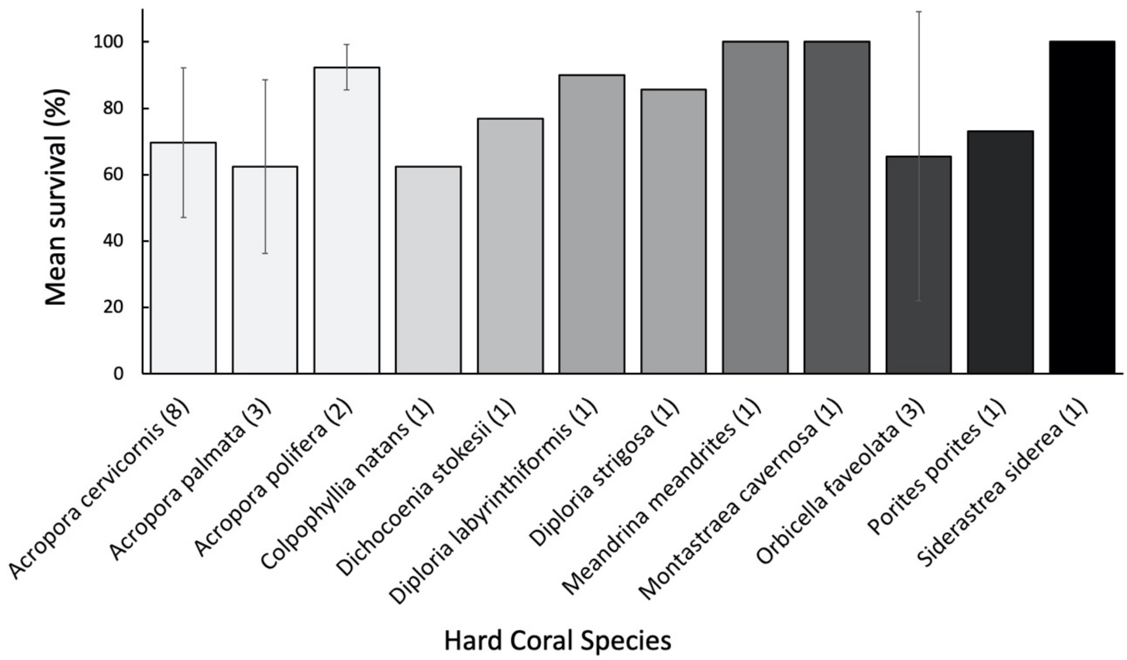


Fig. 3.6 The percentage survival across 8 genera and 12 coral hard coral species. Individual grey tone delegated each genus. Some species does not show standard error due to limited data. Parentheses following the spcies name are numbers of times the particular species was included in one of the reviewed studies (n = 11). Sample size varied from 7 to 5175, with the average of 535 ± 1484 coral fragments and median of 14.

Considering the difference in distribution of corals of massive or branching morphology the remaining region show greater consumption of massive species in regard to active restoration, whereas the Caribbean most often uses branching species (Fig. 3.7). This trend corresponds with 88 % of the considered projects within the Caribbean included either one, two or all of the acroporid corals present in the area. Considering the survival rate across morphology the Caribbean scored ~ 2 % point better than the Other for both massive and branching coral species.

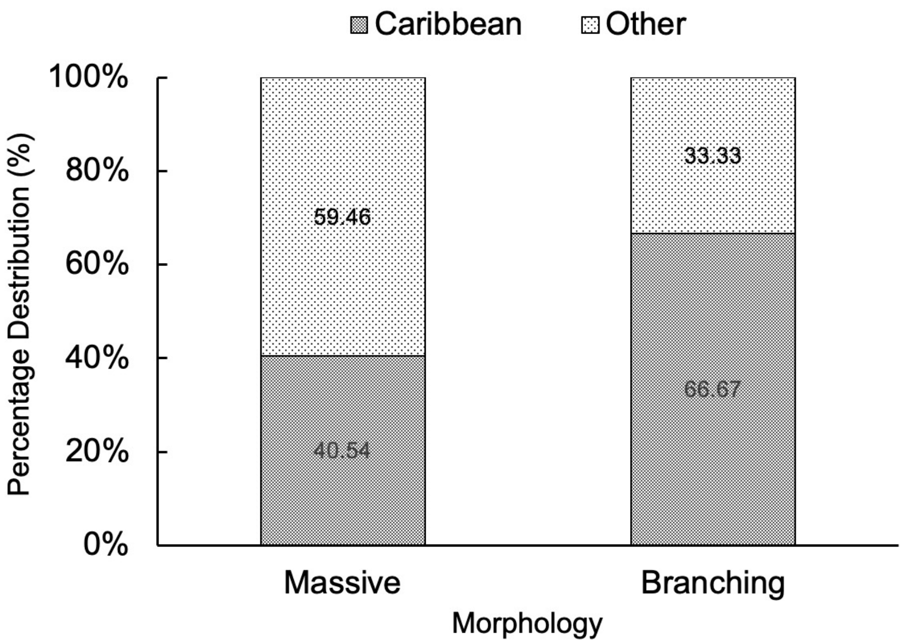


Fig. 3.7: Distribution of morphology across the two considered regions, the Caribbean and the Other. The Caribbean show a greater usage of the branching species (66.67 %) whereas the Other (59.46 %) show the greatest use of massive

In the comparison between place of origin of donor fragments (Fig. 3.8) and survivability the data was divided into the three categories, 1) Coral of Opportunity (COO), 2) Nursery (N) and 3) Wild Population (WP). The data showed that the COO fragments had the lowest survival rate (71.14, ±14.84 %), whereas the fragments collected from a nursery showed the highest survival percentage (90 ± 14.14 %). Dividing the origin of donor into lifeform showed fragments of WP had a slightly lower survival rate for the massive lifeform compared to the branching. Unfortunately, further investigation could not be explored due to between COO and N, due to lack of data in opposite group.

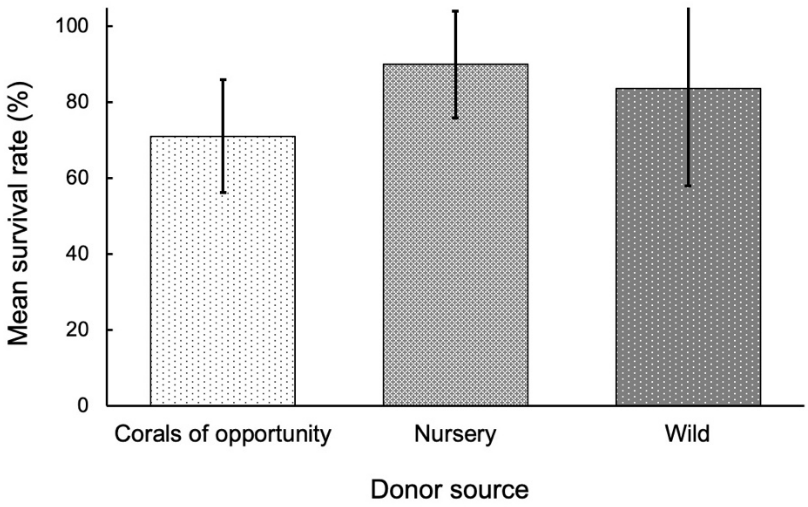


Fig. 3.8: Percentage survival of outplants originating from coral of opportunity, nursery grown and ramets collected from live coral colonies, no considerations of species, lifeform etc included. Nursery grown fragments showing overall greatest survival rate with a narrow range in outcome when comparing to the other two groups. Fragments collected from live wild growing colonies show the greatest range in survival, whereas corals of opportunity show lowest survival rate.

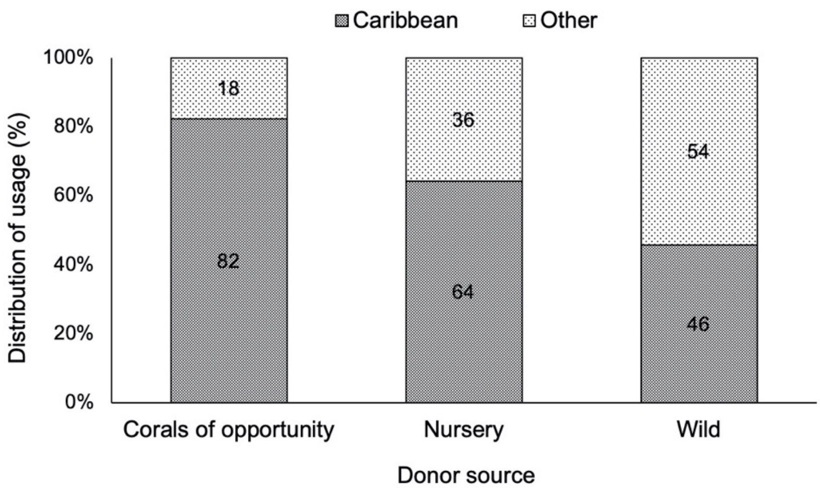


Fig: 3.9: The distribution of usage of fragments from each of the three sources, corals of opportunity, nusery gwon or harvested from the wild colonies. For both the Caribbean (Caribbean) and the other regions across the world (Other) an identical preference of donor transpired. Coral fragments harvested from wild coral colonies occured 59 times in all the projects considered, slightly dominated use in the other locations outside of the Caribbean, accounting for 54 % of the projects. Secondly, fragments harvested from nurseries (n=28) and last, the collected coral of opportunity (n=17). Projects conducted within the Caribbean used coral of opportunity almost 4.5 times as frequent as other parts in the world with a total of 82 %. Caribbean also dominated in the use of fragments grown in and harvested in nurseries (64 %).

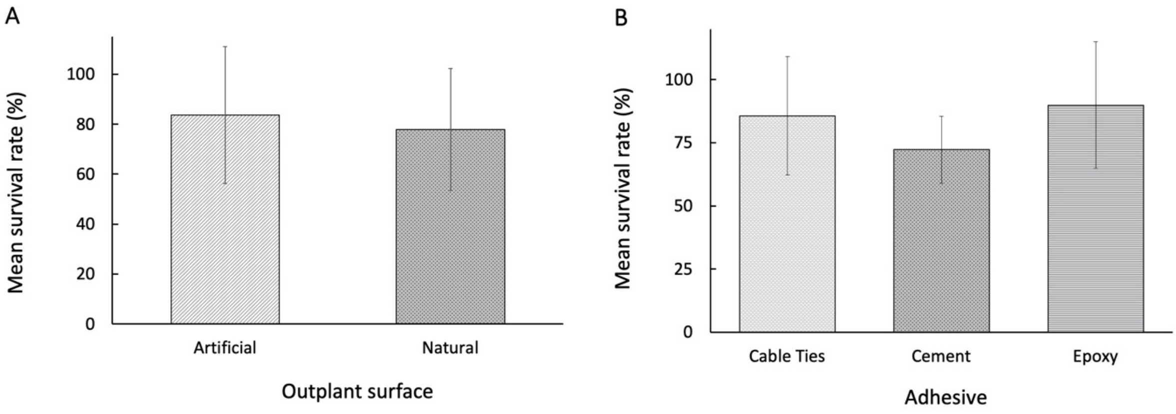


Fig. 3.10: Comparing the survival rate with the type of substrate used for outplanting and the adhesive used to attach the coral fragments to the substrate. **A)** A higher survival rate was revealed for fragments outplanted onto artificial frames (e.g. A-frames, cement blocks) when compared to fragments outplanted directly onto a natural substrate (e.g. dead coral reefs, rocks). **B)** In comparing the three types of adhesives epoxy showed the best result, follwoed by cable and lastly cement. Lost or dead fragments were not differenciated.

3.4: Discussion

The aims of the meta-analysis were to determine the differences in survival rate between regions, lifeforms and species. Further, whether a difference in survival could be detected between the origin of donor colony, the substrate the fragments had been outplanted onto and which adhesive had been used. Finally, in the second review, data was investigated to compare results between the Caribbean and the remaining world.

First analysis showed a difference between regions within the Caribbean, with Bonaire having best results of survival rate and Jamaica the lowest survival rate. Second lowest was US Virgin Island, followed by Puerto Rico, USA and Belize close the Bonaire in survival rate. Data extracted from each region varied in number of research projects. USA contributed with the most studies, in total five. Puerto Rico with two and lastly Belize, Bonaire, Jamaica and US Virgin Island all with only one study. This could skew the result and display an unrealistic trend. Had the projects been evenly distributed across regions, with similar number of single fragments, the results of this finding could be weighed higher. Comparing lifeform, the results showed little difference between branching and massive colonies. This could be due to either, there were less data on the massive lifeform than the branching, thus could skew the results. Also, it could testify of restoration methods being more well adapted to handling the slow growing massive lifeform.

The complexity of coral reefs has been linked to species richness, abundance and biomass. A degradation of complexity may affect the diversity of associated species inhabiting the coral reefs causing a shift in composition with ecological and socio-economic consequences (Alvarez-Filip et al. 2009, Yanovski and Abelson 2019). Considering the survival rate between species, solely focusing on species of the Acroporids showed a higher survival rate amongst *A. prolifera*, followed by *A. cervicornis* and last *A. palmata*. This could perhaps be related to a greater genetic diversity of the hybrid *A. prolifera* when compared to the two remaining species, two genetically distinctive species (Vollmer and Palumbi 2007). Of the remainder species, the three species which had a 100 % survival rate was *Montastraea cavernosa*, *Meandrina meandrites* and *Siderastrea sidereal*, the latter two included in same research project. All three of massive lifeform, from a USA based project. Comparing the donor colony, fragments collected from *in situ* nursery showed the overall greatest chance of survival when reintroduced to the coral reefs. This could imply a low level of stress of the corals prior to the outplanting, making the corals more resilient to change. Research show that *in situ* often provide a safe environment for the coral fragments to grow with a low level of corallivores praying on the fragments, a low level of sedimentation as nurseries are often placed midwater or on a table like structured elevated above the seafloor, some studies have even produced spawning coral in fragments that were grown in *in situ* nurseries and outplanted onto coral reefs (Carne and Baums 2016). These optimum conditions have shown to enhance growth, thus leaving the coral fragments better equipped to combat stressors, e.g. diseases, sedimentation (Afiq-Rosli et al. 2017). Whereas, the fragments Collected COO for replacement onto the reef showed the lowest level of survival, which would relate to other research stating that these coral fragments experience a high level of stress due to the tumult existence on the sea floor being dragged across the substrate through the water movement, leading to smothering and damage. Further, these fragments may also experience either an extreme increase or decrease of light and UV levels. Investigating the difference in survival of fragments outplanted on an artificial substrate contra natural favour the artificial substrate (78 ± 22 %) compared to the natural substrate (72 ± 25 %). This could be related to better possibility of attachment, or the structure is placed elevated above the substrate thus decreasing chances of sedimentation and smothering of the fragments. Further, some restoration project now uses a method where the fragments are relocated to onto the artificial substrate whilst still attached to the tiles or ropes from the nursery. Thus, the outplanting occur without adding the stress of detaching the fragments from their placement in the nursery. That could reduce stress, thus increasing chances of survival. By evolution corals will adapt to new levels and endure in their own way a new structure. Result being an alternative ecology but a healthy and naturally modified reef. In a time where anthropocene is absent, the argument is highly rational and probably the most prosperous solution. Unfortunately, what the massive decline within the Caribbean region showed throughout and following the 80s is that due to human impact and varied sources of pollution there are simply too many stressors for the coral reefs to naturally adapt and withstand the trials introduced at present time. We will risk the coral reefs dying off before having the time to adapt. Thus, an argument for persisting on an active interfering in assisting coral reefs has been pushed forward. In addition to current projects, many studies included in both reviews disregarded the use of control groups. Especially when harvesting wild donors, it would be interesting to install a control group in site of origin whilst transplanting other colonies. Having control groups could highlight potential harm that the disruptive action of harvesting and replanting could cause. This would add considerable value to our knowledge and perhaps present reasons for final results in projects.

However, what is noticeably evident when considering papers and projects regarding coral restoration, is the need of expanded collaboration and communication, and perhaps a mutual decision, in a standardised methodology when launching and presenting data of the matter.
During the process of extracting papers relevant to the subject investigated for the meta-analysis of this thesis several general impediments became apparent. First, there are many current projects regarding coral restorations but only few papers where useful data can be extracted for a comparison between similar projects, or projects that focus on gathering data for a baseline on the general health of the Caribbean reef.
The meta-analysis explored the data available on coral restoration within the area to scrutinise possible relationships between methodology and end-result. But whereas there are plenty of papers on the topographic evolution of the reefs, complications during the degradation during recent decades, papers addressing current development and gathered knowledge on active restoration lack constancy in representation of location, origin of coral fragments, numbers used at origin and during the period of the project and final results. In other events, the methods are given through a descriptive text that fails to present detailed information, which may cause

errors of interpretation. In other words, the deficiency of well-defined information may lead to faulty interpretation thus leaving out details and possibly let valuable experiences go unnoticed. A suggestion from the experience gained through this thesis; As an appendix, to add a standardised form to studies regarding active coral restoration A collective protocol designed to acquire consistent information via any funded projects. These protocols could at first be developed in each country's CZMU until fully integrated, hereon after perhaps a regional organisation could be launched. A more readily available data base on coral restoration results would possibly reduce expenses and valuable knowledge could be shared. A suggestion for such a form has been added to this thesis as the "Standardised Coral Restoration Form" (Appendix 4).

Chapter 4: General discussion and recommendations

Through the results of both the case-study and the meta-analyses, it is evident that coral restoration requires a wide range of knowledge and detailed planning to obtain best result possible. In order to adjust project ambitions an estimate of the historic composition and events should be examined, e.g. through questions considering following aspects:

- Is there available documentation or data of the reef prior to its deterioration?
- What were the causes of the damage? Anthropogenic? Natural causes?
- Have these issues been addressed? If so, have they been resolved?
- Are there any signs of natural habitat improvements, such as recently settled juvenile coral colonies?

If no improvement has been observed in the area, any attempt of coral restoration will almost certainly be wasted resources. Time and finance may be better invested through alternative approaches. Managers may overlook this pragmatic view in their determination, launching the project prematurely, thus giving themselves poor odds of success and leave investors and initiators with a feeling of over empowerment and demotivation. Instead, implementing passive restoration, before the active phase, may yield greater long-term success. A combination of passive restoration followed by active restoration has through the last decade become more common (Shokry and Ammar 2009) than previously. Should the project proceed and advance to the active restoration, further questions could be addressed:

- Have any prior studies been conducted on the particular reef? Or neighbouring sites?
- Have these produced any published reports or papers on the results?
- What were the aims and methods? Similar to the current project?
- What was the success rate? How was this measured? On growth rate? Survival percentage?
- Were pre-estimated goals met? What is the status of those projects at present time?
- Did the previous project provide knowledge or experience that the current project may benefit from?

This investigative research should be completed prior to, or at least in the early stages of, applying for funding or deciding on which direction to steer project. Based on the retrieved information, the next step should consider and decide:

- Considering goals, is the project focusing on coral restoration?
- Or coral rehabilitation? To improve the coral coverage? Or a combination?
- Who is the project aimed at? Fishery? Tourism? Or to add intrinsic value?
- Perhaps to assess a newly developed coral restoration or coral rehabilitation methods?

Have in mind, testing a new methodology or managing coral restoration/rehabilitation are both areas that require a considerable level of time and finance. Focusing on one of the two at a time may be more efficient. Embracing too great a task in too little time may inhibit proper focus leaving none of the tasks to be addressed fully.

When trying out new methods, or when choosing which is best for a given site or corrals species, the first decision should be whether the project will be constructed in or out of the water. Land based methods are often more expensive due to the technical requirements and continuous maintenance. In comparison, an in-water approach can offer a setup at lower cost and maintenance, whilst, provide a higher level of success of growth and survival. Regardless, a land or in-water based study, deciding which specific tools the method will require is the next step.

- Does your method require expensive gear? Where can you acquire these tools?
- Will staff members require additional education on techniques or safety measures?
- What is that cost of those?
- Wherefrom will coral fragments be collected? If collected from wild coral colonies, consider the conditions in which these fragments will be placed in comparison to where they originate. Are there any variables you will need to consider? May these affect your end results? What species are you using, and why?
- Will you have to mitigate certain conditions to improve the well-being of the fragments?

Investigate any resources that may provide information on the species and their optimum environments. Will you need to acclimatise collected fragments? Also, include a consideration for impact on donor colonise. Are these colonies healthy without signs of stress?

Consider the statistical approach in the early phase of preparation will make it possible to consider control group and variables. Here are some questions to consider:

- If the study is based *in situ*, is it possible to construct a control group at the site of origin? When *in situ*, what variables may impact the control group? If the control group is located *ex situ* under controlled conditions, which variables need to be constant?
- What statistical analyses are possible with the data available? How can you manage your study to produce a strong data set with high quality analytic results?

Lastly, a step that often appear disregarded, through a cost-benefit analysis, will the estimated cost of the project merit the benefits gained from the project?

The support CORALL received from the general population, stakeholders and the current Barbadian government suggests a collective attitude of encouragement towards coral rehabilitation in Barbados. From interviews and conversations conducted between the spring of 2018 till the summer of 2019, the project appeared to aspire from an idea that evolved within the government; The establishment of a coral nursery to assist local coral reefs to increase coral coverage with an opportunity of economic gain for local fishery and tourism. Although, evaluating the launch and management of CORALL two issues in particular seemed to cause challenges and an ultimately strain on the project; communication and planning. More specifically, following issues were revealed from the people involved:

1. **Management**. From the very beginning, the project lacked a clearly defined management team, which eventually caused frustration within the team causing people to abandon the project and leaving CORALL with an ambiguous direction during its initial phase. This resulted in a high level of expenses in the early phase of the project which later limited the budget and restricted the flexibility of adapting to unforeseen events (i.e. repair of laboratory, production of additional fragments after first generation died following outplanting).
2. **Planning**. An absence of a clearly defined time-schedule and reasoning of decisions meant that the staff members felt frustrated, ignored and confused. This led to

unorganised preparation and response to events in the laboratory during the outplanting phase and later during monitoring. Eventually causing more staff members to abandon the project.

3. **Focus**. The projects dispersed across three different outplanting sites. These sites were chosen without prior observation or testing of water quality, differences in abiological and biological conditions or without considering what the geographical distance would later demand in regard to finance and staff capacity. Instead, these factors were realised and planned for as the project progressed.
4. **Data.** The absence of a consideration for and integration of the variations between sites meant a comparison of data growth and survival would generate a statically weak results due to the number of variables (i.e. substrate type, fishery pressure, inside vs. outside marine reserve). Further, did the project not incorporate a control group, which meant that it has not been possible to determine whether the mortality rate was due to treatment in laboratory, outplanting methods or conditions at the *in situ* sites.

Although coral larvae settlement were present at the sites, suggesting a level of favourable conditions, Barbados, like other islands in the region, struggle with a reoccurring invasion of *Sargassum spp.*. Additionally, the island has struggled with a sewage system incapable of accommodating the increasing numbers of tourists visiting each year. Those factors plus a high pressure of runout from chicken farms and general agriculture impact the surrounding water quality. Coral colonies overgrown by turf algae were spotted at the three sites and could indicate the presence of, at least periodic, increased levels of nutrients. While the water quality test results included in this study showed values considered within healthy levels, the tests were only done once, during the low season of tourists, and only after the ramets had been outplanted. Preferably, to monitor changes in water quality, these tests should have been conducted throughout a period leading up to the project launch before deciding on which locations were suitable for outplants. Previous studies have shown nutrients and sedimentation (GESAMP 2001, Fabricius 2005, Wooldridge and Done 2009, Wooldridge 2013) to have a significant impact on coral health. Collecting water samples and running quality tests could have provided answers to whether the *Sargassum spp.* and nutrient levels were of concerns at the outplant sites and whether adjustments (i.e. relocate outplants sites, schedule outplant at a different season) should be done to increase chances of growth and survival success.

Considering an integration of preliminary and continued ecological assessments would equally have provided knowledge of fluctuations in the abiotic and biotic habitat. These supplementary assessments which could have provided information for the projects to likewise adapt to (i.e. matching coral species with suitable location) for potential increased success. These assessments would further have offered knowledge of the ecological composition before, during and after the coral restoration, an additional measurement of success.

Comparing the development and results of CORALL to other projects confirm similar experiences. Previous studies show that the human interaction can either restrain or enrich a project outcome (Fletcher et al. 2015). Projects may be rushed through the planning process to quickly launch and accommodate restricted time or budget plans and. Or projects may be steered by thinkers that in large neglect the existing source of scientific knowledge whilst scientists disregard the creative thinker's innovative understanding. Other projects are based on a premise to restore damaged coral reefs back to historic state. That goal has proven to be a task practically impossible. Spending time and finances trying to achieve something almost impossible may disrupt a more realistic goal. Expectancies benefit from a realistic ambition. Consequently, a future bearing coral rehabilitation rather than coral restoration may be more sensible. Having a standardised procedure for any coral restoration or rehabilitation study could potentially encourage appropriate communication and planning, potentially reduce challenges within the management team, and steer focus from personal ideas or ambitions to collective goals.

A recommendation for future projects in Barbados is to learn from the experiences gathered through this case-study and learn from its weaknesses and value thorough communication and planning. A list based on the ideas of the management tool "The SMART objectives" (Doran 1981) has been produced to incite a pragmatic approach (Fig. 4.0). The example given here is a general list of objectives to consider during the early phase of a project. Simplified suggestions to reduce challenges like the ones mentioned in this chapter. As a project progresses the objectives can be moulded and adapted to new events or challenges.

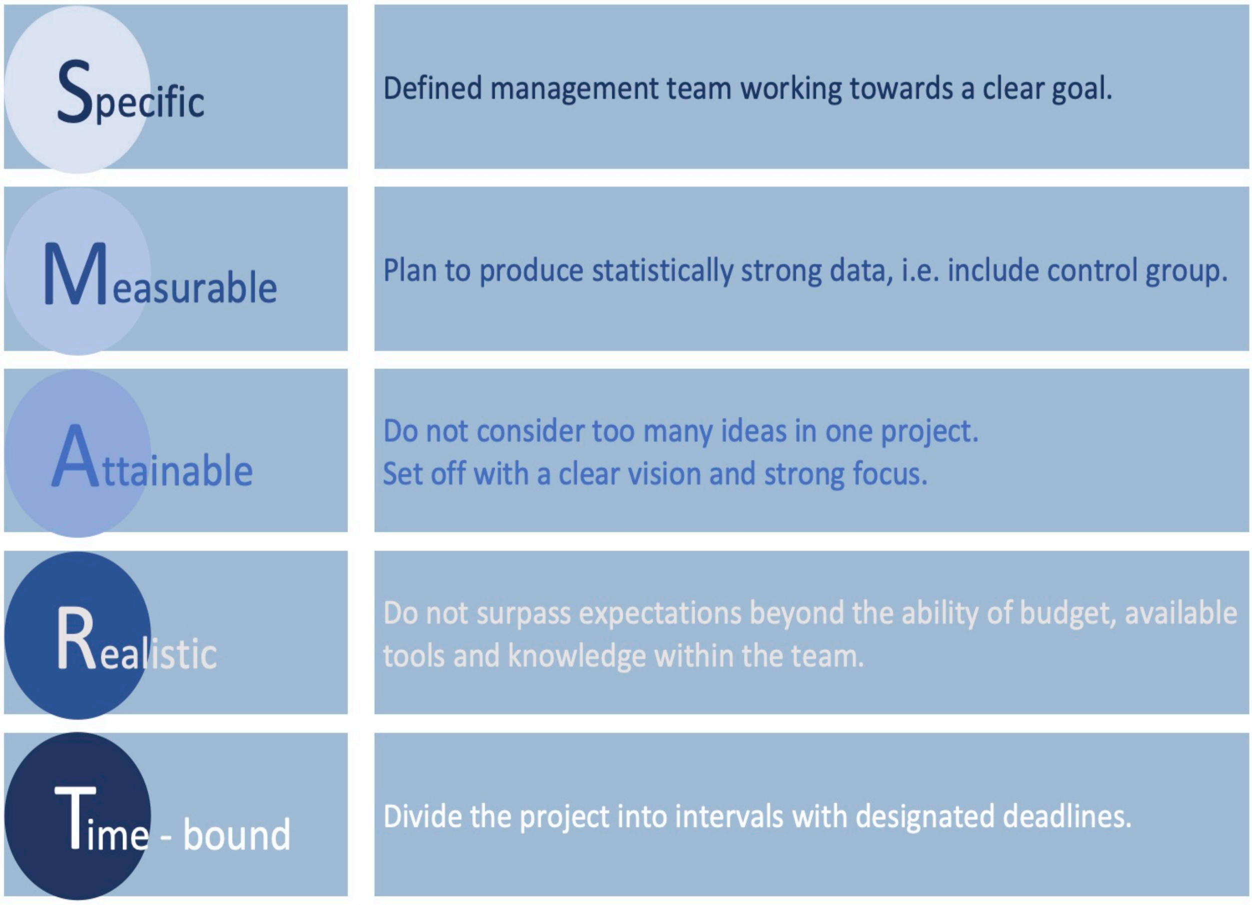


Fig.: 4.0: Inspired by the project management tool SMART (Doran 1981), this general diagram show suggestions for ways to improve communication and planning when developing a coral restoration project. The diagram can be adapted to individual projects and specified to each management group or period of schedule.

Previously, passive restoration was the most common approach when assisting degraded coral reefs. As active restoration evolved a combination of the two practices appeared to be an ideal method and is now commonly the most accepted solution (Possingham et al. 2015).
Considering trends further into the future, newer advanced techniques could offer further support to coral reefs. One example are the new genetically enhanced corals, referred to as “super corals” (Camp et al. 2018). In these experiments, corals are kept under conditions similar to those in the future oceans. Through selective breeding these studies seek to reinforce transgenerational plasticity and produce coral stocks of increased resilience and stress tolerance (Van Oppen et al. 2011, 2015, Torda et al. 2017). Other studies shows encouraging results by using an acoustic enrichment method to attract fish larvae for settlement (Gordon *et al.*, 2019;), which could have the potential to assist coral reefs through a more indirect measure.
Most studies gathered for the meta-analyses were scheduled to last less than one year. A few were planned for five years.

As projects increasingly include both the passive and active aspect of restoration, or rehabilitation, it would be reasonable to extend the timeframe most projects follow today. Depending on existing conditions of the area a passive phase would require some time for planning and implementations of laws and regulations, along with adaptation of local attitudes and change in behaviour (e.g. relocation of fisher if restrictions are added). By combining and planning for both the passive and active process in early stages of a project, time may be utilised for thorough planning of the active phase, perhaps even with minor pilot project, whilst the passive process is established. This could possibly leave time for incorporating some of the new methods earlier mentioned. It would be interesting to gather multiple methods to strengthen positive results.

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Appendix 1: Species and substrate recorded in the ecological assessment

Invertebrates
Hermit crab
Christmas tree worm (*Spirobranchus spp.)*
Feather Duster Worm (*Sabellastarte magnifica*)
Feather stars (Crinoidea)
Sea Stars (Asteroidea)
Brittle stars (Ophiurida)
Long Spined Urchin (*Diadema antillarum*)
Pencil Urchin (*Eucidaris tribuloides*)
Sea Cucumber (Holothuroidea)
Coralliophila (*Coralliophila abbreviate*)
Nudibranch (Nudibranchia)
Lobster (Nephropidae)
Fireworm (Amphinomidae)
Impact (i.e. rubbish, anchor damage, fishing gear)
Substrate
Rock
Hard Coral (Scleractinia)
Soft Coral (Alcyonacea)
Sand
Silt
Rubbles
Sponge (Porifera)
Other, fire coral (Milleporidae), anemone (Actiniaria), etc.
Fishes
Angelfish (Pomacanthidae)

Butterflyfish (Chaetodontidae)
Atlantic Lizardfish (Synodontidae)
Sergeant Major (*Abudefduf saxatilis*)
Brown Chromis (*Chromis multilineata*)
Grouper (Serranidae)
Fairy Basslet (*Gramma loreto*)
Sand Perch (*Diplectrum spp.*)
Grunt (Haemulidae)
Parrot Fish (Scaridae)
Wrasse (Labridae)
Goby (Gobiidae)
Blenny (Chaenopsidae)
Cowfish (*Acanthostracion spp.*)
Goatfish (Mullidae)
Trunkfish (Lactophrys)
Trumpet (*Aulostomus maculatus*)
Surgeon (*Acanthurus spp.*)
Stonefish (*Scorpaena plumieri*)
Squirrel Fish/ Soldier fish (Holocentridae)
Jacks (Carangidae)
Pipefish (Syngnathidae)
Pufferfish (Tetraodontidae)
Morrey Eel (*Gymnothorax spp.*)
Flounder (*Bothus spp.*)
Spotted eagle ray (*Aetobatus narinari*)

Appendix 2: List of papers included in first review

Author	Title	Year
Afiq-Rosli, L., Taira, D., Loke, H. X., and Toh, T. C.	In situ nurseries enhance coral transplant growth in sedimented waters	2017
Al-Horani, F. A.	Sustainable Resources of Corals for the Restoration of Damaged Coral Reefs in the Gulf of Aqaba, Red Sea	2013
Becker, L.C. and Mueller, E.	The culture, transplantation and storage of Montastraea faveolata, Acropora cervicornis and Acropora palmata: what we have learned so far	2001
Bowden-Kerby, A.	Restoration of threatened Acropora cervicornis corals: intraspecific variation as a factor in mortality, growth, and self-attachment	2008
Bowden-Kerby, A. and Carne, L.	Thermal tolerance as a factor in Caribbean Acropora restoration	2012
Bright, A.J., Miller, M. W. and Bourque, A. S.	Tracking growth and survival of rescued boulder corals	2016
Calle-Trivino, J., Rivera-Madrid, R., Leon-Pech, M. G., Cortes-Useche, C., Sellares-Blasco, R. I., Aguilar-Espinosa, M., and Arias-Gonzalez, J.E.	Assessing and genotyping threatened staghorn coral Acropora cervicornis nurseries during restoration in southeast Dominican Republic	2020
Dizon, R. M. and Edwards, A. J.	Comparison of three types of adhesives in attaching coral transplants to clam shell substrates	2009
Forrester, G.E., O'Connell-Rodwell, C. and Baily, P.	Evaluating methods for transplanting endangered elkhorn corals in the Virgin Islands	2010

García Rueda, A. L.	Cría de fragmentos de Acropora palmata y Montastraea cavernosa en una guardería a media agua en la Bahía de Gayraca (Parque Nacional Natural Tayrona) como …	2012
Goergen, E. A. and Gilliam, G. S.	Outplanting technique, host genotype, and site affect the initial success of outplanted Acropora cervicornis	2016
Goergen, E. A., Ostroff, Z. and Gilliam, G. S.	Genotype and attachment technique influence the growth and survival of line nursery corals	2017
Kumar, J. S. Y., Satyanarayana, C., Venkataraman, K., Beleem, I. B., Arun, G., Chandran, R., Ramkumaran, K., and Kamboj, R. D.	Coral restoration in Singapore's sediment-challenged sea	2017
Ladd, M. C., Shantz, A.A., Nedimyer, K., and Burkepile, D. E.	Density Dependence Drives Habitat Production and Survivorship of Acropora cervicornis Used for Restoration on a Caribbean Coral Reef	2016
Liñán-Cabello, M. A. and Flores-Ramírez, L. A.	Acclimation in Pocillopora spp. during a coral restoration program in Carrizales Bay, Colima, Mexico	2011
Lirman, D., Thyberg, T., Herlan, J., Hill, C., and Young-Lahiff, C.	Propagation of the threatened staghorn coral Acropora cervicornis: methods to minimize the impacts of fragment collection and maximize production	2010
Lohr, K. E., Ripple, K., and Patterson, J. T.	Differential disturbance effects and phenotypic plasticity among outplanted corals at patch and fore reef sites	2020
Lohr, K.E., Mcnab, A. A. C., Manfrino, C., and Patterson, J. T.	Assessment of wild and restored staghorn coral Acropora cervicornis across three reef zones in the Cayman Islands	2017

Lustic, C., Maxwell, K., Bartels, E., Reckenbeil, B., Utset, E., Schopmeyer, S., Zink, I., and Lirman, D.	The impacts of competitive interactions on coral colonies after transplantation: A multispecies experiment from the Florida Keys, US	2020
Mbije, N. E. J., Spanier, E., and Rinkevich, B.	Testing the first phase of the 'gardening concept'as an applicable tool in restoring denuded reefs in Tanzania	2010
Meesters, H. W. G., Boomstra, B., and Hurtado-Lopez, N	Coral restoration Bonaire: an evaluation of growth, regeneration and survival	2015
Mercado-Molina, A. E., Ruiz-Diaz, C. P., and Sabat, A. M.	Demographics and dynamics of two restored populations of the threatened reef-building coral Acropora cervicornis	2015
Nithyanandan, M., Le Vay, L., and Raja, D. K.	Coral nursery and transplantation of the staghorn coral, Acropora downingi in Sabah Al-Ahmad Sea City, Kuwait, Arabian Gulf	2018
Page, C.A., Muller, E. M. and Vaughan, D. E.	Microfragmenting for the successful restoration of slow growing massive corals	2018
Quinn, N. A. and Kojis, B. L.	Evaluating artificial means to increase Acropora coral populations and increase associated fish communities in Jamaica	2007
Quinn, N. J. and Kojis, B. L.	Evaluating the potential of natural reproduction and artificial techniques to increase Acropora cervicornis populations at Discovery Bay, Jamaica	2006
Ross, A.M.	Genet and reef position effects in out-planting of nursery-grown Acropora cervicornis (Scleractinia: Acroporidae) in Montego Bay, Jamaica	2014
Schopmeyer, S. A., Lirman, D., Bartels, E., and Gilliam, D. S.	Regional restoration benchmarks for Acropora cervicornis	2017
Shaish, L., Levy, C., Katzir, G., and Rinkevich, B.	Coral reef restoration (Bolinao, Philippines) in the face of frequent natural catastrophes	2010

Tortolero-Langarica, J. J. A. and Rodríguez-Troncoso, A. P.	Accelerated recovery of calcium carbonate production in coral reefs using low-tech ecological restoration	2020
Ware, M., Garfield, E. N., Nedimyer, K., Levy, J., Kaufman, L., Precht, W., Winters, R. S., and Miller, S. L.	Survivorship and growth in staghorn coral (Acropora cervicornis) outplanting projects in the Florida Keys National Marine Sanctuary	2020
Williams, D. E., and Miller, M. W.	Stabilization of Fragments to Enhance Asexual Recruitment in Acropora Palmata, a Threatened Caribbean Coral	2010

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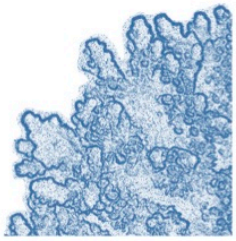
Zarza, E., Vargas, A., Londoño, L., and Pacheco, A.	Ensayo preliminar de crecimiento de fragmentos del coral amenazado Acropora cervicornis en una guardería colgante y experiencia piloto de trasplante en el Parque …	2014

Appendix 4: Standardised Coral Restoration Form

The Coral Collect

- A project seeking to gather and share knowledge in our quest to support the coral reefs.



Name of project			
Project area		Number of sites	
Name of site(s)			
Date of project start		Date of project end	
Will site monitoring occur during the project?	Y / N		
Frequency of monitoring			
Location of coral harvest			
Control group installed at harvest site?	Y / N		
Number of control colonies			
Source of colonies (i.e. nursery, wild, corals of opportunity)			
Number of colonies harvested			
Method of transport			
Duration of transport		Exposed to air	Y / N
Number of transplant sites			
Number of fragments per site, start			
Number of fragments per site, end			
Coral are considered survived when			
Percentage survival			
Transplantation substrate (i.e. nursery, cement blocks, wild)			
Adhesive used for attachment			