LETTER

An ounce of prevention: cost-effectiveness of coral reef rehabilitation relative to enforcement

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Abstract

Blast fishing has destroyed many coral reefs in Southeast Asia by creating large fields of dead coral rubble where new coral recruits settle but cannot survive and grow. Possible management responses include reef rehabilitation of damaged areas, and/or increased enforcement to protect still-living ones. Here we show that in Komodo National Park, Indonesia, rehabilitation by installing locally-quarried rocks on blasted rubble fields can be relatively low cost (~US\$4.80 per m^2) and simple, but it is not economically viable at large scales. Although rehabilitation without enforcement is unlikely to be effective, we compared rehabilitation data (costs and coral growth over 8 years) and enforcement costs to conduct two economic analyses: cost-per-area calculations and a cost-effectiveness model over 7 years, and found that rehabilitation costs ~70 and ~5-times more, respectively, than marine patrols to enforce blast fishing bans. Hence, we recommend that marine protected area managers prioritize investment in achieving compliance with regulations above investment in rehabilitation to ensure that reefs continue to generate biodiversity and fisheries benefits and tourist revenues.

Introduction

Coral reefs, among the most diverse ecosystems on the planet, are in accelerating global decline, threatened by pollution, overfishing and destructive fishing, disease, and climate change (Bryant *et al.* 1998; Hughes *et al.* 2003; Birkeland 2004). Threats are pervasive and severe; ~35% of reefs are threatened and 19% are already nonfunctional worldwide, with estimates even higher in Southeast Asia (45% and 40%, respectively) (Wilkinson 2008).

Conservationists worldwide are turning to restoration and rehabilitation as options to protect, maintain, and enhance biodiversity in threatened ecosystems (Dobson *et al.* 1997; Roberts *et al.* 2009). Coral reef restoration tests scientific understanding of reef ecosystems (Precht 1998), and could help minimize the damaging effects of ship groundings, bleaching events, and destructive fishing (Normile 2009); however, few methods are economically feasible at broad spatial scales or for developing nations (Edwards & Clark 1998) because costs can range from US\$13,000 to \$100 million/ha (Spurgeon & Lindahl 2000). Given that the opportunity cost of public funds is high in poor countries, funds for management are limited. Most managers report that marine protected areas (MPAs) are under-funded (Balmford *et al.* 2004), so managers face a difficult trade-off between investment in compliance through enforcement, education, and other initiatives and investment in rehabilitation of damaged areas. Economic analyses can provide useful information when making these decisions, but reviews suggest that economic studies of MPA effectiveness are limited (Pelletier *et al.* 2005).

Management decisions are difficult in the case of dynamite or "blast" fishing, an illegal practice in which homemade bombs are detonated into schools of fish, because chronic blast fishing results in large areas of broken coral rubble that are unlikely to recover naturally (Fox & Caldwell 2006). Blast fishing has severe biodiversity and

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coral reef habitat impacts because it kills target and non-target organisms and shatters coral skeletons (Alcala & Gomez 1987; Pauly *et al.* 1989; Burke *et al.* 2002). Blast fishing has been frequently practiced in Southeast Asia since WWII (Galvez *et al.* 1989; Djohani 1995).

Numerous methods exist for potential reef rehabilitation or restoration, including Reef Balls™, concrete structures, coral transplantation, and electric fields to encourage more rapid coral growth (Rinkevich 2005). Relatively simple, low-cost rehabilitation methods have shown initially promising results in restoring structural foundations and facilitating new coral growth (Fox *et al.* 2003, 2005; Raymundo *et al.* 2007), but the long-term cost-effectiveness of these measures will depend on the ecological process, scale, time frame, and potential for recovery of blasted reefs. Furthermore, a 10-year-old rehabilitated reef is unlikely to have the same biodiversity or aesthetic appeal as a centuries-old pristine reef.

Coral reefs within MPAs are often protected via enforcement patrols, which can incur high costs for protected area managers (Alder 1996). Compliance with laws and regulations can be enhanced without coercion through legitimacy or self-interest (e.g., education, alternative livelihoods, and community partnerships; Honneland 1999; A. Ramoz-Álvarez and M. Mascia unpublished work/pers. comm.). However, bomb fishermen specifically are unlikely to switch to less destructive methods without strong enforcement measures (Pet-Soede *et al.* 1999), perhaps attributable to higher individual profits associated with blast fishing than legal fishing methods (Ruitenbeek & Cartier 2001). Although private benefits to blast fishers can be high, the costs to society due to loss of sustainable fishery revenues, tourism, and coastal protection from this destructive practice in coral reef ecosystems have been estimated at 6.5–50 times greater (Cesar *et al.* 1997).

In this study, we compared and predicted the cost-effectiveness (as measured by cost/m^2 of coral cover) of maintaining coral cover through enforcement of existing fisheries laws to that of promoting coral growth by stabilizing blasted reef substrate with locally quarried rocks. Based on financial data for enforcement and rehabilitation in Komodo National Park, Indonesia, as well as coral growth on rehabilitation treatments, we modeled 7 years of future coral growth and management costs.

Methods

Study site

Komodo National Park has high marine biodiversity, with over 200 species of reef-building corals (Best & Boekschoten 1988). The area of the park is 1,817 km^2, with 602 km^2 terrestrial and 1,214 km^2 marine habitat, including mangroves, seamounts, seagrass beds, and an estimated 17 km^2 of coral reef habitat (Mous *et al.* 2000), about half of which was estimated to be damaged by blast fishing by the mid 1990s (Holthus 1995). Following a request by the government of Indonesia, The Nature Conservancy began assisting with management of the Park in 1995; this assistance transitioned to the joint venture PT Putri Naga Komodo in 2006.

Rehabilitation

The Nature Conservancy was interested in the feasibility of rehabilitation to enhance coral recovery within the Park. In 2000, reef rehabilitation treatments of locally quarried rocks (~3–5,000 cm^3 each) were installed in piles ~1 m^3 in size distributed throughout plots of ~100 m^2 in nine rubble fields created by chronic blast fishing in 5–10 m depth of water in Komodo National Park. We periodically surveyed six quadrats (1 × 1 m) to obtain coverage (cm^2/m^2) of hard and soft coral for each treatment and for adjacent rubble areas at each site (additional methods in Appendix S1). In 2002, based on initial results, rehabilitation efforts were scaled up with installation of four different designs totaling ~1,500 m^2 in four of the nine sites (Fox *et al.* 2005, Appendix S2). We monitored coral growth as before and measured encroachment by rubble (Figure 1 and Appendix S1).

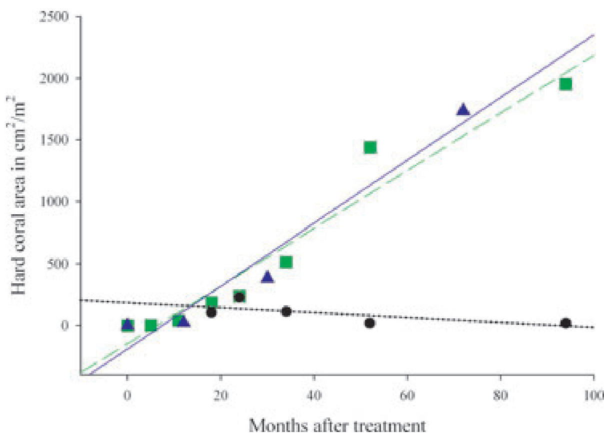

Figure 1 Hard coral growth on untreated rubble (showing no natural recovery; circles; $n = 54$–60^* ($P = 0.1766$, $R^2 = 0.51$, slope $= -2.02$)), and on rehabilitation treatments (squares: 100 m^2 study, $n = 54$–56^* ($P = 0.0001$, $R^2 = 0.94$, slope $= 23.31$); triangles: 1,500 m^2 study, $n = 24$ at 12 months (only one treatment was sampled at each site during the 12 month survey) and 95–100* thereafter ($P = 0.0224$, $R^2 = 0.85$, slope $= 25.41$)). Regression lines correspond to the symbols of the same color. *Variation in sample size because occasionally a seventh quadrat was surveyed.

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Expenses were tracked throughout the rehabilitation project for calculation of the cost per total treatment area of each rock pile design (Appendix S1 and Appendix S2).

Enforcement

Regular enforcement patrols commenced in 1996, staffed by park rangers and police, with assistance from The Nature Conservancy. Since that time blast fishing has decreased by 80% to 100% (Pet 1997; Mangubhai 2008), in part due to several early high-profile encounters between patrols and bomb fishers. Currently, Komodo National Park enforcement includes two boats that serve as "floating ranger stations" staffed by park rangers, local police, and local nongovernment organization (Putri Naga Komodo) staff that patrol the park year round with 50–70 ten-day trips per year (Mangubhai 2008). The floating ranger stations are supported by speed boats that can pursue blast fishers. Dive operators and land-based park rangers also report blast fishing to the ranger stations and speed boats. Although alternative livelihood and educational outreach programs were implemented around the same time as enforcement patrols (and are likely important for ongoing success), resource monitoring data show that the decrease in blast fishing directly coincides with the start of regular patrolling efforts by park staff and police, not the community programs (Mous *et al.* 2005).

Enforcement-related expenses were calculated by summing costs of operating floating ranger stations, speed boat support, fuel, maintenance costs, Park staff salaries, food costs, and stipends for the patrolling rangers. This base value was increased each year by 5% to account for inflation when calculating future expenses (S. Mangubhai, unpublished data). Itemized figures from the Park's workplan and budgets were used, but only aggregated costs are reported here because staff salaries and ranger stipends are not publicly released.

Economic analyses

We used quantitative data on coral cover and growth, and tracked expenses of enforcement and rehabilitation, to conduct two economic analyses: cost-per-area calculations and a cost-effectiveness model. An additional willingness-to-visit thought experiment was also explored (Appendix S3).

Cost-per-area calculations

We determined the cost/m^2 for rehabilitation using the total treatment area (including rubble between rock pile designs). For enforcement, the cost/m^2 was determined using the 17 km^2 area of coral reef habitat where patrols are concentrated, and does not include pelagic areas of the Park.

Cost-effectiveness model

Our cost-effectiveness model compared the amount of predicted coral coverage gained from investment in rehabilitation to the coral damage averted by investment in enforcement, where enforcement protects areas that would otherwise have been blasted (Appendix S1). We only considered the costs of each method, not the value of the coral gained, as would be required for full cost-benefit accounting.

Results

Rehabilitation

In general, live coral coverage increased on the rock treatments, which stabilize the substrate, whereas no natural recovery occurred in rubble plots (Figure 1). Both 100 m^2 treatments and 1,500 m^2 treatments were equally successful at supporting hard coral growth (Figure 1), with as little as 8% and as much as 43% hard coral coverage/m^2 6 years after installation. At 100 m^2 sites, hard coral cover ($\pm$ SEM) averaged 19.5% ($\pm$ 9.1%) and soft coral 28.2% ($\pm$ 21.4%) after 8 years. At 1,500 m^2 sites, hard coral averaged 17.4% ($\pm$ 9.6%) and soft coral 19.2% ($\pm$ 23.2%) with some *Acropora* spp. colonies $\sim$1 m in diameter after 6 years. There was evidence that coral growth varied with site location (Figure 2 and Kruskal–Wallis test, $\chi^2 = 12.53$, df = 3, $P = 0.006$) but not with treatment type or with depth (Appendix S2), although the highest

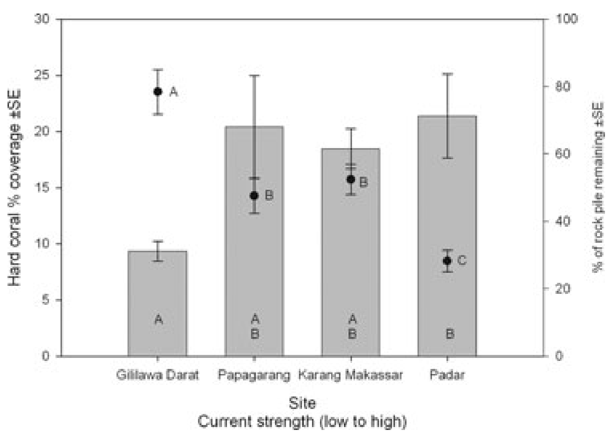

Figure 2 Hard coral coverage per m^2 (bars) and rubble encroachment (points). There was no evidence that sites sharing letters differed (Tukey HSD $\alpha = 0.05$) in coral coverage (letters at base of bars) or percent of treatment remaining (letters next to points). See Tables S1A and S1B for specific post-hoc results.

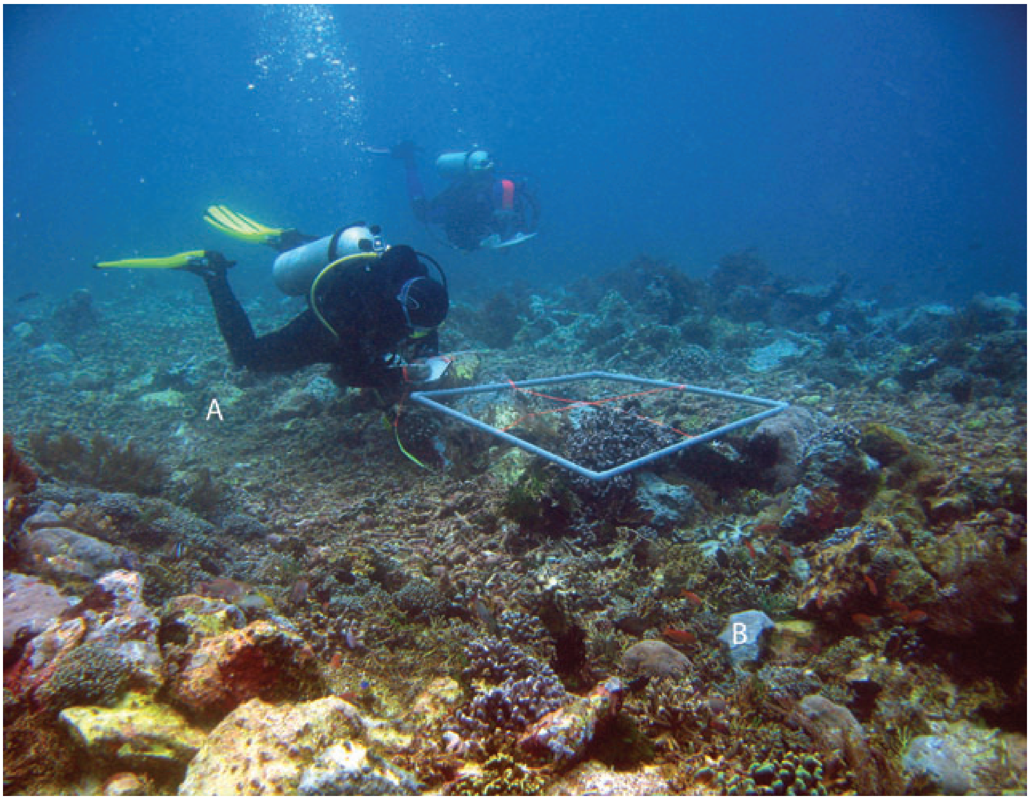

Figure 3 Parallel row treatments (originally 50–90 cm high) at the highest current site (NE Padar). While rocks are nearly completely buried by rubble in some areas (A), they have also stabilized other areas sufficiently to allow corals to grow between rows (B).

coverage quadrats sampled were at 5–6 m depth (Appendix S2).

Although the rocks did promote coral growth, at some sites rocks were being buried by encroaching rubble. This varied with location and current strength (Figure 2 and ANOVA on number of rocks above the rubble surface 6 years after installation: $F = 22.048$, df = 3,97, $P < 0.0001$). Sites with lower current conditions experienced 50.2% less rubble encroachment than sites with high currents, although the site with the least current also experienced sedimentation and had the lowest coral growth (934.1 cm^2/m^2, or 9.34%, Figure 2). Conversely, mean hard coral coverage on the rock treatments was greatest at the highest current site (21.4%), although many rock treatments were nearly completely buried by rubble (Figure 3). Currents in general are strong in Komodo National Park, and cause rubble motion (Fox *et al.* 2003). Treatments situated farther from shore experienced stronger currents and more rubble encroachment (H. Fox, pers. obs.). While slope can influence rubble movement, it was not a major factor in rubble encroachment at the sites selected (Fox 2002). When both coral growth and rubble encroachment were taken into account, the most rehabilitation success occurred at moderate current levels because the high current sites experienced an additional 40% rubble encroachment compared to the moderate current sites. Rehabilitation expenses for rock pile installations were US$30,891 to cover 6,430 m^2 of rubble (further details and comparisons in Appendix S2).

Enforcement

Enforcement expenses were US$183,652 in 2008 and with inflationary increases of 5% each year will cost an estimated US$256,111 in 2014.

Economic analyses

Enforcement of healthy reefs is more cost effective than rehabilitation of damaged ones based on multiple economic comparisons. Rehabilitation expenses average US$4.80 per m^2 of installation ($48,000 per ha; Fox *et al.* 2005, Table S2 and Appendix S2); to install rock piles in all damaged coral reef habitat of Komodo National Park would cost ~US$40,800,000. The net present cost

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Table 1 Cost-effectiveness of rehabilitation relative to enforcement. When rehabilitated coral is treated as a revenue stream as described in the methods, the increase in coral park-wide is 0.09% per % of the park rehabilitated.

	Percent of park rehabilitated			Without enforcement
	1%	10%	50%	
Meters^2 coral after 7 years (increase or *blasted*)*	7,709.5	77,095	385,475	*116,439*
Coral change after 7 years (% increase per specified % of rubble rehabilitated or % *blasted*)	0.0907	0.907	4.535	*1.37*
% coral in park after 7 years† (22.5% starting rate)	22.5907	23.407	27.035	21.13
% change in coral cover (22.5% starting rate)	0.4031	4.0311	20.1556	−6.0889
Cost of rock pile installation (not discounted)	\$40,800	\$4,080,000	\$20,400,000	
Cost of enforcement for 7 years (10% discount rate)				\$1,122,953
Cost per meter^2 coral cover (increase or *saved through enforcement*)	\$52.92	\$52.92	\$52.92	*\$9.64*

*All coral values and percentages are discounted (10% discount rate).
†"without enforcement" equals the amount of coral remaining if blasting were allowed to continue at pre-enforcement rates (Appendix S1) (shows the coral decrease expected).

of enforcement to patrol the Park for the 7-year period is US\$1,122,953.

Cost-per-area calculations

Although reef rehabilitation using locally quarried rocks is cheaper than many alternatives (Spurgeon & Lindahl 2000), it costs ~70 times more than enforcement when the two are directly compared using net total cost-per-area of each method (\$0.0661 per m^2 of park area for enforcement vs. \$4.80 per m^2 of rock pile installation for rehabilitation).

Cost-effectiveness model

Based on our coral coverage model we predicted that coral on rehabilitation treatments would reach 45% after ~15 years in low current areas, ~11 years in mid current areas, and ~10 years in high current areas. The cost-effectiveness model over 7 years predicts rehabilitation costs >5 times more per m^2 of increase in coral cover than enforcement: \$52.92 versus \$9.64 per m^2 (Table 1, discounted at 10%; when discounted from 2%–15% ratios are 4.96 to 5.84, respectively). While rehabilitation costs \$4.80 per m^2 of rock pile installation, 1 m^2 of 100% coral cover costs \$52.92; in other words, ~11 m^2 of rock piles need to be installed to get 1 m^2 of coral in the future due to encroachment by rubble, space between piles, and incomplete coral coverage on rocks. For enforcement the net present total amount of coral coverage saved from blasting over 7 years is 1.37% park-wide or 116,439 m^2 of 100% coral cover, which represents 6.1% of the remaining coral. At the net present cost of enforcement, saving 1 m^2 of 100% coral coverage costs \$9.64.

Discussion

Our results provide a clear illustration that preventing environmental damage is often far more cost-effective than attempting to restore habitats after the damage has occurred, supporting resilience theory (e.g., Hughes *et al.* 2005). In this case, enforcement, which protects the healthy reefs that remain in Komodo National Park, is ~5 times more cost-effective than reef rehabilitation based on our cost-effectiveness model and ~70 times more cost-effective than reef rehabilitation based on our cost-per-area calculations. Enforcement also requires less research and has the additional benefits of controlling other illegal fishing activities, which is important both for local livelihoods and for continued tourism visitation because fish biodiversity and abundance are attractive to tourists, more so than benthic features (Williams & Polunin 2000). Protecting reefs that have thus far not been blasted is especially important because more fish recruit to and prefer to live near living coral assemblages than dead coral (Feary *et al.* 2007a, b). Rehabilitation activities must therefore be seen in a mosaic of management choices, taking into consideration the condition of the reef and available human and financial resources.

However, our results are only a simple, and simplified, example with important limitations and assumptions in the costs considered. Cost-effectiveness analysis is limited because it cannot yield conclusions regarding the optimal level of action, so conducting a full cost–benefit analysis would be a logical next step. Focusing solely on change in coral cover as our metric is also an oversimplification. Because enforcement addresses many issues, the coral protection costs of enforcement would need to be shared with other objectives such as awareness-raising, monitoring, and preventing other illegal fishing methods.

Rehabilitation, too, has additional benefits, because the rock piles themselves provide habitat for fish and invertebrates (Fox *et al.* 2005), and have potential to improve fisheries productivity, whereas blasted areas provide little habitat for fish and will likely remain largely unproductive over time regardless of enforcement. Additionally, rehabilitation costs/m^2 could be reduced through economies of scale if expanded.

Although our economic analyses have limitations, several conservative assumptions strengthen our conclusions that enforcement is, in this case, more cost-effective than rehabilitation. In our study, no coral growth was assumed under enforcement, although soft corals can colonize rubble fields (Fox *et al.* 2003) and isolated blasts can recover naturally over time (Fox & Caldwell 2006). Rehabilitation costs were assumed to be a one-time cost; however, rubble encroachment data suggest that this would not be the case. Our method of rehabilitation is one of the cheaper alternatives. Costs may be much higher if a different method were used (Spurgeon & Lindahl 2000), or if natural recruitment were limited (Edwards & Clark 1998). Most significantly, we treated rehabilitation and enforcement costs as independent in order to examine the tradeoffs more closely, yet in reality, rehabilitation without enforcement is unlikely to be effective (Edwards & Clark 1998). Therefore the cost/m^2 of effective rehabilitation is actually the cost of enforcement plus the costs of rehabilitation.

Clearly, rehabilitation and enforcement are not mutually exclusive interventions. The choice to invoke one, the other, or both will depend on the starting condition as well as the biodiversity and recreational values of particular reefs. A balanced approach of effective enforcement park-wide coupled with rehabilitation in areas of high recovery potential and high tourism or educational value could be more economically viable than either enforcement or rehabilitation alone (e.g., if beach front hotels aim to ensure that their guests have a small reef to visit). Although coral reefs are perhaps the most costly coastal ecosystems to rehabilitate (Spurgeon 1998), their value is also high. Other ecosystems have proved economical to rehabilitate (e.g., reforestation of degraded areas has been widely successful [Dobson *et al.* 1997]), and there are attempts to apply similar forest restoration principles to coral reefs (Epstein *et al.* 2003).

We showed that rehabilitation in the presence of enforcement is possible, but success is variable, depending on the different current and site characteristics (Appendix S2). Indeed, considerable investment is being made in rehabilitation research (Coral Reef Targeted Research and Capacity Building for Management 2008). However, since it can be difficult for developing nations to invest in both enforcement and rehabilitation at large scales, it is crucial that the most cost-effective balance of conservation measures is implemented. We recommend that managers consider the extent of current damage and likely coral growth with or without rehabilitation, as well as the extent of ongoing destruction, to estimate the likely costs and benefits of different strategies for coral reef conservation.

Our results provide a specific example of how prevention of blast fishing through effective enforcement is much more cost-effective than repairing damage through reef rehabilitation, yet enforcement is not the only, and also not necessarily the most cost-effective instrument to achieve compliance with regulations and prevention of damage. Previous research suggests that alternative livelihood programs can be important to the success of coastal resource management (Rivera & Newkirk 1997; Pollnac *et al.* 2001), and Komodo National Park is already engaged in activities to lure fishers away from destructive practices, such as seaweed culture and learning to catch pelagic as opposed to reef fishes (Pet & Yeager 2000; Mehta-Erdmann & Bason 2004). These sustainable livelihoods can provide locals with incomes comparable to blast fishing (Ruitenbeek & Cartier 2001), and could decrease fishing pressures on areas open to fishing within the park. However, such programs are not likely to be as effective at curbing destructive practices as enforcement since they focus on individuals living within park boundaries whereas blast fishers primarily reside outside of the park (Djohani 1995).

Overfishing and illegal fishing cause heavy losses in fisheries, coastal protection, and tourism potential (for blast fishing alone these costs are estimated at \$33,900–\$306,800 per km^2 of Indonesian coral reef over 20 years [Pet-Soede *et al.* 1999]). Our results suggest that increasing investments to enhance effectiveness of enforcement and management is likely to yield high dividends. We encourage extension of the methods we used in this article to evaluate cost-effectiveness of interventions that change behavior and improve marine resources.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Additional methods.

Table S1 (A and B). Tukey post-hoc results.

Appendix S2. Comparing rock pile designs. Figure S2A. Schematic of rock pile designs. Figure S2B. Coral coverage in sampling quadrats, by treatment type and depth. Table S2. Calculation table of cost per m^2 of treatment area for each rock pile configuration at each location.

Appendix S3. Willingness-to-visit thought experiment.

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