Biological Conservation 279 (2023) 109930

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Active coral propagation outcomes on coral communities at high-value Great Barrier Reef tourism sites

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ARTICLE INFO

Keywords:
Coral restoration
Reef ecology
Coral cover
Benthic surveys
Site stewardship

ABSTRACT

Tourism-led stewardship of Great Barrier Reef (GBR) sites has implemented upscaling of coral propagation and out-planting practices. Incorporating ecological monitoring into restoration practices has long been acknowledged as a vital component of reef restoration projects – thereby demonstrating success in relation to improved ecosystem function within impacted reefs. As such, one goal for this tourism-led stewardship project was to identify whether and how activity increased coral cover and target species populations within high-value tourism sites. We therefore investigated coral cover and relative cover of commonly out-planted genera within seven tourism sites over six reefs on the northern GBR via benthic surveys, both prior to the commencement of out-planting activities in August 2019, and again in August 2021. Over this 24-month time frame, out-planting intensity varied between 3970 and 84 out-plants within marked areas (with no propagation activity conducted within corresponding control areas) across sites. Within all out-planting sites, > 65 % of out-plants were *Acropora* species. One reef showed a significant increase in hard coral cover – 10.3 % ($p = 0.016$) – at out-planting areas compared to control areas by August 2021. At this same reef, proportions of total coral cover attributed to *Acropora* species were higher within the out-planting site compared to the control site. Despite variability in coral community responses observed across reef site/operators, we have demonstrated how coral propagation via this tourism-led stewardship model has the potential to increase hard coral cover within some out-planting sites, particularly of commonly out-planted genera.

1. Introduction

Coral restoration is increasingly considered an integral aspect of local reef ecosystem management in response to dramatic reductions in coral cover, alongside efforts to mitigate climate change (Hein et al., 2021; Quigley et al., 2022). Numerous coral restoration methods have been trialled and/or deployed globally in attempts to aid recovery rates of reef systems (e.g., Suggett and van Oppen, 2022). Notable methods include creating artificial structures akin to natural reefs (Lee et al., 2018), stabilisation of unconsolidated reef substrates (Ceccarelli et al., 2020), and rearing coral larvae in land- and field-based aquaculture settings for subsequent settlement on to degraded reef sites (Pollock et al., 2017; Suzuki et al., 2020; Miller et al., 2022). However, the most common method to date has been in-water coral propagation, whereby naturally and/or artificially produced coral fragments are reared on man-made structures (termed "nurseries") and subsequently attached (out-planted) onto the reef substrate (Boström-Einarsson et al., 2020). In-water coral propagation methods using coral nurseries have been increasingly utilised in the Caribbean (e.g., Young et al., 2012; Ware et al., 2020), Indo-Pacific (e.g. Feliciano et al., 2018), Red Sea (e.g., Rinkevich, 2000; Epstein et al., 2001), Japan (Omori et al., 2016) and, most recently, the Great Barrier Reef (GBR) (Howlett et al., 2021; Cook et al., 2022; Howlett et al., 2022), with the ultimate objective of locally re-building coral populations and/or reef communities.

Reef stakeholders undertaking coral propagation practices have historically met fundamental challenges associated with cost-efficiency

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https://doi.org/10.1016/j.biocon.2023.109930
Received 19 September 2022; Received in revised form 19 January 2023; Accepted 22 January 2023
Available online 1 February 2023

and scalability (Bayraktarov et al., 2019; Boström-Einarsson et al., 2020; Stewart-Sinclair et al., 2021), partly due to time-cost constraints of current methods for out-planting coral material (Boström-Einarsson et al., 2020; Suggett et al., 2019, 2020). For example, attaching coral fragments to the reef substrate using chemical adhesives is often a labour-intensive, expensive, and relatively slow (5–10 corals out-planted per person-hour), thus reducing the rate (and hence increasing the cost) of out-planting efforts (Boström-Einarsson et al., 2020; Gomez et al., 2010; Suggett et al., 2020). Consequently, coral propagation activities have often been implemented over relatively small spatial scales (Boström-Einarsson et al., 2020). When further considered against the relatively short-term (<18 months; Boström-Einarsson et al., 2020) temporal nature of many coral propagation programs (e.g., due to 1-to-3-year funding cycles; Hein and Staub, 2021, and logistical constraints associated with revisiting restoration sites), such limited scale of application has resulted in few ecological studies on broader scale changes to ecosystem health, function, or recovery resulting from out-planting coral (Hein et al., 2021).

Historically, few coral restoration studies (33 %) have examined wider ecological metrics, such as changes to coral cover and diversity, alongside typical measures of fragment growth and survival (Hein et al., 2017). Increases in hard coral cover and structural complexity have been observed within "restored" sites compared to natural levels for several restoration projects with varying propagation methodologies (artificial reefs, steel frames and direct transplantation in Thailand; direct transplantation in Florida Keys, USA; direct transplantation in US Virgin Islands; steel frames in the Maldives; Hein et al., 2020a). Similarly, a rubble stabilisation project in Indonesia demonstrated an increase in live coral cover by 10 % over two years at reef sites previously destroyed by blast fishing (Williams et al., 2019). However, any associated ecological responses – beyond biomass gained – to restoration efforts, such as coral diversity, recruitment rates and fish communities, have been harder to ascertain across locations and methodologies (Hein et al., 2020a; Hein et al., 2020b). Despite continual improvements to monitoring methods for restoration projects, most of these studies report limited outcomes necessary for facilitating positive feedback mechanisms needed to ultimately reinforce reef resilience (Ladd et al., 2018; Ladd et al., 2019). Fundamentally, there is still a mismatch between measured metrics and restoration goals for many restoration projects (Boström-Einarsson et al., 2020; Hein et al., 2020a; Hein et al., 2021).

Metrics commonly used to assess potential changes in the ecological health of a reef site include percentage hard coral cover, coral diversity, and biomass, and fish communities and abundance (Ladd et al., 2019; Williams et al., 2019; Hein et al., 2020a). Recent monitoring guidelines recommend the integration of ecological indicators into regular maintenance practices for the establishment of future coral restoration programs (Hein et al., 2017; Boström-Einarsson et al., 2020; Goergen et al., 2020). These programs should be able to demonstrate an increase in coral abundance and cover – in addition to the more generally well evidenced growth and survival of out-planted coral fragments – to be deemed successful (Goergen et al., 2020, Hien et al., 2021). If coral population enhancement and increasing reef resilience is the goal of coral propagation programs, measuring coral cover and coral community changes (e.g., diversity) over time is considered a priority for further improving best practice (Goergen et al., 2020; Hein et al., 2020a).

To aid in the scalability of asexually produced coral via propagation and out-planting, a recently developed physical attachment device – Coralclip®, a stainless-steel spring clip with a nail integrated via the spring coil – was developed, which allows for the rapid deployment of coral fragments onto consolidated reef substrate (Suggett et al., 2020). Coralclip® has been deployed on the central GBR as part of the "Coral Nurture Program" (CNP) – a partnership between researchers and tourism operators – with the aim of using coral restoration as a site stewardship practice at "high-value" tourism sites (i.e., sites with regular tourism visitation and high economic value; as per Spalding et al., 2017) (Howlett et al., 2022; https://www.coralnurtureprogram.org/). Within its first three years of operation (August 2018–August 2021), CNP tourism operators out-planted >46,000 coral fragments across six high-value reefs, and efforts are ongoing (Howlett et al., 2022). Whilst out-planting by the CNP is generally haphazard and diffuse across "planting areas" within tourism sites (Howlett et al., 2022), activity at this scale presents a unique opportunity for the assessment of local ecological changes, such as increases in coral cover. Here, we examine changes to hard coral cover – as well as the relative cover of commonly planted coral genera – across seven CNP sites over six reef systems, which collectively have been subject to different out-planting intensities over two years (Aug 2019–Aug 2021). We discuss how the out-planting methodology and intensity influences coral populations within planting sites relative to natural recovery rates of within-site controls, and how site conditions can influence out-planting intensity. In doing so, we demonstrate how analysing metrics beyond coral fragment success can further determine whether the broader restoration goals of coral population enhancement can be met.

2. Materials and methods

2.1. Study sites and data collection

In conjunction with the CNP, out-planting was conducted at six reefs offshore from Cape Tribulation to Cairns, North Queensland, between August 2019 and August 2021: Mackay Reef, Opal Reef, Low Isles Reef, Hastings Reef, Upolu Reef and Moore Reef (Fig. 1). These "high value" reefs and subsequent sites within these reef systems were selected due to their accessibility by daily tourism operations – influenced by overall site protection (e.g., from swell and tidal currents), initial coral cover, and suitability for diving and snorkelling activities (see Howlett et al., 2022) – thereby enabling consistent resurveying. From these criteria, we focussed on seven sites across these six reefs for our current study: Mackay Reef – "Angels"; Low Isles Reef – "Low Isles Site"; Opal Reef – "Bashful Bommie"; Hastings Reef – "1770" and "Stepping Stones"; Upolu Reef – "Wonderwall"; Moore Reef – "Moore Reef Site". Sites were differentially impacted by mass bleaching events in 2016/17 (AIMS 2022) and therefore exhibited variable starting conditions (in terms of percentage coral cover and community structure). Benthic cover was determined twice throughout this study by researchers via triplicate line-intercept transects, both prior to commencement of out-planting activities in August 2019 and again in August 2021, to determine overall changes in coral cover, coral community composition and benthic composition (i.e., hard coral, soft coral and abiotic cover) over time.

Triplicate 30 m video (GoPro Hero 5) line-intercept transects were conducted within marked locations at each of the 7 out-planting sites, and within at least one representative control site, where no out-planting or nursery activity would be conducted, per reef. Out-planting and control sites were each approximately 100 m in length and were marked using metal stakes and subsurface buoys for repeat surveys conducted in August 2021. Over time, out-planting was ultimately not restricted to these re-survey plot areas for any given out-planting site but was always prohibited in the controls (verified by periodic visual audit surveys for any installations of Coralclip®). All sites were within 2–5 m depth. Video transects were later analysed, whereby all scleractinian and soft corals were identified to genus level, whilst other substrates were categorised as either macroalgae, sponges, abiotic (e.g., sand, rubble, rock) or "other invertebrates" (as per Howlett et al., 2022).

2.2. Out-planting activities

Out-planting was conducted using Coralclip® as previously described (Suggett et al., 2020; Howlett et al., 2022). Fragments were sourced from either in situ fragmentation of existing coral colonies

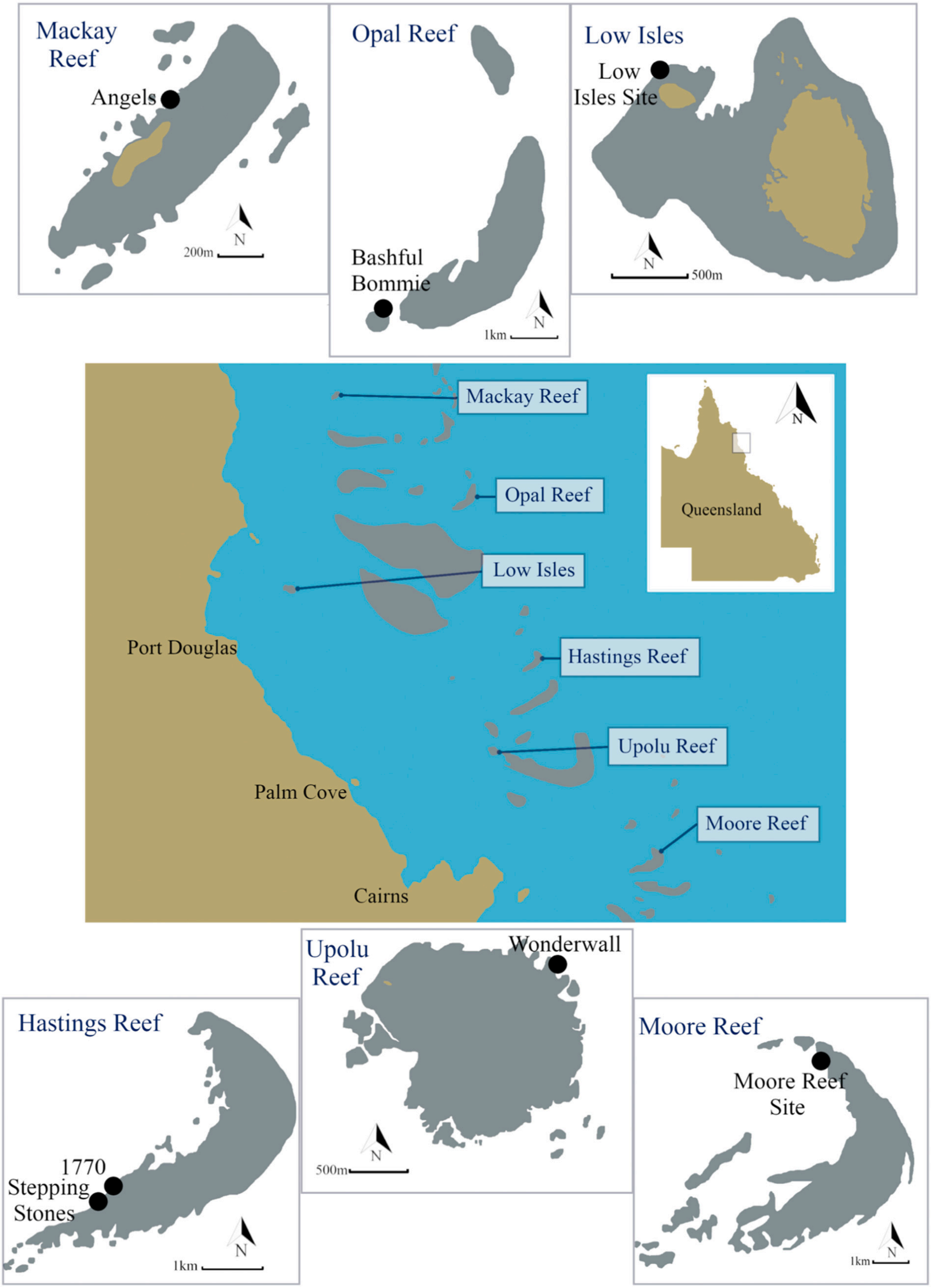

Fig. 1. Locations of all seven survey sites at six reefs within the central Great Barrier Reef, Cairns-Cape Tribulation region considered in this study (in part, adapted from Howlett et al., 2022). Survey sites coincide with "high value" reefs at which tourism operators engaged in coral propagation activities under the Coral Nurture Program.

(according to permit regulations), coral nurseries or "corals of opportunity" (naturally created fragments not attached to consolidated substrate). Coral material was only moved within reef systems and never between reefs. Once coral material was collected for planting, divers would hammer each Coralclip® device onto an area of clear, consolidated substrate, remove algae and debris from the immediate area using a wire brush, and then position the coral fragment firmly beneath the clip (as per Suggett et al., 2020; Howlett et al., 2022). Fragment source, out-plant location, genus and counts of all out-planted coral fragments were reported every 21 days by each operator. Out-planted fragment concentration varied between sites (e.g., due to availability of bare substrate required for Coralclip® deployment) and operations. All out-planting activities were undertaken periodically within the 2-year period according to site accessibility and availability of personnel. Previous fate-tracking experiments conducted at Hastings and Opal Reef demonstrated a survival rate ranging from 70 to 93 % for out-planted fragments (Howlett et al., 2022).

Out-planting effort – characterised as number of outplants between August 2019 and August 2021 – across 4 of the 7 sites chosen for our current study was similar, ranging from 3970 at "Angels" on Mackay Reef to 2792 at "Wonderwall" on Upolu Reef. Number of out-plants for Moore Reef and Low Isles sites were markedly lower (84 and 105, respectively; Table 1). This relates, in part, to the differing out-planting opportunities amongst tourism operations, number of available personnel, and site access throughout this time frame. Despite differences in out-plant numbers across sites, the most common genus of fragments installed with Coralclip® was always *Acropora*. Over the 7 sites where ecological surveys were conducted, the total number of out-plants installed was 14,255.

2.3. Data analysis

All statistical analyses were conducted using the statistical program R (v4.1.2) (RStudio Team, 2015). Using benthic survey data collected in August 2019 (prior to the commencement of out-planting activities), multivariate analyses were used to determine initial differences in hard coral, soft coral and abiotic cover between control and out-planting sites. Out-planting intensity (no. out-plants per m^2) was calculated using reported numbers of out-plants by operators (Table 1) and total surface area (SA) of out-planting plots, including the area outside of plots marked for surveys. SA measurements were conducted within Google Earth using satellite images of operator sites (Google Earth 9.0, 2022). Following square root transformation, a regression analysis was conducted to examine the extent of linear regression (if any) between out-planting intensity and initial hard coral cover (%) for each site. Assumptions of data linearity, homogeneity of residuals variance, normality of residuals and independence of residual error terms were tested via residual vs fitted, scale-location, Q-Q and residuals vs leverage plots, respectively (James et al., 2013).

Additional analyses were conducted on benthic survey data collected in August 2021. Welch's two sample *t*-tests were conducted to examine for differences in proportion of the three most out-planted genera for each reef between out-planting and control sites. Prior to analysis, total coral cover was separated according to genus, given as proportions between 0 and 1, and arcsine transformed. Welch's two sample *t*-tests were also conducted to examine for differences in hard coral cover, given as proportions between 0 and 1, between out-planting and control sites. Data was subsequently found to meet the assumption of normal distribution via Q-Q plots (James et al., 2013). To test for differences in benthic cover between control and treatment sites following two years of CNP activity, multivariate analyses were undertaken on measurements, in metres, of hard coral, soft coral, and abiotic cover for each reef taken in August 2021, and grouped according to treatment. Data was found to meet the assumptions of normal distribution and homogeneity of variance via Q-Q plots and Levene's tests, respectively (James et al., 2013). The extent of linear regression (if any) between average proportions of total coral cover for *Acropora* (%) from each site and overall change in proportions for *Acropora* (Δ *Acropora*; %) was determined via regression analysis, where Δ *Acropora* was the difference in proportions of total coral cover between surveys conducted in August 2021 and 2019. Assumptions of data linearity, homogeneity of residuals variance, normality of residuals and independence of residual error terms were tested via residual vs fitted, scale-location, Q-Q and residuals vs leverage plots, respectively (James et al., 2013).

Table 1
Summary of number of out-planted coral fragments installed according to tourism operator sites spanning 6 reef systems within the central Great Barrier Reef over 24 months, Aug 2019-Aug 2021. Percentage of coral according to most common coral genera was calculated as a proportion of total out-plant numbers per site.

Reef	Site	No. out-plants	Most common genera	% total no. out-plants
Hastings	1770	3257	*Acropora*	66 %
			Pocillopora	27 %
			Echinopora	3 %
	Stepping Stones	677	*Acropora*	67 %
			Pocillopora	32 %
			Echinopora	1 %
Opal	Bashful Bommie	3370	*Acropora*	87 %
			Pocillopora	10 %
			Echinopora	1 %
Mackay	Angels	3970	*Acropora*	83 %
			Seriatopora	8 %
			Echinopora	2 %
Upolu	Wonderwall	2792	*Acropora*	88 %
			Pocillopora	4 %
			Echinopora	3 %
Low Isles	Low Isles Site	105	*Acropora*	76 %
			Pocillopora	24 %
Moore	Moore Reef Site	84	*Acropora*	96 %
			Pocillopora	4 %

3. Results

3.1. Baseline benthic surveys

Benthic surveys conducted prior to out-planting in August 2019 identified that the percentage hard coral cover and total coral cover (as soft and hard coral combined) was lowest for Upolu Reef (3.1 % ± 0.8 and 19.0 ± 4.5, respectively; mean ± SE) and highest for Opal Reef (34.4 ± 3.2 and 40.3 ± 3.3, respectively) (Fig. 2). Of the total percent coral cover, *Porites* was the dominant coral genus at both Hastings Reef and Opal Reef, (36.3 % ± 5.6 and 34.7 % ± 7.1, respectively; mean ± SE) whereas *Acropora* was most dominant coral genus at Moore Reef and Low Isles (45.5 % ± 11.6 and 27.6 % ± 10.1, respectively). Soft coral was the most dominant coral taxa at Mackay and Upolu Reefs (46.3 % ± 3.6 and 75.2 % ± 5.0, respectively) (Fig. 3). Low Isles had the highest diversity of hard coral genera and Moore Reef had the lowest (10.7 ± 1.1 and 3.9 ± 0.8 genera per transect, respectively) (as per Howlett et al., 2022). Additionally, benthic cover between out-planting and control sites was compared within reefs to determine the suitability of control sites. Overall, there were no discernible differences in hard coral, soft coral, and abiotic absolute cover between out-planting and control sites for any reef prior to the commencement of propagation activities (August 2019; $p > 0.05$) (Multivariate ANOVA, Table A.1).

3.2. Changes in hard coral cover from out-planting

Given the variability in "baseline" (August 2019) coral cover between reefs, we first determined the overall change in absolute hard coral cover over time from 2019 to 2021, hereafter referred to as Δ hard coral (cm). Here, Δ hard coral between control and out-planting (treatment) sites was variable between reefs. Only Moore Reef

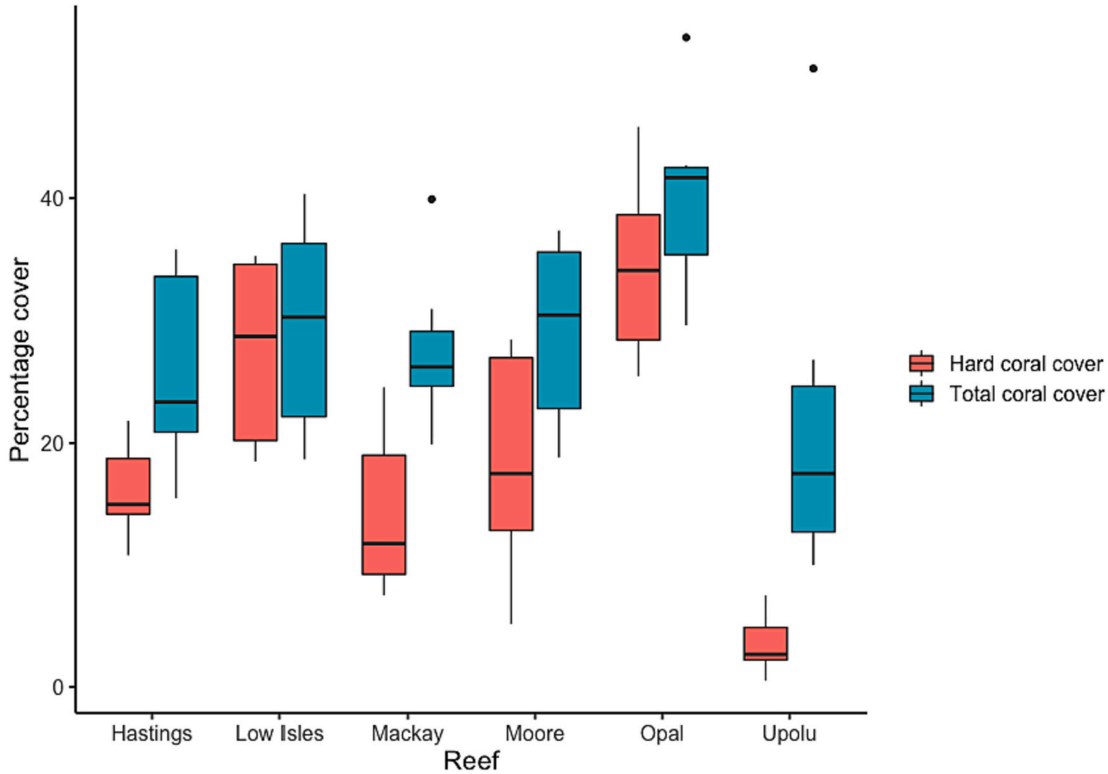

Fig. 2. Boxplot comparison, showing the median and interquartile range, of initial total coral cover (hard and soft coral) and hard coral cover for all six surveyed reefs. Whiskers show the minimum and maximum values and dots represent outliers. Benthic data was obtained via triplicate 30 m line-intercept video surveys (GoPro Hero 5) conducted within each site prior to out-planting activities in August 2019 (see also Howlett et al., 2022).

exhibited a significantly higher mean Δ hard coral at treatment sites compared to control sites ($p = 0.048$), although Δ hard coral within treatment sites was also higher at Opal Reef ($p = 0.552$). Hastings and Mackay Reefs showed little to no differences between treatments, and Upolu and Low Isles Reefs showed lower Δ hard coral at treatment sites compared to control sites (Fig. 4). Moore Reef had the largest difference in Δ hard coral between control and treatments. For Upolu and Moore Reefs, the range of Δ hard coral was greater within the treatment sites as opposed to the control sites, supporting observations of non-uniform out-planting within treatment plots (Fig. 4). Additionally, Δ hard coral within control sites was variable between reefs, ranging from a loss of approximately 6 % coral cover for Moore Reef to an increase of 25 % for Low Isles over the two years, most likely reflecting differences in natural recovery rates between reefs.

Given the subtle responses observed in the change in Δ hard coral for some sites (Fig. 4), we next compared the final absolute hard coral cover for control and out-planting plots for 2021 as a measure of whether out-planting exceeded natural recovery. Here, Upolu Reef showed the largest difference between hard coral cover at control versus out-planting sites ($p = 0.016$; Fig. 5). Following two years of out-planting activity, the mean hard coral cover at Upolu Reef was almost 2.5 times higher in the out-planting site compared to the control site. Two other sites, Moore Reef and Opal Reef, also had higher coral cover in the treatment sites, but these were not statistically significant ($p = 0.479$ and 0.269, respectively; Fig. 5). Hastings, Mackay, and Low Isles reefs showed a decrease in mean hard coral cover at control sites compared to out-planting sites, yet for Hastings and Mackay reefs these changes were subtle and likely a result of natural variability in coral cover ($p = 0.884$, 0.643 and 0.440, respectively; Fig. 5).

Given Upolu Reef exhibited the largest difference in hard coral cover between out-planting and treatment sites in 2021 but also the lowest initial total coral cover and percent coral cover (Figs. 2, 3), we considered that this could be a result of planting effort being influenced by initial (background) reef status. In our case, the method of attaching coral fragments – Coralclip®– requires availability of bare consolidated reef substrate within the site. Percentage of consolidated substrate within benthic surveys prior to the commencement of out-planting activity was highest at Mackay Reef (72.2 %), similar for Low Isles, Hastings and Opal Reefs (45.7, 45.6 and 42.8 %, respectively) and lowest for Upolu and Moore Reefs (28.1 and 16.6 %, respectively). Out-planting intensity across sites ranged from 1.9 to 0.3 outplants per m^2 (sites Bashful Bommie - Low Isles, Fig. 6); however, no discernible linear regression was evident between initial coral cover and out-planting intensity $R^2 = 0.01$; $p > 0.05$ (Fig. 6). At face value this suggests that differences in out-planting intensity between sites most likely reflected differences in operator capacity and site access (see also Howlett et al., 2022). However, removing the site Bashful Bommie from this analysis, which had both high initial hard coral cover and out-planting intensity, resulted in a negative correlation and a clearer linear regression between out-planting intensity and initial hard coral cover for the remaining sites ($R^2 = 0.37$; $p = 0.12$; Fig. A.1). After further investigation via the residual vs leverage plot alongside the regression analysis, we were able to identify Bashful Bommie as an influential data point within the original regression model.

3.3. *Coral community composition*

In addition to analysing changes in hard coral cover for control and treatment sites (and over time), we examined the core coral genera contributing to cover. The total coral cover for each transect was separated according to genera, and differences in proportions of commonly out-planted coral genera were compared between control and treatment sites. Upolu and Mackay Reefs exhibited a higher proportion of total coral cover attributed to *Acropora* within out-planting sites compared to control sites, with the difference at Upolu Reef being statistically significant (August 2021; $p = 0.02$ and 0.08, respectively; Table A.3). At

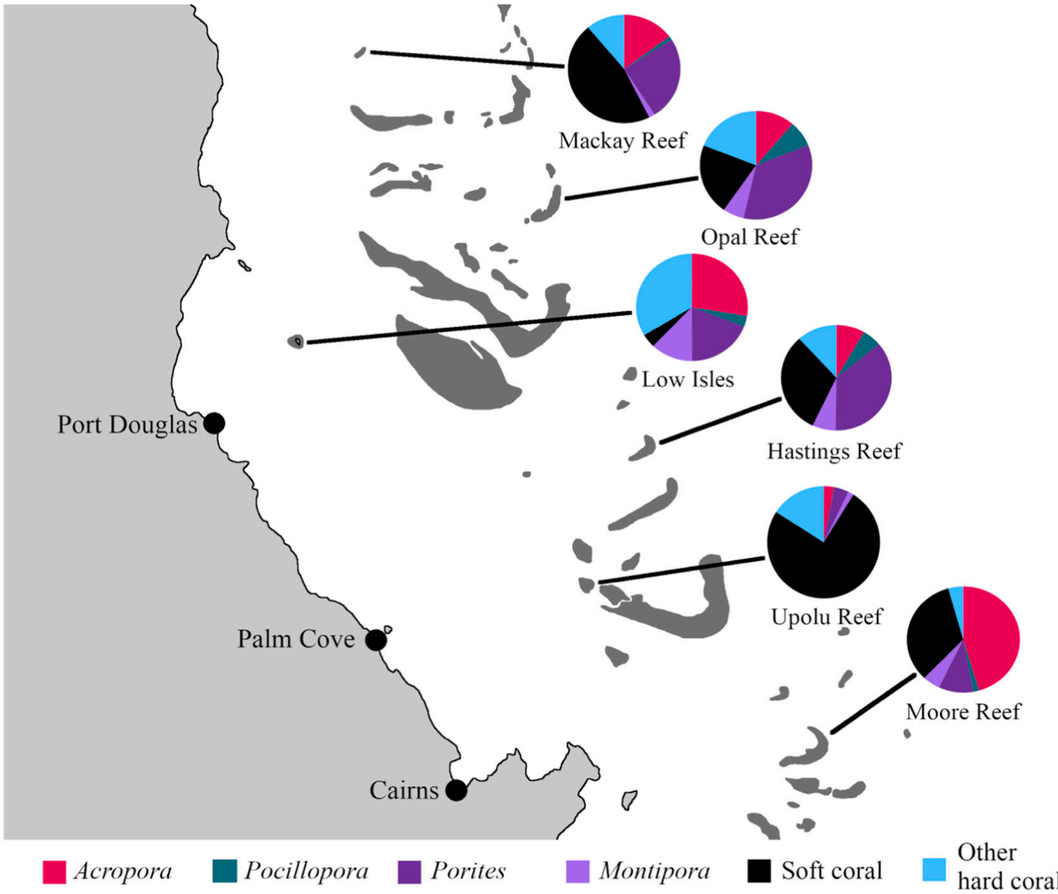

Fig. 3. Proportions of total coral cover according to common coral genera for each survey site/reef on the central Great Barrier Reef between Cape Tribulation and Cairns (shown as the mean of triplicate surveys; see Table A.2). Coral cover determined via triplicate 30 m line-intercept video surveys (GoPro Hero 5) conducted within each site prior to out-planting activities in August 2019 (see also Howlett et al., 2022).

Upolu Reef, there were no *Acropora* colonies identified within the control plots, whereas, within out-planting plots, 4.6 % of total coral cover was identified as *Acropora*. At Mackay Reef, the proportion of total coral cover attributed to *Acropora* was 19.2 % in control plots and 33.5 % within out-planting plots. No other changes in coral genera were found between control and out-planting plots at Mackay and Upolu Reefs, nor for any commonly out-planted genera within the other four reefs.

Given Upolu Reef had the lowest initial cover of *Acropora* (Fig. 3), we again evaluated whether this reflected differences in extent of *Acropora* between control and treatment after largely out-planting fragments of this genus (Fig. A.2). However, following regression analysis, no linear regression was found between initial proportions of coral cover attributed to *Acropora* and overall changes in proportions of *Acropora* observed after two years of out-planting activity ($R^2 = 0.07$; $p > 0.05$; Fig. A.2). Therefore, notable increases in proportions of *Acropora* at out-planting sites compared to control sites on Upolu Reef are likely a reflection of the increase in hard coral cover demonstrated previously, given *Acropora* species are relatively fast growing (Howlett et al., 2022).

Finally, a multivariate Analysis of Variance analysis revealed a notable difference in hard coral, soft coral and abiotic absolute cover between out-planting and treatment sites at Upolu Reef following two years of out-planting activity (August 2021; $p = 0.04$; Table A.4). In contrast, all other reef systems did not exhibit differences in hard coral, soft coral and abiotic absolute cover between out-planting and treatment sites (August 2021; $p > 0.05$; Table A.4).

4. Discussion

Monitoring of coral propagation activities to date have been mostly limited to short-term (<18 months) measures of coral fragment survival and growth (e.g., Boström-Einarsson et al., 2020), with very few studies exploring project success via long-term (>18 months) ecological impacts. Here, we examined the outcome of two years' worth of out-planting activity through reef stewardship by tourism operators on coral cover and community composition at high-value tourism reef sites. Out-planting effects on hard coral cover were variable between reefs, where average hard coral cover significantly increased within the out-planting sites compared to control sites at one reef (Upolu Reef; $p < 0.05$), with a further two reefs showing a non-significant increase within out-planting sites. Significant changes in the proportions of commonly out-planted coral genera were also observed at out-planting sites on Upolu Reef, and a notable change in that on Mackay Reef. Such variations in coral cover and target species responses likely reflect the natural variability in fragment survival, environmental parameters, and out-planting intensity (Bowden-Kerby, 2001; Forrester et al., 2014; Howlett et al., 2022), and how these factors manifest through further differences in baseline environmental factors. Interestingly, out-planting intensity negatively correlated with initial hard coral cover for most sites. Such an outcome appears logical operationally since Coralclip® is designed for use on consolidated reef substrate (areas of low live coral cover) (Suggett et al., 2020) – hence, more space available likely results in faster and easier out-planting (less time required to find planting space). That said, such a finding was not ubiquitous nor supported by the percentage cover of consolidated substrate within sites, highlighting important differences in operations and how these may impact how ecological gains are observed. For example, Opal Reef and Low Isles Reef exhibited similar levels of hard coral cover and consolidated substrate prior to the commencement of out-planting activities, yet these sites had

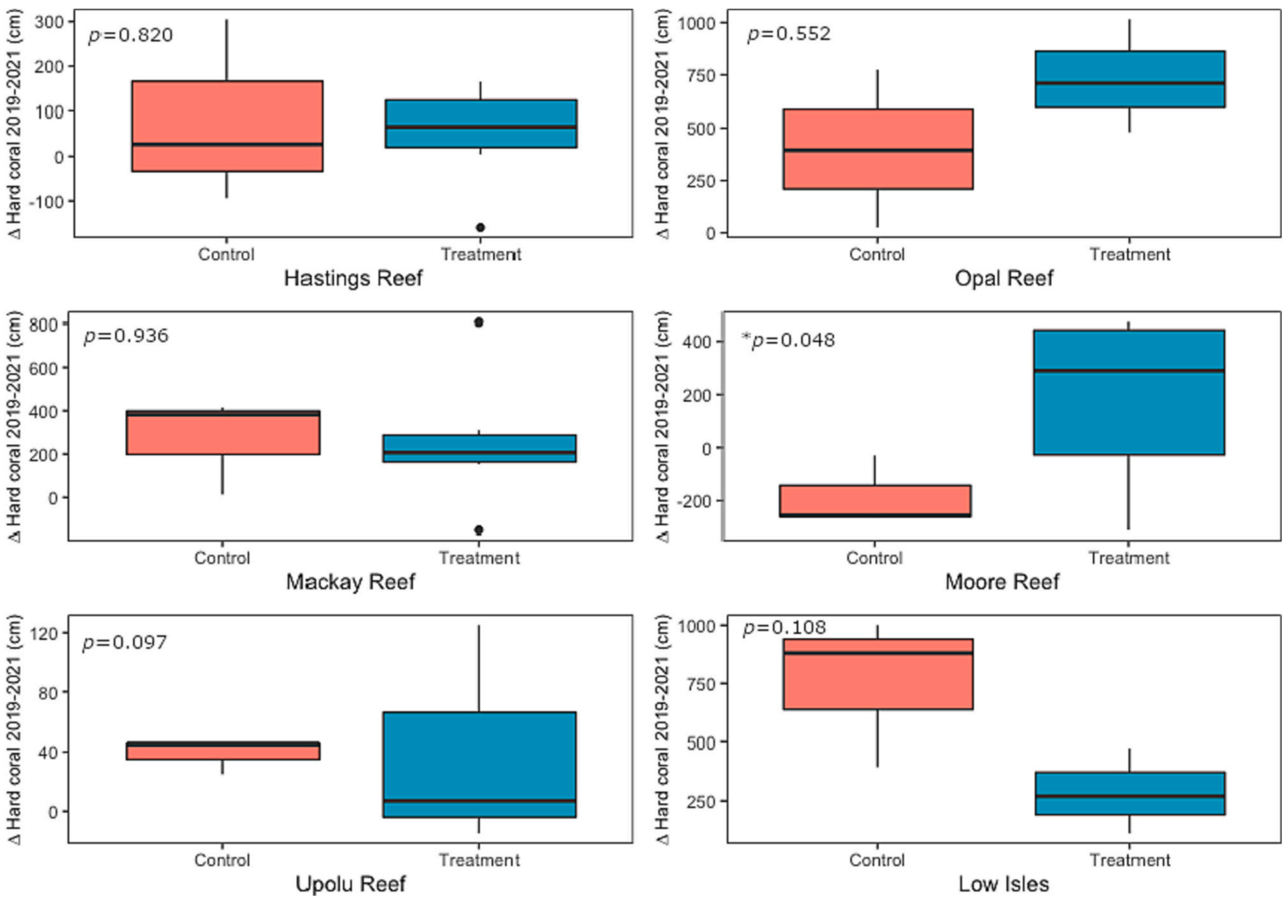

Fig. 4. Boxplots displaying the median and interquartile range of absolute changes in hard coral cover from August 2019 to August 2021 (Δ hard coral, cm) for control and out-planting (treatment) sites, according to reef. Whiskers show the minimum and maximum values and dots represent outliers. Hard coral cover was determined via triplicate 30 m line-intercept video surveys (GoPro Hero 5) conducted within each site and given as absolute measurements (cm). p-values displayed are based on Welch's two sample t-tests comparing means of control and treatment sites ($^*p < 0.05$).

vastly different levels of out-planting intensity. In the case of Bashful Bommie, it is clear operators can overcome the lack of available bare substrate required for out-planting by planting fragments haphazardly and more densely amongst existing coral cover. Additionally, the comparatively high out-planting intensity achieved at Opal Reef alongside high initial coral cover suggests, in this case, that Coralclip® may have been utilised as a tool for site maintenance (retaining existing cover), rather than facilitating site recovery (boosting new cover). Here, sites such as Bashful Bommie on Opal Reef could likely be less impacted by out-planting activities given the higher rates of natural recovery. However, this would need to be confirmed by additional studies on natural rates of recruitment.

Sites with low coral cover exhibit inherently lower capacity to naturally recover (Hughes et al., 2019) since coral recruitment is directly correlated with the density of existing coral colonies at reef sites (Bramanti and Edmunds, 2016; Davidson et al., 2019). Thus, the greatest out-planting-induced change in hard coral cover (and the proportions of *Acropora*) observed at Upolu Reef, which had the lowest initial hard coral cover, was perhaps to be expected where natural recovery rates are likely to be low when compared to that of other surveyed reefs (Bramanti and Edmunds, 2016; Davidson et al., 2019). Again, this would need to be confirmed via metrics capturing differences in natural recruitment rates between reefs and treatments (out-planting vs control). Additionally, whilst overall differences in natural recovery rates were captured via control sites – 6 % loss for Moore Reef to 25 % increase for Low Isles – our two year study did not fully capture yearly changes in recruitment and growth rates. Undoubtedly, monitoring benthic changes for a longer time frame (e.g., ≥ 5 years, as per Goergen et al., 2020) will help resolve these issues as coral recruitment and growth continues.

Long-term (>18 months) ecological studies necessary for monitoring restoration success require financial support that is not often covered by typical academic funding (i.e., due to 1-to-3-year funding cycles; Hein and Staub, 2021). Costs associated with revisiting reefs present additional constraints to coral restoration projects that are not typically shared within terrestrial systems (Omori, 2019). Therefore, novel restoration financing and financial stability for coral reefs is needed (Quigley et al., 2022). Potentially, this can be achieved via multiple funding sources, such as private investment and government support, particularly given the economic benefits of coral reef ecosystem services (e.g., shoreline protection and tourism) (Brathwaite et al., 2022; Quigley et al., 2022; Fezzi et al., 2023). Additionally, economic benefits of coral restoration should be further demonstrated through improvements to cost-effectiveness and scalability, in addition to long-term monitoring of ecological changes, such as improvements to coral cover and coral communities (Hein et al., 2020a; Quigley et al., 2022).

Whilst environmental dynamics associated with any given reef will inevitably influence changes in coral community structure over time, the lack of consistency in observable out-planting impacts may, in part, be due to the survey method. Use of the line-intercept method used for our study assumes the majority of out-planted fragments fall within the small 'strip' covered by the transects, which was not the case for the majority of sites as CNP activities were not restricted to the areas marked for re-surveying (Howlett et al., 2022) and out-planting was conducted haphazardly within sites. Since all surveys were undertaken at tourism sites, in-field time-constraints limit the size of representative surveyed

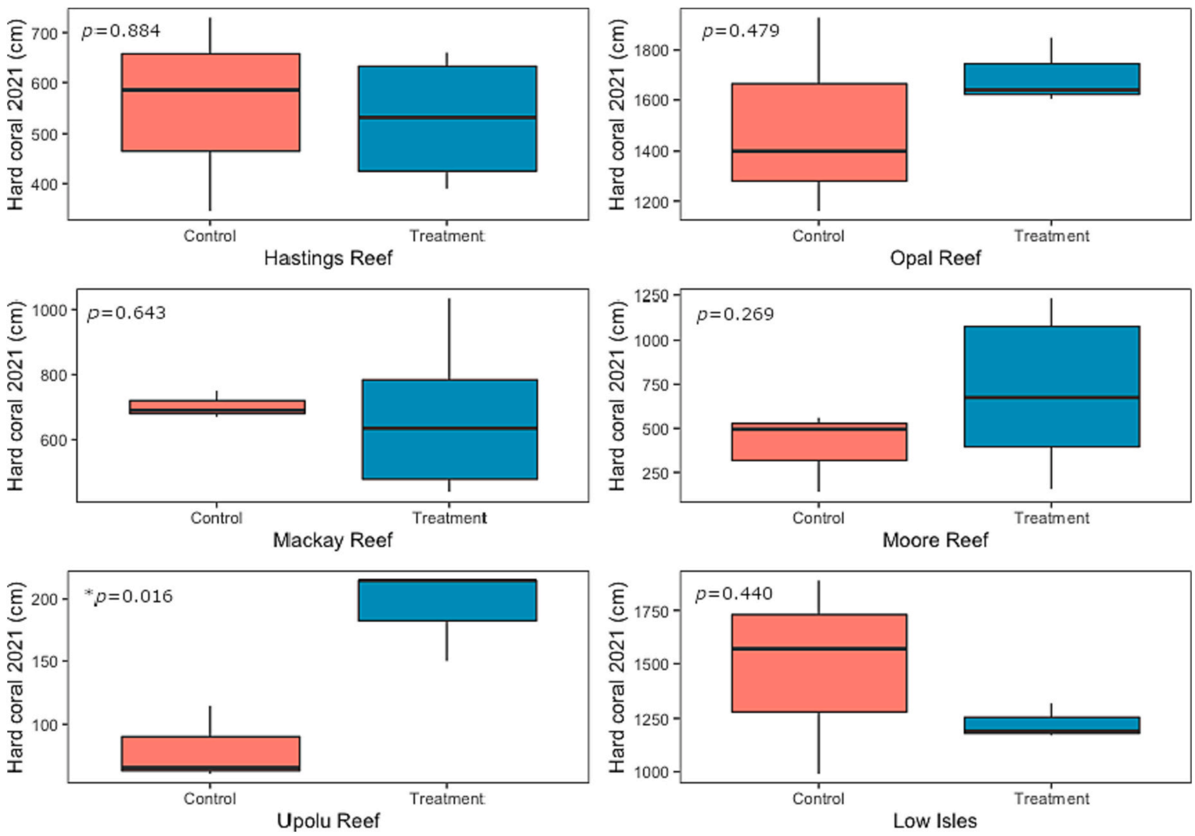

Fig. 5. Boxplots displaying differences in the median and interquartile ranges of hard coral cover in August 2021 between control and out-planting (treatment) sites, according to reef. Whiskers show the minimum and maximum values. Hard coral cover was determined via triplicate 30 m line-intercept video surveys (GoPro Hero 5) conducted within each site and given as absolute measurements. *p*-values displayed are based on Welch's two sample *t*-tests comparing means of control and treatment sites (*$p < 0.05$).

areas within the treatment site. Whilst embedding out-planting and associated surveying within routine tourism operations is important to maximise cost-effectiveness of practice (see Howlett et al., 2022), it introduces trade-offs for time available for monitoring versus detail. Restoration efforts utilising similar survey techniques (i.e., representative measures of hard coral cover) demonstrated more consistent increases in coral cover over longer time periods (i.e. >10 years; Hein et al., 2020a), or when restoration activity was contained to one area (Williams et al., 2019). The CNP approach involves haphazard planting across large areas, thereby introducing large heterogeneity to the "treatment" plots (compared to more homogenous plots used for mono-species restoration elsewhere, e.g., *Acropora tenuis* in Philippines; Harrison et al., 2021). It follows that longer monitoring may be required to capture how planting with a biased taxonomic group but haphazardly into an existing diverse coral community manifests. Additionally, whilst the line-intercept method is a recommended and well used ecological performance metric for restoration monitoring (Georgen et al., 2020), there is scope within the CNP to explore additional survey methods that can capture small-scale changes over a larger area (e.g., underwater photogrammetry; Nocerino et al., 2020; Rossi et al., 2020).

In addition to percentage hard coral cover, species and functional diversity are other key metrics associated with reef resilience and recovery (Quigley et al., 2022; Shaver et al., 2022). At all survey sites, the dominant out-planted genera of coral were fast-growing taxa with high structural complexity (*Acropora* and *Pocillopora*) due to a generally high availability of out-planting material within sites where they were commonly found as "corals of opportunity" (Howlett et al., 2022). On the GBR, *Acropora* assemblages are highly susceptible to stress events such as marine heat waves, cyclones, and Crown-of-Thorns starfish (COTS) outbreaks, and have experienced subsequent declines in cover, recruitment, and recovery in recent decades (Ortiz et al., 2018; Hughes et al., 2019; Ortiz et al., 2021). Given an increased presence of *Acropora* colonies at reef sites can drastically increase larval settlement and overall reef recovery (Bramanti and Edmunds, 2016; Ortiz et al., 2021), out-planting *Acropora* species could have a cumulative positive impact on out-planting sites with low initial levels of *Acropora* cover. This could be the case at Upolu Reef, where increases in coral cover within "treatment" plots were reflected by an increase in the most commonly planted coral taxa. However, this was not the case for most sites, where we did not observe notable changes in the proportions of commonly out-planted taxa between control and "treatment" plots. Such an outcome in retrospect is perhaps not unsurprising where our study only spans two years, given corals are planted initially as small fragments (typically 5–10 cm) and growth rates range from ~235 to 2736 mm^2 $month^{-1}$ (Howlett et al., 2022). However, it is important that out-planting effort allows for the retention of coral diversity and structural complexity within reef sites (Komyakova et al., 2013; Hein et al., 2020b; Seraphim et al., 2020). Though further work on the subject is warranted, it is possible that focussing on out-planting vulnerable and ecologically important coral species can be a method of improving best practice for coral propagation.

5. Conclusions

Increasing hard coral cover is an important aspect of ensuring reef resilience in the face of future stress events and remains a central aim of coral restoration practices (Hughes et al., 2010; Hein et al., 2020a). In the case of CNP, it is apparent that an increase in out-planting intensity can – but does not consistently – result in an increase in hard coral cover within out-planting sites over two years. For the first time on the GBR we

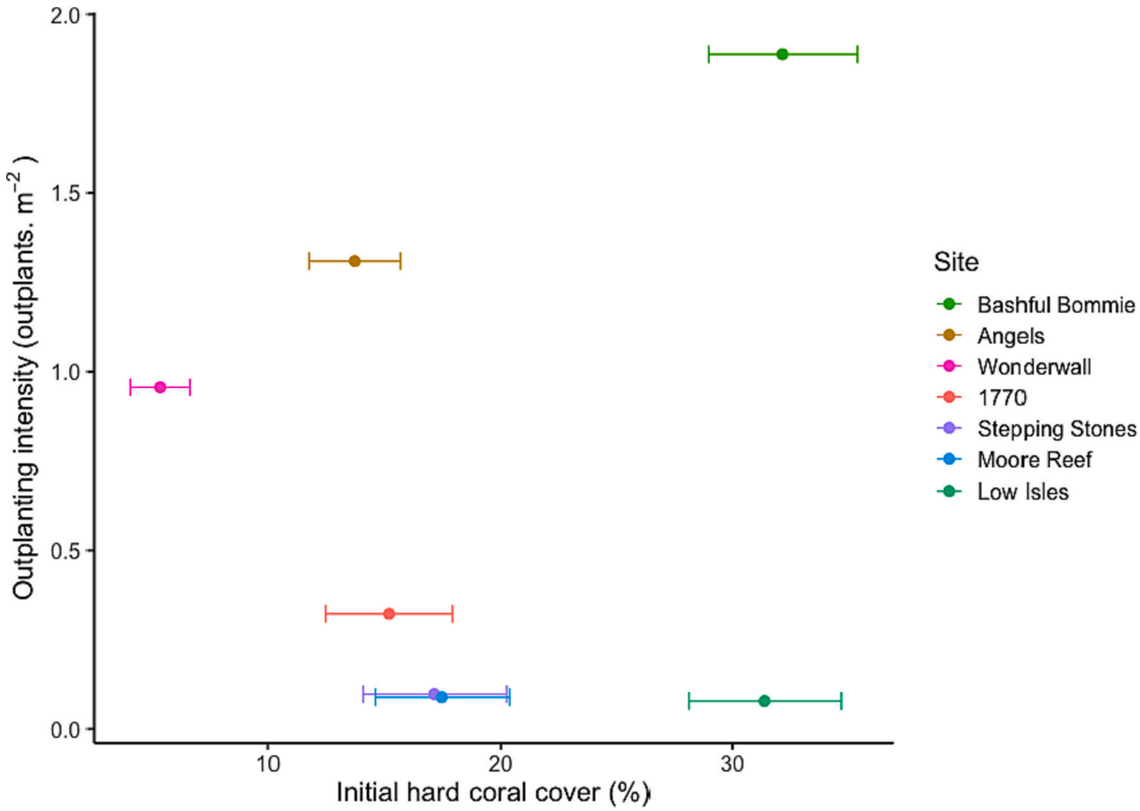

Fig. 6. Mean initial hard coral cover (±standard error) versus out-planting intensity (no. of out-plants per m^2) for each Coral Nurture Program site. Benthic data was obtained via triplicate 30 m line-intercept video surveys (GoPro Hero 5) conducted within each site prior to out-planting activities in August 2019 (see also Howlett et al., 2022).

have demonstrated how coral propagation practices at high-value tourism sites have resulted in different reef benthic communities compared to unamended control areas. In addition, out-planting effort by tourism operators does not necessarily result in an increase in coral cover, but may be dependent on the initial state of the benthic community. Thus, ecological improvements resulting from propagation activity could potentially be maximised by focussing out-planting effort at sites with initially low coral cover. Through consistent – but likely not optimum methodology of – ecological monitoring since the commencement of the CNP, we have shown the potential for coral propagation practices to aid reef recovery at local tourism sites where natural coral cover is relatively low. Therefore, it is important to measure metrics to determine restoration success beyond planted fragment survivorship and growth, such as coral cover, diversity and community composition. At reef sites where natural recovery is substantial enough to maintain coral cover, it is worth exploring the benefits of out-planting activities beyond ecological improvement, such as socio-economic support for tourism operators. This further highlights the need for continued, tailored and long-term monitoring of out-planting efforts, particularly as these efforts begin to upscale and potentially influence reef ecosystem services.

CRediT authorship contribution statement

Lorna Howlett: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing – original draft, Writing – review & editing, Visualization, Project administration. **Emma F. Camp:** Conceptualization, Methodology, Validation, Resources, Writing – original draft, Writing – review & editing, Supervision, Funding acquisition. **John Edmondson:** Conceptualization, Investigation, Writing – review & editing, Funding acquisition. **Russell Hosp:** Methodology, Investigation, Writing – review & editing. **Ben Taylor:** Methodology, Investigation, Writing – review & editing. **Philip Coulthard:** Methodology, Investigation, Writing – review & editing. **David J. Suggett:** Conceptualization, Methodology, Validation, Resources, Writing – original draft, Writing – review & editing, Supervision, Funding acquisition.

Declaration of competing interest

The authors declare no financial interests/personal relationships which may be considered potential competing interests.

Data availability

Data will be made available on request.

Acknowledgements

The authors wish to express immense thanks to the Great Barrier Reef Marine Park Authority (GBRMPA), whose support established the permits for out-planting and survey activity at Mackay Reef, Low Isles, Opal Reef, Hastings Reef, Upolu Reef and Moore Reef (G20/43740.1 and G18/40023.1), as well as owners and staff from Sailaway Port Douglas, Wavelength Reef Cruises, Passions of Paradise, Ocean Freedom and Great Adventures (notably Steve and Katrina Edmondson, Louisa Payne, Andrea Asúnsolo Rivera, Amy Jones, Alan Wallish, Scott Garden, Perry Jones, Taryn Agius and Dougie Baird), who have provided essential reef site access, operational assistance and assisted with data collection. EFC was supported by an ARC Discovery Early Career Research Award (DE190100142) and University of Technology Sydney Chancellor's Postdoctoral Research Fellowship.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2023.109930.

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