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RESEARCH ARTICLE

Large-scale coral reef rehabilitation after blast fishing in Indonesia

Susan L. Williams[1†], Christine Sur[2,3], Noel Janetski[4], Jordan A. Hollarsmith[2,3], Saipul Rapi[4], Luke Barron[4], Siobhan J. Heatwole[4,5], Andi M. Yusuf[4‡], Syafyudin Yusuf[6], Jamaluddin Jompa[6], Frank Mars[7]

The severely degraded condition of many coral reefs worldwide calls for active interventions to rehabilitate their physical and biological structure and function, in addition to effective management of fisheries and no-take reserves. Rehabilitation efforts to stabilize reef substratum sufficiently to support coral growth have been limited in size. We documented a large coral reef rehabilitation in Indonesia aiming to restore ecosystem functions by increasing live coral cover on a reef severely damaged by blast fishing and coral mining. The project deployed small, modular, open structures to stabilize rubble and to support transplanted coral fragments. Between 2013 to 2015, approximately 11,000 structures covering 7,000 m^2 were deployed over 2 ha of a reef at a cost of US$174,000. Live coral cover on the structures increased from less than 10% initially to greater than 60% depending on depth, deployment date and location, and disturbances. The mean live coral cover in the rehabilitation area in October 2017 was higher than reported for reefs in many other areas in the Coral Triangle, including marine protected areas, but lower than in the no-take reference reef. At least 42 coral species were observed growing on the structures. Surprisingly, during the massive coral bleaching in other regions during the 2014–2016 El Niño–Southern Oscillation event, bleaching in the rehabilitation area was less than 5% cover despite warm water ($\geq$30°C). This project demonstrates that coral rehabilitation is achievable over large scales where coral reefs have been severely damaged and are under continuous anthropogenic disturbances in warming waters.

Key words: blast fishing, coral, Coral Triangle, reef, rehabilitation, restoration

Implications for Practice

- Inexpensive modular structures make coral reef rehabilitation practical on large spatial scales where the reef's physical integrity has been destroyed by blast fishing, storms, or groundings.
- Structures that (1) allow unrestricted water flow, (2) trap broken coral fragments and rubble, and (3) stabilize the substratum effectively support high coral recruitment, growth, and diversity.
- Removal of territorial damselfishes that garden algae can prevent algal overgrowth on corals early in rehabilitation projects.
- Large-scale coral reef rehabilitation must supplement fisheries management and marine protected areas as a practice for sustaining reefs into the future.
- The center of the Coral Triangle in Indonesia is a priority for coral restoration and conservation because its exceptional biodiversity is under some of the most severe anthropogenic threats.

The authors dedicate this work to the memory of Dr. Susan Williams, a pioneer in marine ecology and a devoted colleague, mentor, and educator to scientists and students around the world.

Introduction

Coral reef restoration is the process of assisting coral reef ecosystems to recover from myriad disturbances to a state

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[1]Bodega Marine Laboratory and Department of Evolution and Ecology, University of California—Davis, PO Box 247, Bodega Bay, CA 94923-2047, U.S.A.
[2]Bodega Marine Laboratory and Graduate Group in Ecology, University of California—Davis, PO Box 247, Bodega Bay, California 94923-2047, U.S.A.
[3]Address correspondence to Jordan A. Hollarsmith, email jahollarsmith@ucdavis.edu; Christine Sur, email csur@ucdavis.edu
[4]Jl. Kima 10 Kav A-6 Daya, Biringkanay, Makassar, South Sulawesi 90241, Indonesia
[5]School of Biological Sciences, University of Wollongong, Wollongong, NSW 2522, Australia
[6]Department of Marine Science and Fisheries, Hasanuddin University, Makassar, South Sulawesi, Indonesia
[7]Mars, Inc., 6885 Elm St., McLean, VA 22101, U.S.A.
[†]Deceased 24 April, 2018.
[‡]Deceased 23 March, 2017.

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in which their native structure and function is self-sustaining (Suding 2011; McDonald et al. 2016). Although activities to assist reef recovery have long been focused on fisheries and marine protected area (MPA) management (McClanahan et al. 2006; Mumby & Steneck 2008; Aswani et al. 2015; Table S1), there is a dawning recognition that these measures need to be supplemented with other interventions, as reefs are unlikely to be able to return to former, less degraded states (Rinkevich 2008; Hughes et al. 2017; van Oppen et al. 2017). Thus while "restoration," the act of returning a degraded ecosystem to its predisturbance state, is often the goal in active intervention projects, what is most often achieved is "rehabilitation" in which desired ecosystem functions are restored but the ecosystem remains novel (Precht 2006; Edwards 2010).

Although coral reef ecological restoration is in its infancy, practitioners and scientists agree that active interventions in the form of partially or, more rarely, fully replacing structural or functional attributes of a reef are required. That is, the process of reef "rehabilitation" must supplement other activities to assist reef recovery (Precht 2006; Edwards & Gomez 2007; Edwards 2010). Rehabilitation acknowledges that full recovery ("restoration") of coral reefs is circumscribed by the severe anthropogenic threats assaulting them, which must be mitigated before full recovery is achievable. It is critical to recognize that no single management objective will be sufficient for coral reef ecological restoration. Ideally, a coral reef ecological restoration project will be defined by carefully planned activities, including habitat protection, fisheries management, and other management frameworks, that are implemented to assist an ecosystem to fully recover its condition prior to disturbances, as represented by a local native reference ecosystem (Edwards & Gomez 2007; McDonald et al. 2016). In practice, there are many reasons that impede full recovery, from lack of appropriate techniques, resources, or planning, to an environment so changed that even the most appropriate reference ecosystem has diverged from a previous state considered "undisturbed."

Many techniques have been developed to increase coral cover on a degraded reef: stimulating coral growth using electricity; deploying different substrata as a base for coral recruitment or the attachment of coral fragments; "gardening" corals by first establishing a nursery from recruits or fragments to supply later transplantations (Bowden-Kirby 2001; Borell et al. 2010; Lirman & Schopmeyer 2016). These techniques have been developed in many small projects but scaling them up to large projects has been limited (Fox & Caldwell 2006; Bayraktarov et al. 2016; Montoya-Maya et al. 2016). Rehabilitation and active restoration activities have also been questioned on the grounds of not being cost-effective when compared to the costs of enforcing fishing and MPA regulations or even when accounting for the high economic value of ecosystem services to be gained (Spurgeon 2001; Haisfield et al. 2010; Rinkevich 2017). Economic data, however, are limited.

Here we describe a large coral reef rehabilitation project in the center of the Coral Triangle where the coral diversity is the highest on earth (Sanciangco et al. 2013). This diversity is threatened by overexploitation of marine resources, destructive fishing practices, coastal development, pollution, disease, and climate change (Edinger et al. 1998; Bruno & Selig 2007; Burke et al. 2012). The project was conducted on a typical small island (Pulau Badi) within the Spermonde Archipelago in South Sulawesi, Indonesia. The archipelago supports the largest coral reef fishery in Indonesia (Pet-Soede & Erdmann 1998). Many reefs in the archipelago have experienced eutrophication and illegal blast fishing (Sawall et al. 2013; Plass-Johnson et al. 2015). Blast fishing continues unabated in the Coral Triangle (Praveena et al. 2012; Muallil et al. 2014; Glaser et al. 2015), obliterating the reef framework and leaving unstable coral rubble unsuitable for the survival of reef-building corals (Fox et al. 2003; Fox & Caldwell 2006). Blasted reefs are thus prime candidates for rehabilitation interventions that add stable structures to which coral can be transplanted and/or recruit (Edwards 2010; Rinkevich 2015), provided that blast fishing is reduced and the environment otherwise remains suitable (Fox 2004; Fox et al. 2005; Raymundo et al. 2007).

The objective of the project described here was to restore key functions of a reef by installing specially constructed structures onto which coral was transplanted. The project was initiated in 2013 by a private sector entity (Mars Symbioscience, a business segment of Mars, Inc.; http://www.mars.com/global) in collaboration with the islanders. It was assumed that increasing live coral cover on the reef would lead to improved fisheries resources and livelihoods, but stabilizing the damaged section of the reef and increasing coral cover is only one important piece in the full suite of management efforts required to improve fisheries and livelihoods. As part of the Coral Reef Rehabilitation and Management Project ("COREMAP"; World Bank 2005), the islanders had established a small no-take zone (*Daerah Perlindungan Laut*, hereafter "DPL") on a section of reef in 2007.

The rehabilitation project we studied was not designed as a scientific experiment and ecologists joined the project only after it was underway; however, it presented an opportunity to document a large active intervention to improve a reef's condition. Our study of the project aimed to (1) describe the technique used to stabilize the substratum and increase coral cover, (2) document the size and costs of the rehabilitation project, (3) census live coral cover on the structures and compare it to a reference area in the DPL, and (4) disseminate the successes, failures, and lessons learned (Edwards 2010).

Methods

Study Site

The project site was Pulau Badi ("Badi island" in Bahasa Indonesia) (4º58′3.02″S, 119º17′15.00″E, Fig. 1; Plass-Johnson et al. 2015). The rehabilitation occurred over 2 ha of reef that was primarily a rubble field interspersed with some remnant dead coral structures covered by invertebrates and algae and some smaller colonies of mound corals. According to islanders, this portion of the reef was damaged by the construction of a boat channel 30–40 years ago, blast fishing approximately 30 years ago, and massive coral mining to build houses and a breakwater 20 years ago. Storms further eroded the reef. The rehabilitation began close to a major jetty and

Figure 1. Map of Pulau Badi showing the rehabilitation site on Google Earth, including the four sections of the rehabilitation and the DPL (yellow border). Section 1: deployed March–June 2013; Section 2: deployed June 2014; Section 3: deployed July–November 2014; Section 4: deployed July–September 2015. The nursery area was established in 2015 using spiders to produce fragments for limited transplantations in the rehabilitation area.

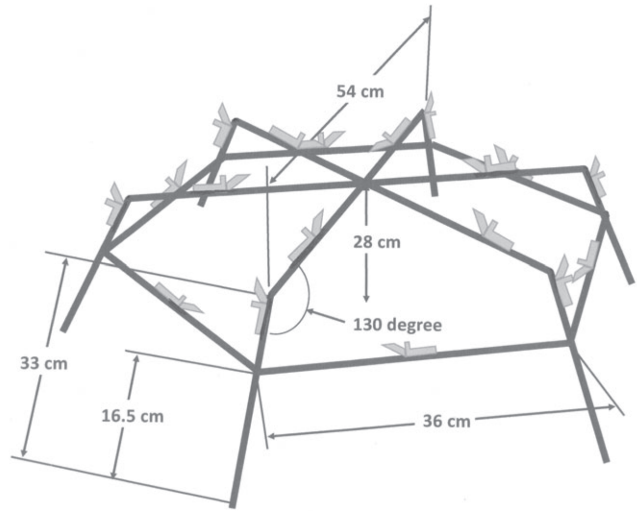

Figure 2. Diagram of a spider.

sections were added successively to wrap around the south end of the island and into the shallow DPL, which also had been damaged (Fig. 1). The rehabilitation area was continuous but we censused discrete sections of it corresponding to deployment dates starting with the first deployment (March–June 2013, Section 1), followed by Section 2 (deployed June 2014), Section 3 (deployed July–November 2014), and Section 4 (deployed July–September 2015). Sections 3 and 4 were in the shallow damaged area in the DPL. The largely undamaged area of the DPL abutting the rehabilitation site was considered the best reference reef for the project.

Coral Reef Rehabilitation Technique

Coral fragments were attached to standardized hexagonal-shaped structures, termed "spiders" (Fig. 2). The spiders were made of reinforcing steel rod ("rebar"), rust protected (R-10, Industrial Chemical Technologies, Inc.) and then dipped in fiberglass resin and coated with coarse beach sand to provide a rough substratum for coral recruitment. The perimeter of a spider's hexagon was 216 cm, enclosing an area of 0.337 m^2, which we surveyed (see below). The height of the middle of the spider was 28 cm above the plane formed by the perimeter of the hexagon, which was elevated by six "legs" (16.5 cm long).

Approximately 11,000 spiders were deployed over an estimated area of 7,000 m^2 within the site between March 2013 and September 2015 (Fig. 1). During each deployment a team of four trained divers, 36 trained islanders, and two boat hands collected loose coral fragments ("corals of opportunity") from the back reef and shallow reef at the rehabilitation site where they cable-tied 18 evenly spaced fragments of roughly 15 cm length to each spider. No attempt was made to control species diversity. Fragments were mostly *Acropora* spp., although later deployments included more genera. Later, Pulau Badi fragments were augmented by loose fragments collected from a submerged reef approximately 4.5 km away (4º56′37.00″S, 119º15′23.00″E) and also by limited numbers broken off from a nursery established in 2015 using natural fragments or those on damaged spiders (Fig. 1). Immediately after coral attachment, the spiders were continuously sprayed with seawater as they were transported in small boats to the deployment site on the reef (20 minutes maximum). There, divers situated spiders to fill small gaps in the existing damaged reef or linked spiders together with cable ties in a web to cover large areas. The spider legs were anchored in the predominately rubble substratum and the web was secured with stainless steel rods at points deemed vulnerable to dislodgement. On average, the team was able to deploy 550 spiders in areas 300–400 m^2 over a 3-day period. The start and end points of each deployment were marked, the number of spiders deployed and the area of rehabilitated reef were estimated, and the costs of the rehabilitation efforts were itemized.

Live Coral Cover on the Spiders and in the DPL Reference Area

We visually surveyed the percentage of live coral cover (Jokiel et al. 2015) to the nearest 1% on individual spiders in the four sections of the project. We surveyed coral cover twice (February to March, July to September) in each year of the study beginning in 2014, with a final census in October 2017. Starting at a haphazardly selected spider at one end of each section, we surveyed 20 randomly selected spiders along each of three unmarked transects following reef contours: the section's shallow edge ("shallow," approximately 1.5 m water depth), midway between the shallow and deep edges ("middle," approximately 3 m), and along the deep edge ("deep," 3.6–4.0 m). All of Section 4 was deployed at 2 m depth or less and so we conducted transects on the shoreward and seaward edges and along the middle. The sample size varied when the section was smaller than

100 spiders (e.g. along the deep edge of Section 1), a spider was inadvertently added or missed, or more observers were available. The observer estimated the percent live coral cover within the plane formed by the perimeter of the hexagon by hovering approximately 1 m away from, and perpendicular to, that plane. The cover estimates were conservative in that they included only the planar coral cover and not growth vertically or beyond the perimeter of the spider. Altogether, there were 12 observers, one of whom surveyed every time and four who surveyed multiple times. All were trained similarly and the mean coefficient of variation between estimates by paired observers was 21% ± 27% SD ($n = 102$ spiders). Damaged spiders were periodically replaced, but they were unmarked and thus cover estimates included replacement and natural and other human disturbances in the rehabilitation site.

Starting in August 2015 we also surveyed the live coral cover in the largely undamaged area in the DPL adjacent to the rehabilitated area, as the closest approximation of a reference site (Fig. 1). We used a lightweight wire hexagonal frame of the same dimensions as the spiders and surveyed along the shallow edge (approximately 2 m depth), the middle (approximately 6 m), and deep edge (approximately 14 m) contours by placing the frame at randomly selected positions ($n = 20$) along each depth contour starting at haphazardly chosen points.

Observers visually estimated the animal-stressed (sensu Fitt et al. 2001) bleached coral cover on the spiders and in the DPL reference site from August 2015 onward. In October 2017 we also surveyed natural recruitment by counting the number of small colonies not obviously tied onto randomly selected spiders in Sections 1 and 2. We counted only nonacroporid colonies to increase the certainty that they had not been transplanted, making the recruit count conservative. Recruits could not be differentiated in Sections 3 and 4 due to high coral cover. A nonsystematic survey of coral species on the spiders and growing naturally among them was made at this time.

In August 2014, we installed one Hobo Pendant Temperature/Light logger (UA-002-64, Onset Computer Corp, Bourne, MA, U.S.A.) in Section 2 at 0.9 m depth (4°58′18.54″S; 119°17′10.77″E). Two more were installed in the DPL at 3 m depth: the first (DPL1) was 5 m offshore from Section 4 (4°58′21.65″S; 119°17′3.13″E) and the second (DPL2) was centered in the DPL at 180 m north of DPL1 (4°58′17.78″S; 119°16′59.89″E). In July 2016 a fourth logger was installed in Section 1 at 3 m (4°58′14.78″S; 119°17′14.58″E). Data were recorded every hour and downloaded within 3–5 months.

Although the rehabilitation project was not an experiment, we tested whether live coral cover in the sections differed from each other, differed across depth, and differed from the DPL reference site. Cover data were bound between 0 and 1 using the data transformation in Smithson and Verkuilen (2006), then analyzed using generalized linear mixed models with a beta distribution and logit transformation in the glmmTMB package in R (Brooks et al. 2017; R Core Team 2017). Site (Sections 1–4, DPL) was a fixed effect; depth and census date were random effects. The bleached coral cover was similarly analyzed.

Estimation of Live Coral Cover on the Reef Before and After Spider Deployment

We estimated the existing cover of live coral in the rehabilitation site before and after spider deployment in the following ways. In Section 4, the existing live coral cover was estimated immediately before and immediately after spider deployment and 1 year later in quadrats (1 m^2, $n = 17–40$ randomly placed within a haphazardly selected 10×10 m area). At the end of the study (October 2017) we used transect lines to demarcate an area 60 m long parallel to shore and 13 m from the shallow to the deep edge of Section 1. A 60×13 m area was similarly demarcated immediately adjacent to, but inshore of, Section 2. An additional 21 lines were laid every 3 m perpendicular to the 60 m line to identify the substratum type (sand, rubble, live natural coral, live coral growing on spiders, and various other types—primarily other invertebrates and crustose coralline algae) under each transect. The length covered by each substratum type was summed across all transects to estimate the percent of each type.

The area inshore of Section 2 previously had been censused in July 2015 using the same quadrat method described above just prior to spider deployment to increase the size of Section 2. However, the coral did not do well in the inshore area for unknown reasons and thereafter the spiders were removed. The inshore area did not recover naturally and thus it was a proxy for the condition of the adjacent Section 2 prior to spider deployment.

Results

Rehabilitation Costs

The total cost of installing 11,000 spiders was US$174,000 (Table 1). Costs included materials, construction labor, transporting spiders by truck to a jetty and by boat to the deployment site, and coral attachment and installation labor. When volunteer divers participated, the installation costs were reduced. Maintenance costs, if required to remove algae-gardening damselfishes and to replace dead fragments during the first 3 months, were estimated at an additional US$3 per spider.

Live Coral Cover in the Rehabilitation and the DPL Reference Area

Section 1 was the oldest deployment in the rehabilitation project. The mean live coral cover attained on the spiders over all survey dates along its middle depth contour was 38% (4% SE, $n = 8$ surveys, Fig. 3). At times the mean cover was greater than 50% and individual spiders became completely covered by live coral. Cover decreased with depth in Section 1, as it did in the DPL and across all rehabilitation sections (Fig. 4, Table S2). Although we have no data on the live coral existing before deployment in Section 1, by the end of the study (October 2017), 21% of the substratum was live coral growing on spiders, 41% was live coral growing naturally, and rubble, sand, rock, and coralline algae composed the remaining substratum types.

Table 1. Costs of the Pulau Badi coral reef rehabilitation for constructing and installing 200 spiders based on 2015 values.

	$US/unit	*Unit*	*No. spiders/ unit*	*$US/ spider*
Materials				
Steel rods	3.78	12 m	2	1.89
Resin coating	2.89	1 kg	1.8	1.63
Tools	0.03			0.03
Rust preventer	23.33	L	80	0.29
Cable ties	0.04	1	0.03	1.41
Labor				
Welding	2.59	1	1	2.59
Coating	1.19	1	1	1.19
Attaching corals	1.33	1	1	1.33
Transport from construction site				
Truck	107.41	1	400	0.27
Boat	133.33	1	100	1.33
Installation				
Divers	37.04	day	50	0.74
Boat and SCUBA tanks	618.52	2	200	3.09
Total				15.76

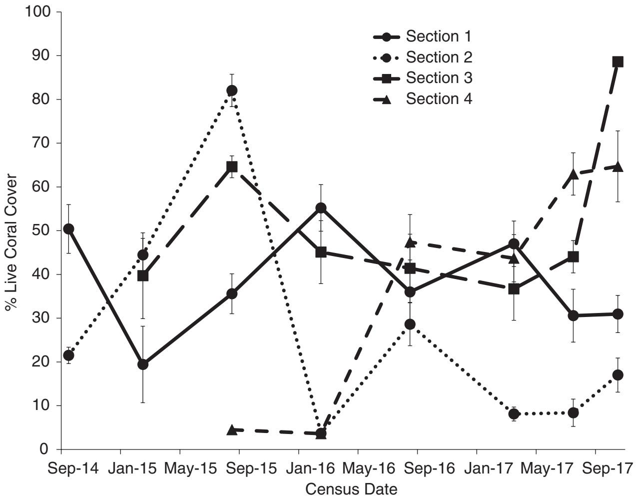

Figure 3. Changes in live coral cover on spiders along the middle depth of Sections 1–3 and the middle of Section 4 (spiders deployed only at a shallow depth). Mean ± SE ($n = 19–53$ spiders, but mostly 20). See Figure 1 for deployment dates of the sections.

The live coral cover on spiders deployed after 2013 (Sections 2, 3, and 4) attained at least 40% within the first year (Fig. 3, Fig. S1, Video S1). The initial mean cover on the spiders (5% ± 0.1% SE, $n = 20$) was estimated in Section 4, where the deployment and census dates coincided. By the study's end live coral covered spiders so completely that it was hard to see them in Section 3. The live coral cover on the spiders in all sections was lower compared to in the DPL reference area where it was >50% (Figs. 3 & 4, Table S2).

The trajectories in live coral cover differed among the four sections (Fig. 3, Table S2), which is not surprising given their different deployment dates, locations, and disturbances. For

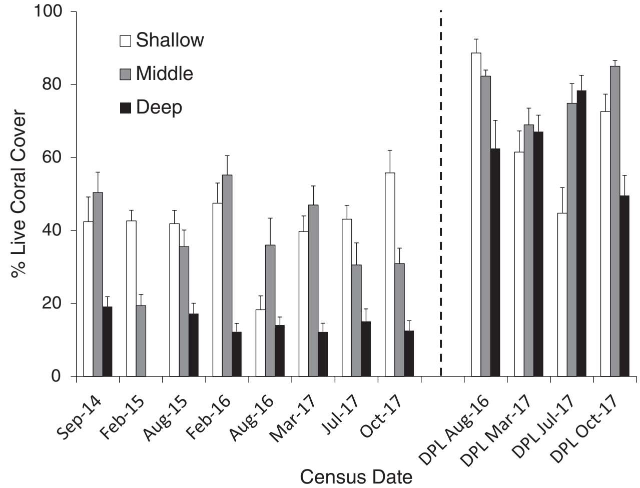

Figure 4. Mean live coral cover on spiders in Section 1 (the oldest section deployed in March–June 2013), surveyed along the shallow edge (approximately 1.5 m water depth), the middle (approximately 2 m water depth), and the deep edge (up to 4 m water depth) of the section and for comparison in the no-take DPL reference site on Pulau Badi (deep edge = 14 m). In September 2014, Section 1 was at least 25 months old. Mean + SE ($n = 19–53$, but mostly 20).

example, the mean live coral cover in Section 1 dropped from 50% in August 2014 (approximately 20 months postdeployment) to 19% in February 2015. Although we cannot definitively ascribe the reason for this decline, villagers watching over the rehabilitation area observed cyanide fishermen interfering with the rehabilitation area between November and December 2014. After their illegal fishing for ornamental fishes was reported to the authorities, the fishermen retaliated by ripping out and piling up spiders. Yet, this section showed resilience. The mean cover on the remaining and approximately 400 repaired spiders increased to 55% in February 2016 (approximately 14 months postvandalization).

In Section 2, the mean live coral cover on the spiders along the middle transect initially increased quickly and reached 82% (4% SE, $n = 19$) within 14 months (August 2015). Then, in September–October 2015, the coral was severely affected by an unknown event, leading to less than 5% live coral cover in the following survey in February 2016 (Fig. 3) when mean dead coral cover on the spiders was 48% (6% SE, $n = 20$). Previously, the mean dead coral cover was 9% (2% SE, $n = 24$). Although some natural recovery occurred (29% live coral cover ±5% SE, $n = 19$, in August 2016), this section was deemed problematic and thereafter many spiders were removed or added, which altogether probably accounted for low cover.

In February 2017 a large storm damaged Section 4 and the DPL and lower cover was evident in the March 2017 census (Figs. 3 & 4). Poststorm, the cover included the surviving coral attached to the spiders as well as alive fragments that the spiders had trapped. Section 4 showed resilience: by July 2017, the live coral cover on spiders increased to roughly prestorm levels (Fig. 3).

One important result was that in the first weeks after deployment, it was critical to remove aggressive algae-farming damselfishes to prevent algal overgrowth on the spiders (Forrester et al. 2011). After roughly 3 months of removing the large white damselfish (*Dischistodus perspicillatus*) and Cross's damselfish (*Neoglyphididon crossi*), the coral on most of the spiders escaped algal overgrowth.

Estimates of Live Coral Cover on the Reef Before and After Spider Deployment

The following results are estimates of the total live coral cover in the rehabilitation site before and after the spider intervention. The existing natural live coral cover was low (7% ± 2% SE, $n = 37$ quadrats) in Section 4 before spider deployment. Another 8% (1% SE, $n = 17$) live coral cover was added when the spiders were deployed. The total live coral covering the substratum 1 year after deployment was 48% (4% SE, $n = 40$), of which 25% (3% SE, $n = 40$) was live coral on spiders. The increase from the initial low cover represented natural recruitment and growth and also growth of coral on and extending off the spiders.

Another estimate of the predeployment coral cover in the damaged reef came from the area immediately inshore of Section 2. Prior to spider deployment in this inshore area, live coral cover was 6% (2% SE, $n = 20$ quadrats). Immediately after spider deployment the cover was 14% (2% SE, $n = 20$). However, these spiders were the ones removed after the area was deemed problematic and coral did not recover naturally, as evident from the transect line survey in October 2017 when the live coral cover was 3% and the substratum was primarily rubble (92%). This estimate is a reasonable proxy for the original damaged condition of the rehabilitation site.

Coral Recruits and Species in the Rehabilitation

Natural recruitment on the spiders was estimated to be one nonacroporid recruit per spider by the end of the study in October 2017 (Table S3, Fig. S2). At this time there were at least 42 coral species on the spiders (Table S4) and at least 58 species growing nearby or among spiders within the various sections of the rehabilitation site (Table S4). Of the species that have been assessed by the World Conservation Union (2017), there were 29 species of least conservation concern, 10 vulnerable species, and 13 near-threatened species. Their population trends were reported as either unknown or decreasing (Table S4). Although our observations were not systematic, this result indicated that the rehabilitation supported a diverse coral assemblage.

Bleaching

Surprisingly, minimal coral bleaching occurred within the rehabilitation area (Fig. 5). Bleaching along the middle transects of all sections never exceeded a mean of 5% of the spider area when surveyed, despite high temperatures (Fig. 5). Temperatures during the monitoring period exhibited annual cycles with the coolest period in August during the dry season

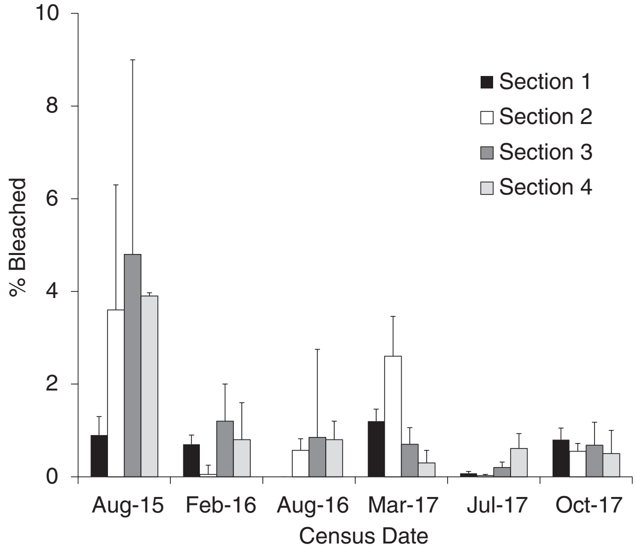

Figure 5. Coral bleaching as mean percent of spider area (+SE) along the middle depth of Sections 1–4 of the rehabilitation and the DPL reference site ($n = 19–20$ spiders). See Figure 1 for deployment dates of sections. The middle transect of Section 4 was less than 2 m depth.

(Fig. 6). The mean temperature in the DPL reference site was 29.5°C (1.2 SD). Temperatures at the rehabilitation and reference sites started exceeding 30°C in July 2016. Temperatures at the shallow rehabilitation site in Section 2 exceeded 32°C and reached >37°C into early 2017. Temperatures in Section 1 were recorded only beginning in 2016 (mean 30.2°C ± 0.7 SD), but they tracked the DPL temperatures. Although our bleaching data were most complete for the middle transect (approximately 3 m depth), little bleaching occurred in Section 4 (≤2 m depth), along the shallow edges of other sections, or in the DPL reference site, and there were no differences among sections, depths across sections, or sections versus the references site (Table S2). Over the study, we observed small (approximately 1–2 m^2) bleached, but not obviously diseased, areas of coral that possibly resulted from cyanide. Occasionally, we found small areas (<0.5–1 m^2) of coral killed by *Acanthaster plancii* (crown-of-thorns seastar) or the corallivorous gastropod *Drupella* sp.

Discussion

Scalability has been a major limitation of coral reef restoration, particularly where an unsuitable substratum for coral growth first requires physical rehabilitation through deployment of structures, followed by natural recruitment of coral or coral transplantation, which render coral restoration expensive compared to other marine ecosystems (Edwards & Gomez 2007; Edwards 2010; Feliciano et al. 2018). The project's objective was to rehabilitate a large section of a reef damaged to the point at which it was an unstable rubble field with minimal live coral cover. To achieve this objective, the structures used to stabilize the substratum needed to support growth of transplanted fragments and be scalable in terms of deployment ease and expense.

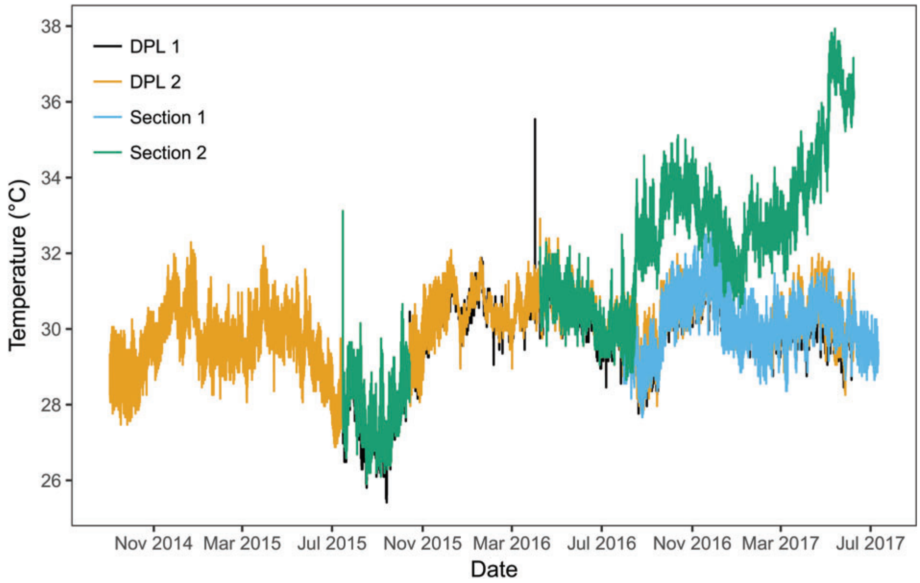

Figure 6. Water temperatures (°C) in DPL1 and DPL2 sites and Sections 1 and 2.

Our study's aim was necessarily limited to documenting the rehabilitation technique, its size and costs, and to monitoring coral cover on the structures (spiders) and comparing it to the nearby, relatively undamaged reference no-take reef. Although the project was initiated in the hope that local fishing would improve if the reef could be rebuilt, there was no formal restoration plan, which would have included quantitative baseline surveys of the reef, establishment of reference sites, systematic monitoring of the project, and mitigation of stressors, as recommended for ecological restoration (McDonald et al. 2016). Monitoring fisheries species and livelihoods was beyond the scope of our study, which focused on coral cover, and the rehabilitated reef was primarily outside the no-take zone on the reef.

Nevertheless, a 2 ha section of damaged reef was rehabilitated by adding 0.7 ha of spiders, which supported a rapid increase in live coral cover at a relatively low economic cost ($24.85/m^2), demonstrating that large-scale rehabilitation of coral reefs is achievable. Costs reported from comparable rehabilitation projects that required substrate stabilization range from $35–277/m^2 (Clark & Edwards 1999) to $50–200/m^2 (Edwards 2010), not accounting for inflation. In addition to stabilizing the substratum, the spider structure is conducive to coral growth because its minimal structure reduces surface area and drag and allows relatively unimpeded water flow, which is essential for coral metabolism (Nakamura & van Woesik 2001; West & Salm 2003), promotes better mixing of coral spawn, and supplies essential planktonic food for heterotrophic feeding (Borell et al. 2008). The spiders also intercept loose fragments, as occurred in Section 4 during a big storm, allowing them to regrow. Although the spiders add artificial material to the reef, their coatings prevent rust for at least 5 years and they quickly become coated with coral and crustose coralline algae. Coral recruited to the spiders and a diverse assemblage of coral species, including ones of conservation concern, grew on the spiders by the end of the study. Overall, live coral cover within the rehabilitation site (on and off spiders) was estimated to increase by >40% since 2013, although it was lower than in the reference site where live coral cover is exceptionally high despite lack of sustained management. Although we have no quantitative data on herbivore abundances, reef fishes seemed more diverse and abundant on the rehabilitation over time, so much so that it attracted ornamental species collectors. Herbivorous rabbitfishes, parrotfishes, and surgeonfishes rapidly colonized Section 4, which was built closest to the reference site and adjoining the sections deployed earlier. By focusing on scalability, large-scale coral reef rehabilitation is feasible, a conclusion also reached in a large research project (0.52 ha) in a no-take reserve in the Seychelles (Montoya-Maya et al. 2016).

Furthermore, despite the disturbances, the estimates of live coral cover on spiders and on the reef substratum in the rehabilitation area are higher than reported for locales within the Spermonde Archipelago and elsewhere in the Coral Triangle, including MPAs (Crabbe & Smith 2002; Selig & Bruno 2010; Polónia et al. 2015; Ponti et al. 2016; Teichberg et al. 2018). The estimates are also higher than coral cover reported for islands in the central Pacific Ocean (including uninhabited ones; Smith et al. 2016) and elsewhere throughout the tropics (Graham & Nash 2013).

It will be important to validate the spider technology by rigorous replication in different locations. Replication of the project on other islands in the Spermonde Archipelago was not feasible at the time, as is typical in ecological restoration (see Montoya-Maya et al. 2016). However, similar modular frame methodology has been used in the Maldives to rehabilitate nearshore reefs; this project resulted in 1,250 m^2 of live coral cover, which also initially began with primarily acroporid fragments, in addition to pocilloporids, with 90% survival (Edwards 2010). In addition to being an unreplicated project that was not designed as an experiment, monitoring was delayed, logistical and resource limitations occurred, and personnel turnover was inevitable in the long-term project. For example, predeployment coral cover could only be approximated for the most part after rehabilitation had commenced. The principles of ecological restoration (McDonald et al. 2016) are essential to incorporate in future projects throughout the Coral Triangle (Hein et al. 2017). Based on the Badi example, in July 2017, another rehabilitation was begun on a neighboring island (Bontosua), after a year of baseline ecological and socio-anthropological studies, followed by monitoring of the coral cover and fishes in replicated reference, rehabilitated, and unrehabilitated damaged sites and continuing anthropological study as the project is proceeding.

Ultimately, for these rehabilitation projects and ecological restoration in general to succeed, ongoing human disturbances must be managed better, in addition to concerted systematic planning and adherence to best practices by practitioners. Throughout much of Indonesia and certainly the Spermonde Archipelago, humans have few alternatives to fishing livelihoods and basic education about the marine environment is often lacking (Glaser et al. 2015). This situation reinforces the importance of socioecological approaches in this region (Hein et al. 2017). Community awareness and appreciation for the rehabilitation and coral reefs in general has been improving in response to visits by government agency staff, nongovernment organizations, elected officials, and other researchers. The Pulau Badi project also provides an example of a collaboration of islanders, the private sector, and scientists. Private sector involvement in coral restoration tends to be within the context of funding projects, which is a continuing issue if they are to be scaled up (Bottema & Bush 2012; Chabbi et al. 2017), but such partnerships can also bring together stakeholders.

This project demonstrates that, although the rehabilitation did not achieve the same degree of live coral cover as in the DPL, rehabilitation did affect a transition from a heavily damaged reef and rubble fields to good coral cover under less than ideal conditions. A similar result was found for rehabilitations of damaged reefs in the Philippines (Gomez et al. 2014). However, serious issues remain and they must be assiduously managed for ecological restoration to advance (Burke et al. 2012; Spalding & Brown 2015). Illegal fishing still occurs in the Spermonde Archipelago as it does throughout the Coral Triangle. Due to the lack of island sanitation systems, eutrophication also is happening in the archipelago (Sawall et al. 2013; Plass-Johnson et al. 2015). Seagrass is under threat (Williams et al. 2017) and marine debris abounds (Sur et al. 2018). These issues, which are common to tropical marine environments, are essential to manage in order to give coral reefs the best chance of surviving through a warming and acidifying ocean.

It is hopeful that the coral in the Pulau Badi rehabilitation and reference sites showed surprising resilience under combined stressors, including thermal. During the 2014–2016 El Niño–Southern Oscillation (ENSO) event when massive bleaching occurred on the Great Barrier Reef (Hughes et al. 2017), only "mild" (sensu Oliver et al. 2009) bleaching occurred at the rehabilitation and reference sites despite water temperatures that exceeded the 30°C threshold for coral heat stress (Hoegh-Guldberg 1999; Kleypas et al. 1999; NOAA 2017). The recorded temperatures also exceeded the long-term (1985–2006) mean sea surface temperature (SST) of 29°C (Peñaflor et al. 2009). We observed nothing that indicated bleaching due to lowered sea level also associated with this ENSO event, unlike reefs in north Sulawesi (Eghbert et al. 2017). Temperature monitoring has been limited in marine environments in the Spermonde Archipelago but it should be instituted to better predict coral stress and areas most suitable for restoration. The Pulau Badi corals fell outside the general predictions for thermally induced bleaching, which reinforces that there is much to learn about local variation and how to prioritize restoration sites and reduce stressors more amenable to management in the short term than climate change (van Hooidonk et al. 2016).

Eventually, coral restoration and rehabilitation projects will rely on coral nurseries to provide seed materials, guided by accumulating knowledge on the importance of species diversity, the genetic structure of coral populations, and local adaptation of corals (Baums 2008; Shearer et al. 2009; van Oppen et al. 2015). With such knowledge comes the potential creation of "designer" rehabilitations made of stress-tolerant corals. In the meantime, the spider technique outlined here offers the ability to rehabilitate large tracts efficiently, increase coral cover and diversity, and enhance the capacity of corals to acclimate or adapt to a worsening ocean climate while increasing habitat for dependent organisms. Rehabilitation belongs alongside fisheries management, MPAs, and reduction of pollution (Lamb et al. 2018) as objectives of coral reef restoration, which also depends on the conservation and restoration of interdependent seagrass and mangrove systems in the coastal zone (Williams et al. 2017).

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Supporting Information

The following information may be found in the online version of this article:

Table S1. The extended bibliography provides other relevant papers.
Table S2. Results from generalized linear mixed models for differences in % live coral cover.
Table S3. Natural coral recruitment on spiders, October 2017.
Table S4. Coral species transplanted on or recruited to spiders.
Figure S1. Time series photos of part of Section 4 in the Pulau Badi rehabilitation project.
Figure S2. Naturally recruited coral on a spider deployed in 2013.
Video S1. Coral rehabilitation.

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