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

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Implementation of coral restoration practices within reef management strategies is accelerating globally to support reef resilience and recovery. However, full costs underpinning restoration project feasibility have historically been underreported yet are critical to informing restoration cost-benefit decision-making. Such knowledge is especially lacking for Australia's Great Barrier Reef (GBR), where a coral restoration program led by reef tourism operators, Coral Nurture Program (CNP), was initiated in 2018 (northern GBR) and continues to scale. Here we describe the early outcomes and costs of implementing similar tourism-led asexual coral propagation and outplanting practices in a new region, the Whitsundays (central GBR) through the CNP. Specifically, we detail the local operational and environmental context of CNP Whitsundays, describe the costs of implementation and continuation of restoration activities, as well as evaluate survivorship of coral outplants across three restoration sites for nine months after project establishment (August 2022 to June 2023). Baseline benthic surveys revealed relatively low hard coral cover at restoration sites (ranging from 3.22-8.67%), which significantly differed in benthic composition from coral collection sites (ranging 16.67-38.06%), supporting strong motivation by tourism operators to undertake restoration activities. Mean coral survivorship of coral outplants in fate-tracked plots differed between the three restoration sites after 267 days (ranging 23.33-47.58%), with declines largely driven by coral detachment. Early-stage cost-effectiveness (costs relative to outplant survival) associated with implementation of restoration activity varied widely from US$33.04-178.55 per surviving coral (n = 4,425 outplants) depending on whether 'in-kind' costs, restoration activity (outplanting only *vs.* total costs encompassing planning through to monitoring), site-based survivorship, or a combination of these factors, were considered. As coral reef restoration projects continue to be established globally, our results highlight the need for ongoing, long-term monitoring that can inform adaptive practice, and fully transparent cost-reporting to understand and improve feasibility for any given project. We further highlight the inherent context-dependency of restoration costs, and the importance of considering local social-environmental contexts and their associated cost-benefits in economic rationale for reef restoration projects.

KEYWORDS

reef restoration, coral propagation, coral outplanting, restoration costs, cost-effectiveness, Great Barrier Reef, reef tourism, reef stewardship

1 Introduction

Adoption of coral restoration approaches for targeted, local-scale site intervention is accelerating globally in efforts to support the resilience of coral reef ecosystems under persistent anthropogenic pressures (Boström-Einarsson et al., 2020; Shaver et al., 2022). In parallel with the urgent mitigation of global climate change and management of local stressors, coral restoration practices are now considered a central pillar of strategies to conserve the socio-ecological value of coral reefs (ICRI, 2021; Kleypas et al., 2021; Suggett et al., 2024), including Australia's Great Barrier Reef (GBR) (GBRMPA, 2017). Coral reef restoration has been implemented in over 50 countries in the last two decades (Boström-Einarsson et al., 2020), yet was not considered as a reef management strategy on the GBR until the World Heritage Area was severely impacted by consecutive mass coral bleaching and mortality events in 2016 and 2017 (Anthony et al., 2017; McLeod et al., 2022). Widespread mortality of corals across the GBR marine park (Hughes et al., 2017; Hughes et al., 2021) prompted trialing of active interventions from 2017, notably via small-scale funding into various community-led restoration activities (McLeod et al., 2022) and parallel large-scale funding into restoration research and development (Anthony et al., 2020). Restoration approaches implemented *in situ* since 2018 have spanned asexual coral propagation and outplanting (Cook, 2022; Howlett et al., 2022), coral relocation (Smith et al., 2024), substrate stabilization (Cook et al., 2022; McLeod et al., 2022; Nuñez Lendo et al., 2024), macroalgae removal (e.g., Smith et al., 2023), and coral larval-based restoration approaches (e.g., Randall et al., 2021, 2023). As with many reef regions globally, scaling of restoration activity on the GBR continues to be largely delivered through partnerships of diverse reef stakeholders (Howlett et al., 2022; McLeod et al., 2022).

Reef tourism operators are particularly playing a critical role in the implementation of restoration projects (Hein et al., 2020b; GBRMPA, 2020; Howlett et al., 2022; McLeod et al., 2022). When co-designed together by restoration practitioners, reef managers, traditional owners and stakeholders, and effectively resourced, such reef restoration-based site stewardship can support local site recovery (Hein et al., 2020a; Calle-Triviño et al., 2021; Howlett et al., 2023; Knoester et al., 2023; Lange et al., 2024; Nuñez Lendo et al., 2024), as well as provide positive feedback loops to reef stakeholders though socioeconomic and cultural benefits (Kittinger et al., 2016; Hein et al., 2019; Westoby et al., 2020; Suggett et al., 2023). Reef stewardship practices have been shown to promote shared responsibility amongst practitioners thereby solidifying sustained participation (e.g., Kittinger et al., 2016; Hein et al., 2019; Virdis et al., 2021), generating alternative livelihoods and revenue (Bayraktarov et al., 2020; Suggett et al., 2023), improving social license through community education and awareness of reef threats (Quigley et al., 2022; Palou Zúniga et al., 2023) and importantly, reducing costs through in-kind contributions of time, knowledge or resources (e.g., dela Cruz et al., 2014; Hein et al., 2018; Suggett et al., 2020, 2023; Scott et al., 2024). Partnerships and collaborations amongst reef stakeholders are thus integral to sustain assisted reef recovery initiatives at socioeconomically and ecologically relevant temporal and spatial scales (Bayraktarov et al., 2015; Westoby et al., 2020; Suggett et al., 2023).

On the northern GBR, reef tourism operators and researchers initiated asexual coral propagation and outplanting activity in the Cairns and Port Douglas region in 2018 via the "Coral Nurture Program" (CNP) (detailed in Howlett et al., 2022; Suggett et al., 2023; Scott et al., 2024). CNP was conceived with the dual aims to support local site recovery and enhance stewardship capacity of reef tourism operator staff through maintaining and improving hard coral cover at tourism reef sites (Howlett et al., 2022). Staged implementation of activity by six tourism operators over six years has resulted in ca. 100,000 coral fragments outplanted, and >120 coral nurseries established at 27 reef sites in the Cairns-Port Douglas region (as of Q1 2024). Detailed monitoring of CNP activity between 2018-2021 demonstrated average coral outplant survivorship (up to 3 years post-outplanting) of 77% across 5 diverse reef systems (Scott et al., 2024) and positive outcomes of outplanting through enhanced recovery dynamics of key species (Roper et al., 2022), particularly at sites with lower initial hard coral cover (Howlett et al., 2023). Whilst stakeholder-led coral restoration models have demonstrated capacity to scale in reef regions elsewhere (e.g., The Caribbean, see Lirman and Schopmeyer, 2016; Bayraktarov et al., 2020; Carne and Trotz, 2021; Blanco-Pimentel et al., 2022; and Indonesia, Lamont et al., 2022), how the CNP model could be feasibly adopted outside of Cairns and Port

Douglas remained unknown. However, in August 2022, coral propagation and outplanting activity was implemented at three inshore, fringing-reefs in the Whitsundays (Coral Nurture Program Whitsundays, CNPW) to determine if and how activity could be tailored to this region.

As with Cairns and Port Douglas, the Whitsundays represents a major GBR tourism hub, where the reef tourism sector provides 28% of total employment (Tourism Research Australia, 2023) and generates upwards of US\$900,000/km^2 in estimated annual recreational and tourism ecosystem service value (Spalding et al., 2016, 2017). In March 2017, the Whitsundays was impacted by a slow-moving Category 4 tropical cyclone ("Cyclone Debbie"; Bureau of Meteorology, 2018), resulting in damage to tourism infrastructure and an average loss of 55% in coral cover in the region (Williamson et al., 2019). Whilst observations have documented early evidence of natural recovery via hard coral larval recruitment (McLeod et al., 2019; AIMS, 2022, 2023; Thompson et al., 2023), recovery of inshore coral assemblages has been challenged by persistent high nutrient and sediment loads from coastal runoff (Waterhouse et al., 2021; Thompson et al., 2023). Given the slow rate and suppressed capacity of natural recovery of reef habitats in the Whitsundays (Thompson et al., 2023), equipping tourism operators with new and additional site stewardship capacity may therefore support the assisted recovery of reef sites with high tourism ecosystem service value (Spalding et al., 2017), as in Cairns-Port Douglas (Howlett et al., 2022; Suggett et al., 2023). Coral restoration projects and techniques have been trialed in Whitsundays region in the last five years including asexual propagation and outplanting (Cook, 2022; McLeod et al., 2022), sexual larval re-seeding (McLeod et al., 2022), coral repositioning post-cyclone (McLeod et al., 2019), and a coral relocation project to mitigate construction damage (Smith et al., 2024). However, detailed implementation costs of these efforts, and specifically, detailed outcomes of asexual propagation and outplanting are unclear. For example, Cook (2022) reported survivorship of propagated corals in nurseries of approximately 70% after six months, yet the survivorship of coral fragments outplanted to the reef – a key outcome underpinning cost-effectiveness – remains unresolved.

Understanding the feasibility, and ultimately the sustainability of conducting restoration interventions, often rests on their financial viability or cost-effectiveness (Cook et al., 2017; Iacona et al., 2018; Suggett et al., 2023, 2024). However, in coral reef restoration practice, costs are rarely and inconsistently reported. Few reports detail project life-cycle costs including implementation, training, maintenance, and monitoring (Spurgeon and Lindahl, 2000; Bayraktarov et al., 2019) and/or quantify the contribution of 'in-kind' resources such as volunteer or researcher time (Edwards et al., 2010). Such a knowledge gap impedes collective understanding of the 'true costs' of restoration efforts (Hein and Staub, 2021), thereby limiting the ability of reef management, funding agencies and restoration practitioners to adequately budget for, invest in, and deliver effective and sustainable site intervention (Edwards et al., 2010; Bayraktarov et al., 2015, 2019; Suggett et al., 2023). Such data is especially sparse for the GBR, and previous evaluation of the cost-effectiveness of CNP outplanting activity in the Cairns-Port Douglas region (154 outplanting trips, 5 reefs, 3.5 years) (Scott et al., 2024), yielded variable mean 'realized' costs of coral outplanting (adjusted for outplant survivorship) spanning US\$3.0-21.0 coral^{-1} trip^{-1}. However, this cost-tracking exercise started mid-program, and thus failed to capture early implementation costs, which are inevitably prone to be higher. Initiation of CNP in the Whitsundays therefore provided an opportunity to track restoration costs more rigorously. To achieve this goal, we (i) detail the operational and environmental context for adoption of CNP activity in the Whitsundays, (ii) describe the implementation and associated costs of restoration activity and (iii) evaluate early-stage survivorship of coral outplants across three sites during the first nine months of establishment (August 2022 to June 2023). Collectively, we use these data to examine the early-stage cost-effectiveness of implementing asexual coral restoration practices in the Whitsundays via the CNP stewardship approach, relative to retaining new, surviving coral biomass at reef sites (costs less coral losses). We discuss the key achievements, challenges, and complexities of adapting the existing CNP reef stewardship approach from Cairns and Port Douglas to the Whitsundays reef system as a result of differing environmental conditions and tourism operational contexts.

2 Methods

2.1 Coral Nurture Program Whitsundays operational-ecological context and implementation

Coral Nurture Program Whitsundays (CNPW) is a partnership between researchers from the University of Technology Sydney (UTS) and three Whitsundays reef tourism operators, with coordination support from the local natural resource management organization, Reef Catchments (RC). The three tourism operators, already involved in other reef stewardship activities through the Great Barrier Marine Park Authority's (GRBMPA) "Reef Protection Initiative" (RPI), applied through an Expression of Interest to an open call to partner with CNPW. Establishment and operation of CNPW was financed through philanthropic funding in early 2022 (specifically "venture philanthropy"; Suggett et al., 2023) and supported through in-kind contributions via UTS, RC and tourism operators (herein referred to as 'operators').

CNPW coral propagation and outplanting activity was initiated at three fringing reef sites in the Whitsundays on Australia's Great Barrier Reef (GBR) in August 2022: "Blue Pearl Bay" (BPB) (20°2′48.91"S 148°52′5.76"E) on Hayman Island, "Black Island" (BI) (20°4′57.98"S 148°53′25.97"E) and "Luncheon Bay" (LB) (20°3′52.58"S 148°57′4.69"E) on Hook Island (referred to as "Outplanting sites"; Figure 1). Sites are located approximately 30 km offshore from Airlie Beach, on the north-western side of each respective island and hence were heavily exposed to high winds and storm surges generated by the south-western trajectory of Cyclone Debbie (2017). Whilst no historical data exists for the selected CNPW sites, declines in hard coral cover of 6-24% (2016-2020) were

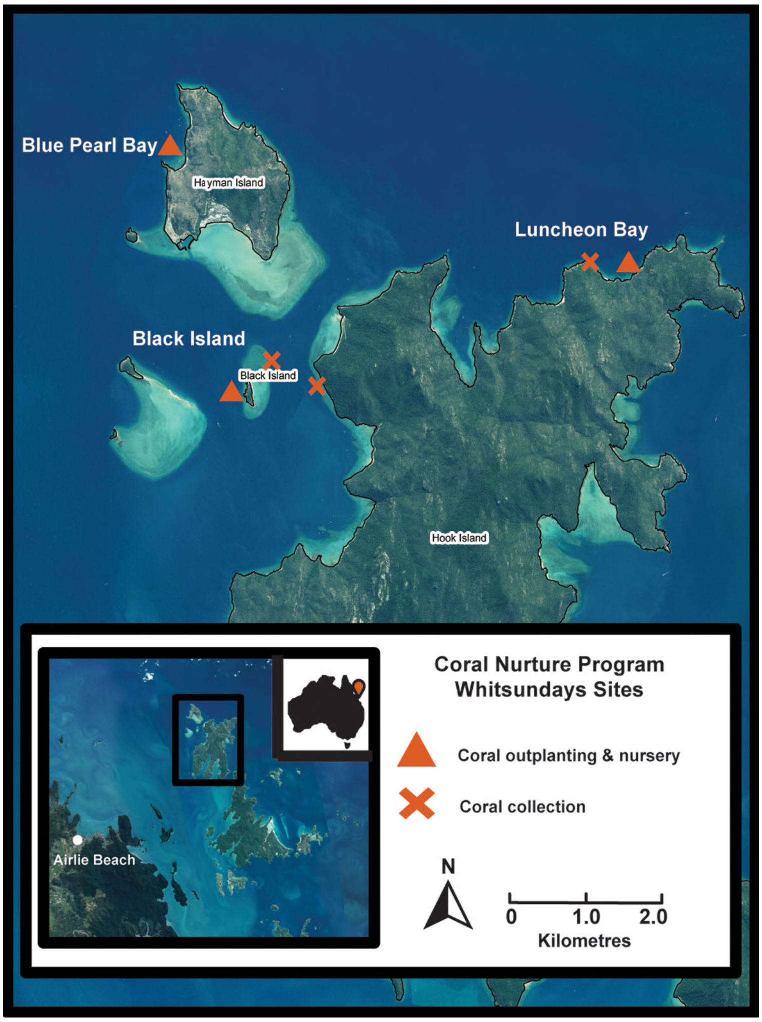

FIGURE 1
Map showing the locations of the three Coral Nurture Program Whitsundays (CNPW) 'coral nursery and outplanting' sites (triangles) and 'coral collection' sites (crosses) in the Whitsundays Islands, Queensland, Australia. All three 'outplanting' sites are located in Marine National Park Zones (no-take zones) of the Great Barrier Reef Marine Park.

documented at nearby reef sites on Hook and Hayman Island, where hard coral cover was estimated at ~15% at the time of CNPW initiation (AIMS, 2022, 2023). Preliminary benthic video surveys conducted at CNPW sites in 2020 and 2021, approximated hard coral cover at <7%, largely composed of 'massive' hard coral taxa (Supplementary Table 1). Even so, sites remain heavily frequented by tourism operators and private charter boats via shared public moorings. Outplanting sites were chosen, in consultation with local tourism operators, for both their operational suitability (i.e., ease of access for routine monitoring and maintenance, alignment to tourism-led stewardship and community engagement activities, and detailed local site knowledge), as well as habitat suitability for rehabilitation activities [e.g., exposure to offshore currents to mitigate sediment deposition (Ceccarelli et al., 2020)], and availability of consolidated substrate for attaching corals with Coralclip® coral attachment devices (Suggett et al., 2020) (Supplementary Table 1; Supplementary Figure 1A)].

Each of the three operators self-nominated as 'lead practitioner' for one of the three sites, based upon their regular visitation to nearby reef sites during routine tourism activity. In this way, operations aligned to those in CNP (Cairns-Port Douglas) where operators steward individual reef sites (Howlett et al., 2022). Specifically, for CNPW: Operator A, Blue Pearl Bay (equipped for 3-5 day diving liveaboard trips for up to 10 passengers, 3 crew); Operator B; Black Island (equipped for snorkeling trips for up to 30 passengers, 3 crew); Operator C; Luncheon Bay (equipped for snorkeling trips for up to 25 passengers, 2 crew). However, in the Whitsundays region, vessel moorings are largely public, and thus, Operators A-C are not the only vessels that visit CNPW sites and are not restricted to CNPW sites on their tourism days. Although each operator led stewardship of CNPW activity at their respective site, activities were largely conducted collaboratively with all three tourism operators, UTS researchers and the RC local coordinator. At project initiation, each operator agreed to the CNP code of operation, a set of key principles designed to align common goals, expectations, and trust across stakeholders (Howlett et al., 2022; see also coralnurtureprogram.org).

Over the nine month period examined in our current study, a variety of activities were undertaken to establish, manage and monitor coral restoration activity, and train restoration practitioners, for which associated costs were quantified (*'see 2.6 Quantifying implementation costs'* below). Activities are described below and further detailed Supplementary Data 1.

2.2 CNPW coral collection, nursery propagation and outplanting

Although installation and maintenance of coral nurseries can introduce considerable additional restoration costs (e.g., Edwards et al., 2010; Scott et al., 2024), establishing nurseries at CNPW outplanting sites was considered critical to overcome locally low coral cover and limited availability of naturally detached coral fragments ['corals of opportunity' (CoO)] for outplanting. In August 2022, table nurseries (n=3) were installed at each of the three restoration sites on sandy areas at a depth of 3-4 m (below low tide) (Figure 2A). Each nursery frame consists of a sheet of diamond aluminum mesh, secured to two parallel 85 x 85 cm stainless steel frames with stainless steel wire. Frames were anchored to the sand with steel rebar stakes and sit 50 cm above the substrate to minimize sedimentation exposure and facilitate water flow. Nursery tables were initially stocked with coral material, as permitted, from nearby donor reef sites ('collection sites') located within 10 km of CNPW 'outplanting' sites and with hard coral cover ranging from 17-38% (*see '3.1 Baseline benthic composition at CNPW outplanting and collection sites* below; Supplementary Figures 1C, 2): "Cockatoo Point" (CP) (20°4′57.42"S 148°53'41.82"E), "Wonderwall" (WW) (20°4′57.55"S 148°54'8.63"E), and "Luncheon Bay Donor" (LBD) (20°3′56.27"S 148°56'36.88"E) (Figure 1). Collected material was largely CoO but was occasionally supplemented with *in situ* fragmentation of donor colonies using a hammer and chisel or wire cutters to enhance diversity of propagated species (within permit conditions: <10% of parent colony, fragments >15 cm in size).

During all coral collection trips, corals were immersed in seawater and kept shaded for transportation by boat to the 'outplanting' sites (<1 hr transit), where they were immediately transferred back into the water. Once nurseries were stocked, colonies were photographed for species identification. On initial stocking, coral fragments were not tied down to prevent use of plastic cable ties (Boström-Einarsson et al., 2020; GBRMPA, 2020); however, coral loss due to suspected wave action and Bumphead parrotfish (*Bolbometopon muricatum*) predation (C. Hayward, J. Unsworth, E. Monacella; personal observation, December 2022), necessitated the use of cable ties to secure coral fragments during subsequent restocking trips.

At each CNPW 'outplanting' site, areas of reef were identified and designated for (i) outplanting, controls (no outplanting), (ii) marked, experimental fate-tracked plots to assess outplant survivorship (see *'2.4 Evaluating coral outplant survivorship'* below), and (iii) 'tourism outplanting areas', where CNPW operators could outplant corals. Outplanting activity was initiated in August 2022 and remains ongoing. During the study period (August 2022 to June 2023), operators could outplant corals at their own will, but instead opted for more coordinated outplanting efforts. Outplanting was therefore conducted collectively by tourism operator personnel and researchers across three 'outplanting blitz' events: at 'Site Setup' in August 2022; after six months during a 'Monitoring and Training' trip in March 2023; and during the global coral stakeholder-led restoration awareness initiative, "Coralpalooza™" in June 2023 (nine months post-establishment) where other volunteer tourism crew were also involved.

During all events, coral material was outplanted using Coralclip® (Suggett et al., 2020) on areas of bare, consolidated substrate adjacent to coral nurseries. A pre-outplanting demonstration was provided to all tourism operators, and initial outplanting efforts were evaluated visually by researchers to ensure proper and consistent deployment. Where possible, outplanted fragments were kept ≥10cm in length, oriented upwards, with exposed skeleton positioned flush with the substrate to encourage self-attachment (Lewis et al., 2022) and to avoid smothering by sediment. Coral material was photographed prior to outplanting for later identification and the number and taxonomy (identified to species where possible, otherwise genus and morphology) of outplants was reported to central CNP management via standardized reporting forms (Howlett et al., 2022; Scott et al., 2024).

2.3 Characterizing baseline benthic composition

Prior to the initiation of restoration activity, continuous line-intercept video transects (n = 3 x 30 m per site) were conducted by researchers (Howlett et al., 2022; Roper et al., 2022; Howlett et al., 2023) at CNPW outplanting sites and at coral collection sites to quantify baseline benthic composition (Figure 1). Transect tapes were laid consecutively along the reef slope (5–15 m apart), perpendicular to the shoreline at 3-5 m depth. Using a GoPro HERO 9®, a diver filmed ~10-20 cm above the transect tape, capturing the substrate directly beneath it. During analysis, substrate directly under the transect line was recorded to the nearest 5 cm and categorized as: hard corals (identified by genera), soft corals, macroalgae (including upright calcifying and fleshy macroalgae), consolidated substrate (rock, attached dead coral), unconsolidated substrate (unattached dead coral rubble and sand), or other invertebrates (e.g., zooanthids, fire coral (*Millepora* sp.)). Notably, all abiotic hard surfaces (i.e., rock, dead coral, dead coral rubble) at sites were covered in epilithic algae ranging 5-30 mm in thickness (Supplementary Figure 1A).

2.4 Evaluating coral outplant survivorship

Documenting outplant survivorship has been a central metric for tracking cost-effectiveness of coral restoration practices in many prior studies (e.g., Edwards et al., 2010; Humanes et al., 2021; Mostrales et al., 2022; Guest et al., 2023; Scott et al., 2024). Therefore, to benchmark initial coral outplant performance across the 'outplanting' sites, fate-tracked plots were established at program initiation, separate from areas designated for outplanting by CNPW operator staff during routine operations. At each site, triplicate 5-7 m^2 control and treatment plots (n=9 total) were each marked with ~10 cm stainless steel rebar stakes and masonry nails for resurvey. Plots at BI and BPB were located at depths of 2-4 m, whereas plots at LB were at depths of 5-7 m, owing to suitable outplanting substrate availability. In each treatment plot, 60-80 coral fragments comprising ~4 different branching species were outplanted largely by CNPW researchers, as well operator staff and volunteers as part of training. Lack of consistent coral material in sufficient quantity at ‘collection’ sites precluded full factorial replication by species, and hence 15-20 fragments of species of similar genera and/or growth morphologies were outplanted per plot, across sites: *Acropora millepora/Acropora spathulata*, *Acropora cerealis*, *Acropora intermedia/Acropora muricata*, *Pocillopora damicornis/Pocillopora verrucosa*. Coralclip®

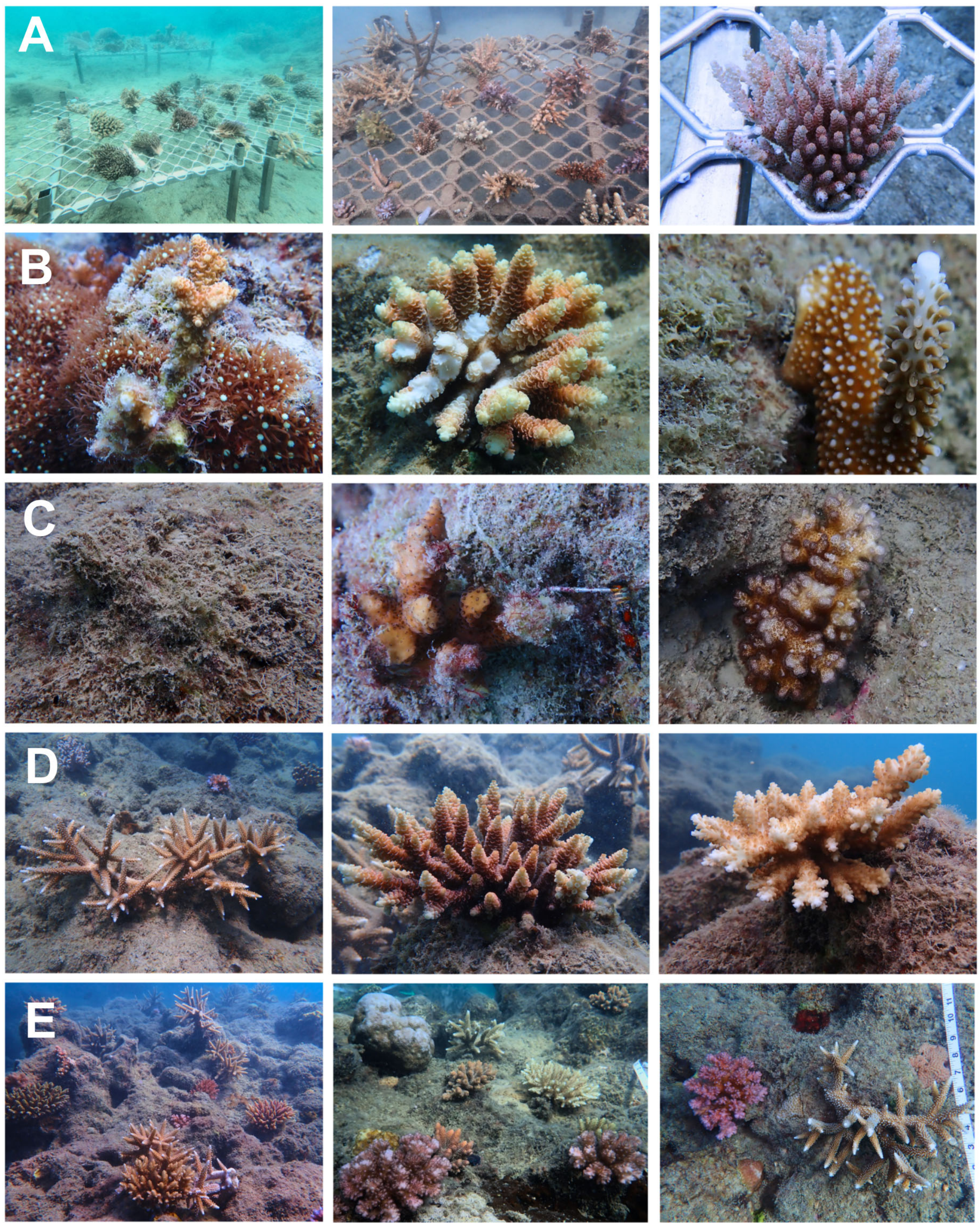

FIGURE 2
(A) Table nurseries installed at sites with larger coral fragments; a coral fragment wedged into the diamond-mesh of the nursery frame (fragments were subsequently secured with cable ties following predation and/or dislodgement). **(B–E)** Images depicting fragments outplanted in triplicate fate-tracked plots across the three CNPW sites. **(B)** depicts challenges experienced by coral outplants including competition with other benthic invertebrates (e.g., zooanthids, left), predation by corallivorous fish (middle) and difficulty self-attaching to algal-turf covered substrates (right). **(C)** depicts challenges in assessing outplant survivorship including smothering of empty Coralclip® units by turfing algae (left), and detachment of outplants and Coralclip® from substrate (middle, right). **(D, E)** depict surviving outplants after nine months in June 2023 (shown left to right).

attachment success and survivorship of coral outplants was visually assessed 1-month post-outplanting (T32-days, September 2022) via visual SCUBA-based surveys, where observed coral fragments were counted and categorized as coral alive (fragment attached, <100% mortality), coral dead (fragment attached and covered in turfing algae, 100% mortality), and coral missing (empty Coralclip® still in place, but fragment missing) (as per Suggett et al., 2020; Scott et al., 2024). Surveys were repeated five times over nine months at T67-days (November 2022), T109-days (December 2022), T191-days (March 2023), T232-days (April 2023), T267-days (June 2023). Counts in each replicate plot were pooled across genera/growth morphologies, except for the final timepoint where counts were conducted for each genera/morphology group. Monitoring was led by two tourism operator staff members and the RC local coordinator, with an accompanying CNP researcher to facilitate data collection and training.

2.5 Data analysis

Statistical analysis and data visualization were conducted in R (v4.0.0) (R Core Team, 2021). Variables for parametric analysis were visualized (qqplot and boxplots) and tested for normality (Shapiro-Wilk) and equal variance (Levene's test). P-values and P_{adj}-values (Tukey's and Bonferroni) less than alpha ($\alpha = 0.05$) were considered significant for all tests.

For benthic composition at sites, the cover of each benthic category (in cm) was expressed as a proportion of the total transect length per replicate transect, and visually compared using a principal components analysis (PCA). To compare benthic composition profiles, separate one-way ANOVA tests were conducted on the extracted ordination axes for principal component 1 (PC1) and principal component 2 (PC2) between sites and site type (collection and outplanting). Site-based differences in mean hard coral abundance (in cm) were also tested using one-way ANOVA. Following ANOVA tests, Tukey's HSD tests were conducted *post-hoc* to determine any significant differences between sites. To visualize mean proportional cover of hard coral genera at sites, stacked barplots were plotted.

For each fate-tracked plot, the count of alive outplants at each timepoint was expressed as a proportion of original outplants, and each of the three plots was used as a statistical replicate per site. In three instances, on occasions where the principal surveyor differed owing to logistical reasons, counts of 'alive' corals were underestimated (surrounded by higher survival values in the preceding and following timepoint, for which there was high confidence). To address this, anomalous values were substituted with the value from the preceding timepoint. Survivorship comparison between sites was performed using pairwise log-rank tests of survival probabilities, derived from Kaplan-Meier survivorship functions (Lee and Wang, 2003), with counts of dead or missing outplants observed in each plot at each timepoint as censored observations. This was performed on survival probabilities of coral outplants alive at T0, T32, T67 and T109 to assess site-based differences and patterns of decline over time. P-values from pairwise comparisons were adjusted with a Bonferroni correction.

2.6 Quantifying implementation costs

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

Costs were first calculated with in-kind contributions included (e.g., labor costs for volunteers, research students, local coordinator, and principal investigator), which are likely a closer reflection of the "true costs" of the intervention (Hein and Staub, 2021) and were thus categorized as "True Costs". However, to examine the costs associated with in-kind time contributions and researcher involvement, costs were again calculated without these as reflective of the actual costs to the CNPW (hence referred to as, "Actual costs"). Finally, to derive a per-coral cost (referred to as 'planting cost' for brevity (PC); US$. coral^{-1}) (Scott et al., 2024), both total "True" and "Actual" costs (i.e., the sum of all cost categories) were divided by the total number of corals outplanted during the study period. The 'realized' cost of activity (PC_R, US$ surviving coral^{-1}) was then estimated, whereby per-coral planting costs (PC) were multiplied by the mean proportion of surviving outplants in fate-tracked plots at the final monitoring timepoint (T267 days) (Edwards et al., 2010). Full details of the assumptions of analysis, and cost calculations are provided in Supplementary Data 1.

3 Results

3.1 Baseline benthic composition at CNPW outplanting and collection sites

PCA visualization of baseline benthic composition showed discrete clustering between CNPW outplanting and collection sites with some overlap (Figure 3C), suggesting differences in benthic communities. PC1 and PC2 accounted for 67.1% and 22.1% of the total variance in benthic cover respectively. ANOVA on these extracted ordination axes confirmed that 'collection' sites significantly differed from 'outplanting' sites along PC1 (ANOVA, $F_{1,16}$ = 42.73, p < 0.001), but not PC2 (ANOVA, $F_{1,16}$ = 0.675, p = 0.42; Supplementary Table 2). The greatest loadings contributing to differences along PC1 were consolidated substrate, soft coral cover and hard coral cover (Supplementary Table 3). Mean hard coral cover was highest at collection site CP at 38.06 ± 4.91% (± standard error, [SE]) of total benthic cover (Figure 3A), and was higher compared to collection site LBD, and the three outplanting sites (Supplementary Figure 2; ANOVA, $F_{5,12}$ = 9.047, p_{Tukey} < 0.001; Supplementary Table 4). At the other two collection sites, WW and LBD, mean hard coral cover was 17-22%; whereas at the outplanting sites, hard coral cover (mean ± SE) was 3.22 ± 0.87%, 7.56 ± 3.42% and 8.67 ± 4.67% at LB, BPB and BI respectively (Figure 3A). The three outplanting sites did not differ in either benthic composition (Tukey's *post-hoc*, p_{Tukey} > 0.05; Supplementary Table 5), or hard coral cover (Supplementary Figure 2; Tukey's *post hoc*, p_{Tukey} > 0.05; Supplementary Table 4). Hard coral cover at outplanting sites was largely composed of genera with massive, submissive and encrusting morphologies with low structural complexity (Figure 3B). Turfing algae cover was present on consolidated rock at all sites but was lowest at collection site LBD (R. Scott, personal observation, August 2022). Macroalgae cover was only observed at outplanting sites BPB and LB, but not BI (Figure 3A).

3.2 CNPW nursery propagation and outplanting activity

During the study period, 4,425 coral fragments were collectively outplanted by CNPW tourism operators, CNP researchers and volunteers at the three CNPW outplanting sites, inclusive of 631 outplants in fate-tracked plots and 3,794 outplants in 'tourism outplanting areas' (Table 1). In total, 15 staff members across Operator A-C, and 9 additional volunteers from other Whitsundays' tourism operators were trained in outplanting with Coralclip®. Between 25-30 different coral species were outplanted across the three sites, of which 68-87% were *Acropora* spp. and 10-18% were *Pocillopora* spp. (detailed in Table 2). Other branching coral species from genera *Echinopora, Porites, Stylophora* and some species with encrusting and massive morphologies were also outplanted.

Nursery frames across sites were stocked with 15-21 species of coral, of which approximately 65% were from the genus *Acropora*. Other genera included *Echinopora, Isopora, Montipora, Pocillopora, Porites*, and *Turbinaria* (detailed in Supplementary Table 6). After 1 month, some coral fragments were observed to self-attach to the nursery frame, but by 3 months several colonies were dislodged (BPB: 12-19%, BI: 7-28%, LB: 30-55%), potentially due to initially being unsecure and/or predated upon. Consequently, frames were restocked with new colonies (secured to frames with cable-ties) in March 2023. To provide time for coral colonies on nurseries to establish, no nursery corals were removed from frames for outplanting during the study period. Furthermore, 1-month post-establishment, sponges, ascidians, turfing and filamentous

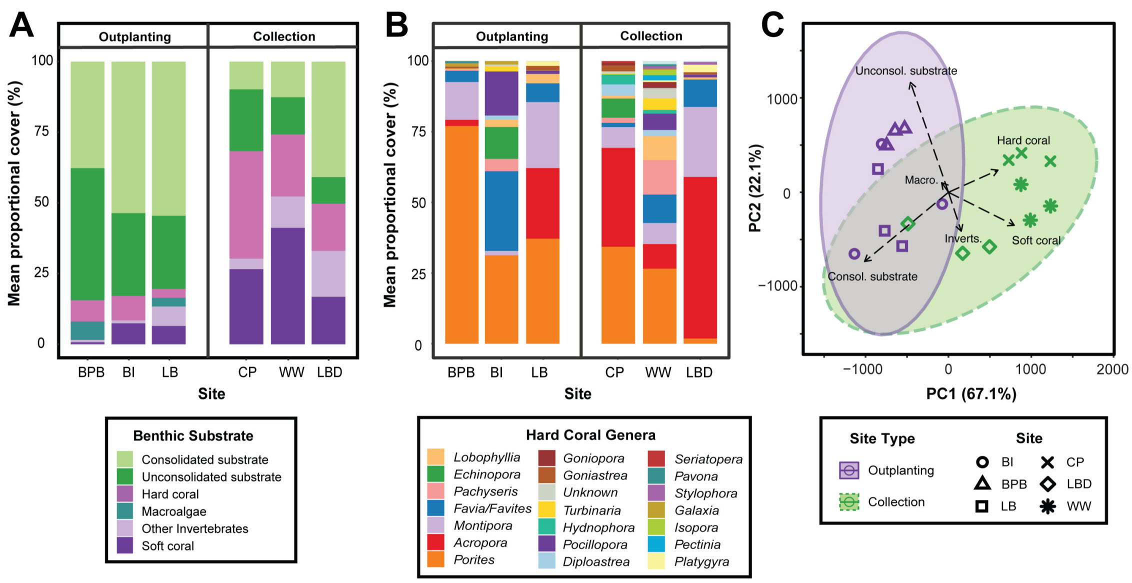

FIGURE 3
(A) Mean proportional coverage of benthic substrates at Coral Nurture Program Whitsundays (CNPW) outplanting (BPB, Blue Pearl Bay; BI, Black Island; LB, Luncheon Bay) and collection (CP, Cockatoo Point; WW, Wonderwall; LBD, Luncheon Bay Donor) sites from triplicate 30m benthic video transects in August 2022. **(B)** Mean cover of hard coral genera* as a proportion of total hard coral coverage at CNPW outplanting and collection sites. *Based upon capacity to identify corals from videos, *Favia* and *Favites* are conservatively grouped together **(C)** Principal components analysis (PCA) of benthic categories grouped by site type (outplanting and collection). Ellipses show 95% CI. PCA loadings of benthic categories (shown as dashed arrow vectors) were scaled to PCA eigenvalues, with vector length indicating the strength of this contribution. Vector direction shows the contribution of each variable to the principal components (PCs): Inverts: other invertebrates, Macro: fleshy and upright calcifying macroalgae, Consol. susbtrate: consolidated hard coral rock covered in turfing algae of varying depths, Unconsol. substrate: unconsolidated sand, coral rubble and dead coral.

TABLE 1 Number of coral fragments at each site collectively outplanted by the three CNPW tourism operator partners, researchers and volunteers during three deployments from August 2022 - June 2023.

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

Corals outplanted during "Site Setup" include outplants in fate-tracked plots.

macroalgae had colonized nursery frames which smothered some smaller coral fragments, and sediment was observed accumulating on plating/foliose colonies (e.g., *Turbinaria*). As per permitting requirements, any diseased and dead colonies were removed from frames during the 6 monitoring trips conducted between September-June 2023 and filamentous algae was removed from any affected colonies. However, time-constraints on these trips precluded intensive cleaning of nursery frames. Frames were intensively scrubbed of fouling organisms at each site during nursery restocking trips in February/March 2023 and again in May/June 2023.

3.3 Coral outplant survivorship in fate-tracked plots

Mean (± SE) survival of outplants after nine months (T267 days post-outplanting) was higher at BI (47.58 ± 3.56%) than LB and BPB, which was 25.70 ± 2.31% and 23.33 ± 2.96%, respectively (Figure 4; Table 3). Throughout this study, mean outplant survival was higher by 9.46-40.48% at BI (across timepoints) than BPB and LB. From T0, Kaplan–Meier survival probability functions significantly differed between BI and the two other sites (Figure 4) (Pairwise log-rank, BI-BPB: p_{Bonf} = 9.6 x 10^{-7}; BI-LB: p_{Bonf} = 1.5x10^{-8}; LB-BPB: p_{Bonf} = 0.72; Supplementary Tables 7, 8), and median survival time across sites was 109 days. When survival probabilities were examined from successive timepoints, BPB was significantly lower than BI and LB based on surviving outplants at T32 days and T67 days (Supplementary Figure 3; Supplementary Table 8). However, no significant differences were observed between sites from T109 days (~ five months) post-outplanting, after which time the probability of survival for remaining outplants to the end of the study (T267) was 70-80%, (Supplementary Figure 3; Supplementary Table 8) suggesting declining mortality over time as surviving outplants established (Figures 2D, E). After nine months, branching *Acropora* species showed higher mean survivorship at BI than the other two sites, whereas at LB, *Pocillopora* outplants showed higher mean survivorship than outplants of *Acropora* genera (Supplementary Figure 4).

Interestingly, despite a large initial decline in observed surviving outplants at LB by T32 days (Figure 4), the mean proportion of empty Coralclip® units, as well as dead, but still attached coral outplants was only 5.7% ± 1% and 3.9 ± 1.5%, respectively. Low survivorship was therefore primarily explained by the proportion of 'corals unaccounted' (i.e., where both the coral fragment (alive or dead) and Coralclip® unit were missing). Indeed, between 33-47% of original Coralclip® units (outplants) were unaccounted for across sites after nine months (Table 3). Whilst the cause of this high proportion of 'unaccounted' Coralclip® units is unclear, it can reflect an underestimation of 'empty' Coralclip® units because of smothering by turfing algae and sediment, and/or complete dislodgement from the substrate (Figures 2B, C). Across sites,

TABLE 2 The number of different coral species and relative abundance (%) of total outplants by coral genera outplanted at the three CNPW sites from August 2022 – June 2023.

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

'Sp./Spp.' denotes where coral species could not be identified.
"Other" denotes coral genera contributing to <1% of total outplant number. At first mention, Acropora and Pocillopora genera are named in full, with genera name subsequently abbreviated for brevity.

'empty' Coralclip® observations steadily increased to 17-30%, on average, by T267 days (Supplementary Figure 5; Table 3), whereas the proportion of observed dead, attached outplants was <5%, which suggests declining survivorship at sites was likely driven by coral dislodgement.

3.4 CNPW implementation costs and 'realized costs'

When all cost categories were collated (including in-kind costs), total "True" costs for the first nine months of coral propagation and outplanting activity at the three CNPW sites was US$253,800.38. Based on the number of corals outplanted during this timeframe, this capital and operational expenditure yields an effective per-coral 'planting cost' (PC) of US$57.36 coral^{-1} (n = 4,425) (Table 4). However, if only 'outplanting' costs were considered (e.g., as per Scott et al., 2024), PC during this timeframe was $10.63 coral^{-1} (Supplementary Table 9; Supplementary Data 2). Overall, 'vessel use' and 'overheads' were the cost categories which accounted for the greatest contributions to total 'True costs' (30% and 48% of costs, respectively, Table 4). As such, when in-kind costs associated with 'overheads' (i.e., researcher and local coordinator time) as well as volunteer labor were *not* included, total 'Actual' costs were 44% lower (US$143,549.05), yielding a PC of US$32.44 coral^{-1} (Table 4).

In-kind costs significantly weighted the 'True cost', and therefore the value, of coral outplants. The costliest activities contributing to the large discrepancy between 'True' PC and 'Actual' PC were 'Project Planning, Management and Administration' activities (49% of the total) due to high overhead costs associated with salaries. This was followed by 'Outplanting', 'Coral material collection' and 'Monitoring' activities which accounted for 19%, 10% and 7% of costs respectively (Table 4), as these activities required the most labor and the greatest proportion of vessel time. Travel and accommodation costs for UTS researchers to travel to Airlie Beach for activities accounted for 6% of total costs.

Further adjusting 'True costs' by the mean nine-month (T267 days) survivorship of outplants in fate-tracked plots (Table 3) resulted in a realized cost (PC_R) of $178.12. coral^{-1} overall but ranging $120.55 - $245.85. coral^{-1} depending on site-based survivorship (Table 5). Again, when only 'Outplanting' costs were considered (i.e., as per Scott et al., 2024), PC_R was $33.04 coral^{-1} (Supplementary Table 9; Supplementary Data 2). When in-kind costs were excluded from total costs ('Actual cost'), PC_R was substantially lower at US$100.75 ($68.18 - $139.05) (Table 5), demonstrating how the methods with which costs are calculated and survivorship is assessed substantially impacts PC and PC_R.

4 Discussion

Further investment and application of reef restoration interventions, including on Australia's GBR, hinges upon addressing uncertainties around the feasibility of approaches for different reef environments (McLeod et al., 2022). A central factor underpinning reef restoration feasibility is comprehensive and transparent understanding of the associated costs – including those potentially unaccounted for as in-kind contributions or project overheads (Edwards et al., 2010; Iacona et al., 2018; Bayraktarov et al., 2019) – and the likelihood of 'success' in terms of delivering on program goals (Bayraktarov et al., 2019). However, owing to the relative novelty of restoration-based management approaches on the GBR, reports on restoration outcomes (e.g., Howlett et al., 2021, 2022; Cook et al., 2022; Roper et al., 2022; Howlett et al., 2023; Randall et al., 2023; Smith et al., 2023; Nuñez Lendo et al., 2024; Smith et al., 2024) are early-stage (<5 years), and reports of restoration costs are rare (Suggett et al., 2020, 2023; Scott

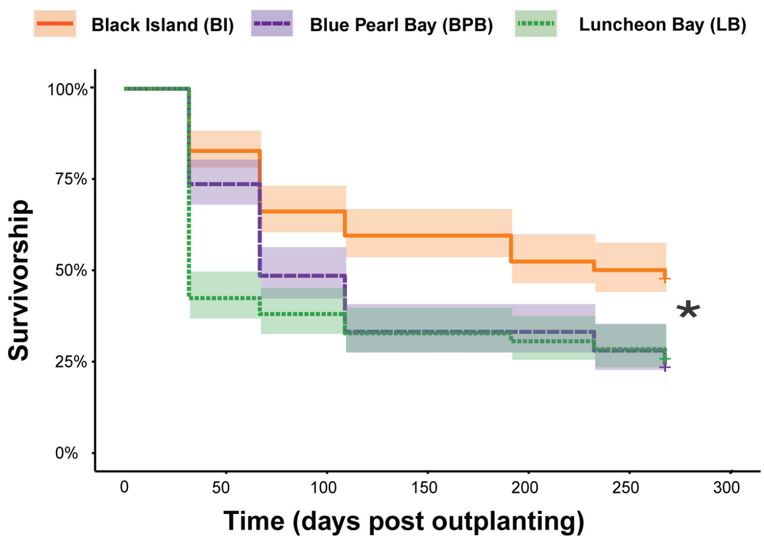

FIGURE 4
Kaplan-Meier survivorship probability for coral outplants monitored over 267 days in fate-tracked plots (n=3) at Black Island (orange line), Blue Pearl Bay (purple dashed line) and Luncheon Bay (green dotted line). Shaded lines indicate 95% confidence intervals. Asterisk denotes significant difference between sites (BI-LB; BI-BPB, see main text).

TABLE 3 Outcomes of coral fragments outplanted with Coralclip® in triplicate fate-tracked plots at the three CNPW sites after nine months (T267 days).

Coral outplant survivorship outcomes at nine months (as proportion of original outplants)			
	Black Island (BI)	Blue Pearl Bay (BPB)	Luncheon Bay (LB)
% corals alive	47.58 ± 3.56	23.33 ± 2.29	25.69 ± 2.31
% Coralclip® empty	17.21 ± 2.39	29.52 ± 5.16	22.80 ± 5.42
% corals dead, attached	2.40 ± 0.50	0.00	4.23 ± 2.22
% corals unaccounted	32.81 ± 4.12	47.15 ± 5.89	47.28 ± 6.17

Shown is the mean proportion (± standard error) of corals 'alive', 'dead', and 'Coralclip® empty' relative to the number of original corals outplanted in plots. 'Corals unaccounted' is the proportion of original outplants that could not be accounted for (i.e., coral fragment and Coralclip® missing).

et al., 2024; Smith et al., 2024). Whilst reef restoration approaches have been implemented in the Whitsundays region during the last five years (McLeod et al., 2019; Cook, 2022; McLeod et al., 2022; Smith et al., 2024), detailed costs of coral propagation-based restoration implementation are currently unclear. We therefore tracked costs from the outset of the Coral Nurture Program Whitsundays (CNPW) to conduct a more robust cost evaluation exercise for the CNP. Here, we discuss the costs and 'realized' costs of CNPW relative to the local operational and environmental context and discuss the importance of comprehensive cost-tracking to support decision-making processes in reef restoration.

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

By comprehensively cost-tracking all CNPW activity from the planning phase, we show that the costs of outplanting represent only a proportion of total project costs (albeit significant: ~20% of costs here and ca. 30-50% of project costs elsewhere, e.g., Edwards et al., 2010; Toh et al., 2017; Humanes et al., 2021) (Table 4). This is consistent with findings for CNP Cairns-Port Douglas where we previously determined that outplanting costs (~ca. US\$2.34 coral^{-1} trip^{-1} from >30,000 outplants) increased 2-to-6-fold where time allocation to additional and essential nursery propagation, site maintenance and practitioner training was accounted for in addition to outplanting cost (Scott et al., 2024). When all costs were considered, the total "True" restoration costs for early-stage activity quantified here (US\$253,800.83, \$57.36 coral^{-1}) is slightly above the median cost previously determined in a global review by Bayraktarov et al. (2019) for 20 coral propagation and outplanting projects (2010 US\$218,305 ha^{-1} yr^{-1}), but noting we have not derived a per-hectare cost in this current exercise. Early-stage costs for CNPW are similar to estimates for coral propagation and outplanting programs in Latin America (e.g., Sociedad Ambiente Marine, Puerto Rico (US\$50.26 coral^{-1}); reported in Bayraktarov et al., 2020), and 'realized' costs for corals outplanted on a seawall in Singapore (US\$122.80 coral^{-1}; Toh et al., 2017) under similar ecological and socioeconomic conditions [e.g., inshore location with high sedimentation, 'high income' country (Bayraktarov et al., 2019)]. In the Whitsundays, at a nearby site, Smith et al. (2024) reported an aggregate cost of AU\$38,500 [~ US\$26,500 (US\$2020)] to relocate and monitor 204 coral colonies over 2 years, though importantly, this was undertaken as an impact mitigation strategy for the construction of a submarine pipeline, rather than to propagate and outplant new coral biomass for assisted site recovery. Restoration costs inevitably vary based on the rationale, goals, scale, methods, materials, and logistical requirements unique to each project, as well as cost-accounting depth, methodology and timeframe (Scott et al., 2024). As such comparing or extrapolating costs between projects, even for projects in the same location, is challenged by myriad contextual caveats, underscoring the importance of comprehensively tracking and reporting primary cost data for restoration activity (Bayraktarov et al., 2019; Suggett et al., 2024). Transparently reporting costs according to standardized frameworks that capture important contextual information, assumptions and metadata (e.g., Edwards et al., 2010; Cook et al., 2017; Iacona et al., 2018; White et al., 2022) can enable such contextual differences to be considered.

Our findings further demonstrate how substantial the costs associated with essential program management, administration, and planning are to restoration projects and highlight the significance of in-kind contributions of volunteer, stakeholder, and researcher time. For large-scale coral propagation and outplanting programs in the Maldives (Montoya-Maya et al., 2016, reported in Bayraktarov et al., 2019) and the Florida Keys (Coral Restoration Foundation, 2023), costs involved in project overheads, fundraising, research, and development similarly accounted for a significant proportion of total expenses (approximately 20-40%). Elsewhere on the GBR, restoration activity on Fitzroy Island has been enabled through in-kind contributions of time, labor and support totaling AU\$150,000 [~ US\$103,000 (US\$2020)] year^{-1} (McLeod et al., 2022). In the instance of CNPW, many essential overhead expenses were 'in-kind' (i.e., salaries paid via other funding sources and/or organizations) as a result of collaboration with a research institution and a local natural resource management organization. Hence, "actual" or direct costs to the Program's budget were reduced by 44%. Such an exercise highlights how essential collaboration and associated in-kind contributions are to cost-effective restoration, which have been shown to effectively halve reported project costs in reef regions globally (dela Cruz et al., 2014; Toh et al., 2017; Bayraktarov et al., 2020; Suggett et al., 2020). However, such critical costs are not free and though paid for elsewhere, are often 'invisible' in restoration project costings (where reported) (Iftekhar et al., 2017), potentially due to an absence of comprehensive cost-tracking capability in stakeholder programs (Iacona et al., 2018; Ferse et al., 2021; Scott et al., 2024) or publishing bias towards successful or low-cost interventions to access competitive grant funding (Edwards et al., 2010; Bayraktarov et al., 2015; Boström-Einarsson et al., 2020). Such under-reporting obscures the "true" costs of restoration efforts (Hein and Staub, 2021), and ultimately disadvantages collective

TABLE 4 Costs (US$) of Coral Nurture Program Whitsundays (CNPW) implementation over nine months by cost category (top row) and activity (first column).

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

Proportional contribution of each category and activity to the overall total project cost ("True cost") is also presented.
Cells with '-' indicate where no cost was incurred. 'Planting Cost' (PC, US$ coral^{-1}) is the total costs relative to the 4,425 coral outplants deployed August 2022 – June 2023. Full costings are presented in Supplementary Data 1.

restoration practice through inhibiting adequate investment and effective budget forecasting for sustained restoration (Suggett et al., 2023) or project initiation elsewhere (Edwards et al., 2010).

4.2 Survivorship-based 'success' varies by site

When reported, coral restoration costs have often been weighted relative to outplant survivorship to yield a cost per surviving coral (referred to here as "realized cost", Scott et al., 2024; see also Edwards et al., 2010; Toh et al., 2017; Bayraktarov et al., 2019; Harrison et al., 2021; Humanes et al., 2021). Survivorship of fate-tracked outplants at CNPW sites displayed distinct site-differences and after nine months (23-48%), was lower than previously assessed fate-tracked outplants for the CNP in Cairns-Port Douglas (which ranged 32-93% in the first year) (Howlett et al., 2022; Strudwick et al., 2023; Scott et al., 2024). As such, CNPW realized costs (PC_R) increased substantially to >US$100 coral^{-1} (ranging US$68-$180 coral^{-1} based on all costed activities (depending upon site and inclusion of in-kind costs, Table 5) or US$33.04 coral^{-1} based on costs for outplanting only (Supplementary Table 9). This contrasts with outplanting PC_R for CNP (Cairns-Port Douglas) of US$2.99 ± 0.24 coral^{-1} trip^{-1} (Supplementary Table 9) (Scott et al., 2024). Lower survivorship and higher 'realized' costs for early CNPW activity are perhaps unsurprising given that poor water quality and proximity to land has been associated with lower outplant survival in reef regions globally (Foo and Asner, 2021). Other outplanting studies at sites impacted by poor water quality have reported variable, species-dependent outplant survivorship estimates of between 40-80% (e.g., Ferse et al., 2013 in Indonesia; Horoszowski-Fridman et al., 2015 in Eilat, Egypt, Bayraktarov et al., 2020 in Costa Rica; Toh et al., 2017 in Singapore). In our study, survivorship was similarly variable across coral genera/morphology groups, depending on site, but was not consistently higher for a particular species or morphology across all sites (Supplementary Figure 4). Interestingly, at a nearby Whitsundays site, Smith et al. (2024) reported high survivorship of

TABLE 5 Realised costs (PC_R, US$ surviving coral^{-1}) of CNPW implementation relative to outplant survivorship (as a proportion of original outplants, Table 3) in fate-tracked plots after nine months (T6, 267 days).

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

95% after 12 months and 77% after 2 years for relocated coral colonies (40-150cm) of largely massive morphologies attached using cement. Whilst several factors may have contributed to these comparably high survival rates [including attachment method, transplant size, and small translocation distance (20-100 m)], survivorship was considerably lower for transplanted branching morphologies (44%, noting n < 10), suggesting that more stress-tolerant, massive species may perform better under the high sediment and nutrient loads in the Whitsundays region (Anthony and Fabricius, 2000; Morgan et al., 2020; Thompson et al., 2023). Additionally, though we endeavored to outplant fragments >10cm, employing larger, whole colonies may mitigate some of the potential stress induced by fragmentation, sedimentation, and turfing algae competition that likely challenged coral fragment self-attachment in our current study.

Regardless of site, declines in surviving outplants were primarily explained by coral fragment and/or Coralclip® dislodgement. Low detachment rates for Coralclip were documented by Suggett et al. (2020) (2-7%, 4-7 months post-outplanting), but higher rates (~20%) have been observed at certain CNP sites in Cairns-Port Douglas between 9-12 months post-outplanting (Scott et al., 2024), and in restoration programs elsewhere during early-stage outplant establishment (e.g., 30%, in Horoszowski-Fridman et al., 2015). In this study, dislodged, surviving fragments were occasionally found nearby outplant areas during surveys, and often self-attached to the Coralclip® (Figure 2C). Successful outplant self-attachment to the reef substrate was challenged by turfing algae and sediment accumulation, thereby likely increasing chances of physical dislodgment by strong water movement, and fish grazing activity or predation (Figure 2B) (challenges similarly noted at Whitsundays sites; Cook, 2022; Smith et al., 2024). Although care was taken in selecting and handling collected Corals of Opportunity (CoO) to minimize stress, successful outplant self-attachment may also be impacted where coral material is weakened at the time of collection, by transfer or fragmentation stress, or adaptation to the outplant site (Forrester et al., 2012). Together, these factors highlight how careful selection of outplant material, consolidated substrate, and rigorous removal and maintenance of turfing algae will be essential for the success of outplanting methods at CNPW sites. Notably, the outplant survivorship reported here represents the initial trials of CNP techniques at Whitsundays sites. Further systematic investigations into the site-specific drivers of outplant detachment and mortality, informed by tourism operator observational knowledge, can guide a process of adaptive learning for optimizing outplanting practices (e.g., considering species selection, fragment size, fragment source; Howlett et al., 2022; seasonal algae growth, Brodie et al., 2012; fish interactions, Seraphim et al., 2020; and attachment method; Suggett et al., 2020). Resolving such factors and integrating knowledge into outplanting methods through ongoing monitoring practices may improve survivorship outcomes and realized costs with outplanting experience over time. Ultimately trialing other restoration techniques, such as MARRS Reef Stars (see Nuñez Lendo et al., 2024), or alternative means for coral attachment (e.g., cement) will further inform best-practice restoration approaches in the Whitsundays region and other inshore reef environments.

4.3 Operational-environmental context influences cost-effectiveness of coral restoration

Costs of coral restoration are highly specific to local context, including location, scale, restoration method (Bayraktarov et al., 2015, 2019), and project goals (Hein et al., 2021). For the CNPW, it was apparent that the CNP site stewardship model, originally conceived for seamless integration of coral propagation and outplanting activity into routine tourism operations (Howlett et al., 2022), required adaptation to the unique tourism operational and environmental context in the Whitsundays. For example, low and patchy coral cover at sites targeted for CNPW outplanting required coral collections from more abundant and diverse adjacent sites. These ecological conditions, combined with operational factors such as smaller vessel and crew capacity (compared to Cairns and Port Douglas operations, Scott et al., 2024), meant that CNPW activity during routine tourism trips was largely limited to visual monitoring of restoration sites and nursery structures, precluding regular, *ad hoc* outplanting and nursery maintenance. However, such factors collectively impacted coral outplanting output and had significant cost implications. For example, dedicated vessels for restoration activity absorbed 30% of total expenditure, and costs for coral collection at donor sites accounted for ca. 10% of total costs. These represent common and significant costs in coral restoration (e.g., Edwards et al., 2010) that by comparison are not typical for CNP operations in Cairns-Port Douglas (Howlett et al., 2022; Scott et al., 2024). As such, noting differences in costing methodology (related to outplanting and diving gear calculations, currency year etc., see Supplementary Data 2), there was almost a 5-fold difference in outplanting-only PC estimates across the two programs (Supplementary Table 9) which was largely the result of CNPW vessel cost requirements. Such higher costs may be reflective of future restoration scenarios, where repeat disturbances challenge coral material availability and survivorship or where selected-for material is sourced from land-based aquaculture operations (Gibbs, 2021; Banaszak et al., 2023).

It is important to reiterate that the current study provides cost-analyses at the early-stage of CNPW, and assessments of cost-effectiveness are dependent upon chosen outcomes and are dynamic based upon the timeframe over which they are evaluated (e.g., Harrison et al., 2021; Humanes et al., 2021; Guest et al., 2023). Further cost-evaluations are needed to assess whether PC and PC_R increase or decrease with increasing scale of operations (although evidence for 'economies of scale' in coral restoration is not yet apparent; Bayraktarov et al., 2015; Hughes et al., 2023; Suggett et al., 2024), as well as to capture the influence of disturbance events and adaptive outplanting practice on survivorship outcomes (Iacona et al., 2018). The costs of program establishment (e.g., nursery installation, coral collection, planning, researcher travel for training and monitoring) were significant. PC may decline as the program transitions from 'launch' phase to sustained operations, with reduced need for researcher involvement and training, greater outplanting experience of tourism operator staff, and establishment of coral nursery colonies that provide a self-sustaining source of coral material. While the 'realized' cost per coral (PC_R) during the establishment phases of CNPW may be perceived as high, the low underlying coral cover at high-value CNPW tourism sites may justify intensive efforts to improve site conditions. For example, if considered relative to the region's estimated annual tourism ecosystem service value (approx. US \$900,000/km^2) (Spalding et al., 2016; De Valck and Rolfe, 2018; Suggett et al., 2023), costs incurred may deliver positive cost-benefit in retaining such value (Naidoo et al., 2006). Further work is needed to confirm this notion via detailed cost-benefit analyses and longer-term, goal-based ecological monitoring that can capture ecological changes underpinning ecosystem function, resilience, and associated ecosystem service value (e.g., Hein et al., 2017; Ladd et al., 2019; Goergen et al., 2020).

Our study benchmarked costs by initial outplant survivorship across nine months, yet we acknowledge that realized costs are likely to increase in the context of dynamic reef systems undergoing intensifying stress events (Reimer et al., 2024). A challenge for coral restoration, particularly as discussions intensify for a biodiversity credit framework and coral mortality events become more frequent, will be establishing a suitable metric and endpoint to assess "success" or "effective" restoration (Edwards et al., 2010; Suggett et al., 2023; GFCR, 2023). Whilst PC_R is a useful indication of technique feasibility in the program establishment stage (i.e., within the first year), it is ultimately limited in its entirety to describe 'realization' of wider restoration goal-related outcomes, e.g., ecological or aesthetic recovery, particularly without comparison to 'wild' coral populations or control areas (i.e., no intervention) (Ferse et al., 2021; Gouezo et al., 2021; White et al., 2022; Hughes et al., 2023). Future long-term cost-analyses may therefore consider transitioning from outplant survivorship as an 'effectiveness' metric to expressing costs relative to areal coral gain (e.g., % change in coral cover), an area-based ecosystem service indice (e.g., Stewart-Sinclair et al., 2021; Suggett et al., 2023), or gain in biodiversity indices (e.g., abundance and diversity of key species (e.g., Goldstein et al., 2008; Abrina and Bennett, 2021) to characterize cost-effectiveness in relation to goals of assisted site recovery.

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

5 Conclusions

Early assessments of coral restoration operational models, techniques and cost-effectiveness are essential to inform ongoing implementation and adaptive practice, and build public, stakeholder and management trust (McLeod et al., 2022; Quigley et al., 2022; Suggett et al., 2023). Here, we have described the adaptation of the existing CNP tourism-led assisted reef recovery approach to three inshore fringing reef sites and different tourism operations in the Whitsundays. We show that activities often unquantified in the delivery of restoration programs (e.g., overheads, planning, in-kind contributions) contribute significant costs, and should thus be included in future cost-tracking efforts for transparent and effective budgeting. Furthermore, we show that monitoring and accounting for initial outplant survivorship to benchmark 'realized' costs can elevate cost-estimates significantly but is critical to inform adaptive learning processes and resource allocation, beyond simply recording "success". We highlight that long-term and locally tailored socio-economic and ecological monitoring is needed to improve holistic understanding of reef restoration cost-benefits to inform sustained financing.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

RS: Visualization, Writing – original draft, Writing – review & editing, Project administration, Conceptualization, Data curation, Formal analysis, Investigation, Methodology. DS: Conceptualization, Project administration, Supervision, Writing – review & editing, Funding acquisition, Investigation, Methodology. CH: Conceptualization, Funding acquisition, Investigation, Project administration, Writing – review & editing. BC: Resources, Writing – review & editing. JE: Conceptualization, Investigation, Methodology, Writing – review & editing. JG: Conceptualization, Data curation, Funding acquisition, Project administration, Writing – review & editing. GG: Data curation, Investigation, Methodology, Writing – review & editing. LH: Data curation, Investigation, Project administration, Writing – review & editing. EM: Investigation, Writing – review & editing. CR: Data curation, Investigation, Writing – review & editing. PS: Data curation, Investigation, Project administration, Writing – review & editing. JU: Investigation, Writing – review & editing. MV: Investigation, Writing – review & editing. SW: Formal analysis, Methodology, Writing – review & editing. EC: Conceptualization, Funding acquisition, Methodology, Project administration, Supervision, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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Publisher's note

Supplementary material

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fmars.2024.1418784/full#supplementary-material

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