LETTER

Evaluating Linked Social–Ecological Systems in Marine Protected Areas

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Abstract

In view of current worldwide coral reef decline, and the shortcomings of traditional top-down management schemes of Marine Protected Areas (MPAs), decision makers and scientists face the important challenge of developing new approaches to generate effective conservation strategies. This study evaluates MPAs as linked social–ecological systems (SES) to inform better management by calculating indices for ecological health, social adaptive capacity, and the impact intensity of overfishing, pollution, and tourism. A series of ecological and socioeconomic indicators are used to estimate these indices and determine relevant conservation strategies in two protected areas in the Colombian Caribbean. Results reveal a precarious situation of high impact intensity combined with low ecological health and adaptive capacity. This study provides further evidence supporting the need for reconciliation of SES and a framework by which decision makers can assess priorities to increase MPA effectiveness. We highlight the need for system reorganization and recommend bottom-up comanagement schemes as a priority strategy to strengthen adaptive capacity.

Introduction

Most Marine Protected Areas (MPAs) worldwide are underresourced, lack evidence-based management plans, and rarely achieve their conservation goals (Mora *et al.* 2006; Burke *et al.* 2011). Low efficacy of protection is a global concern given the high rates of marine environment degradation and declines in fisheries resources (Jackson *et al.* 2001; Hughes *et al.* 2003; Halpern *et al.* 2008). Marine resource management has migrated away from "optimal" harvest models based on command-and-control approaches, static environment assumptions, and separation of social and ecological issues (Berkes *et al.* 2003; Hughes *et al.* 2005; ResilienceAlliance 2010), because management relying on these approaches does not prepare the systems for dealing with changes, making them more vulnerable to anthropogenic pressures and natural disturbances (ResilienceAlliance 2010). Current management approaches have evolved to recognize the dynamics and different dimensions of communities (physical, socioeconomic, biological, institutional), to challenge maximum sustainable yield-based models, and to encompass social and ecological components that interact closely, forming coupled and integrated systems known as social–ecological systems (SES) (Mangel & Levin 2005; Liu *et al.* 2007; Levin *et al.* 2009). Investigating SES is complicated by reciprocal interactions generating feedback loops, exhibiting complex dynamics affected by several factors (e.g., markets, institutions, environmental changes), and nonlinear relationships with thresholds, inherent uncertainties, and varying degrees of resilience (Costanza *et al*. 1993; Hughes *et al*. 2005; Liu *et al*. 2007).

Since SES are dynamic, they can change and transform into multiple alternative states (Folke 2006). Regime shifts involve a change in community structure that alters the generation of ecosystem services, causing subsequent impacts on human society (Elmqvist *et al*. 2003; Folke *et al*. 2004). For instance, coral reefs exhibit several alternative stable states such as coral, macroalgal and sea urchin dominated as a result of stressors that make them vulnerable to change, such as reduced herbivory, water pollution, reduced fish stocks, high-erosion level, and coral disease (Nyström *et al*. 2000; Bellwood *et al*. 2004; Norström *et al*. 2009).

In the study of SES, difficulties arise when trying to consolidate and interpret findings coming from diverse sources and several data collection methods (Ostrom 2007). Moreover, the inherent differences of environmental and social sciences impose challenges for scientists seeking to draw connections to holistic analyses (Ostrom 2009). Hence, recent multidisciplinary approaches and frameworks have been developed aiming to investigate marine SES, and have provided important progress toward their understanding (Cinner *et al*. 2009; McClanahan *et al*. 2009; Pollnac *et al*. 2010; Halpern *et al*. 2012)

Worldwide coral reefs have suffered dramatic declines despite the existence of MPAs and associated management plans (Camargo *et al*. 2009; De'ath *et al*. 2012). This highlights the need to perform comprehensive evaluations of MPA effectiveness while incorporating approaches based on SES, as the success of marine reserves highly depend on the interactions between their ecological and social dimensions (Pollnac *et al*. 2010). The aim of this study is to examine two MPAs in the Colombian Caribbean as linked SES to recommend conservation strategies for enhancing MPA management effectiveness (the extent to which management is protecting natural resources and assuring local community welfare), by identifying factors that may be enhancing or eroding it.

Methods

Study site

We selected two MPAs located in the Colombian Caribbean as the focal systems of the study (Figure 1): Rosario and San Bernardo Coral National Natural Park (CRSB) established in 1977 is 1,200 km^2 in size and composed of two archipelagos including coral reef (67.6 km^2), mangrove and tropical dry forest habitats. Tayrona National Natural Park (PNNT), established in 1969, is a coastal MPA of 150 km^2, also representing coral reef (6.54 km^2), mangroves, and dry and rain forest ecosystems. Historically, coral reefs in these areas have been affected by pollution, sedimentation, overfishing, dynamite fishing, and coral mining during the 1980s; minor bleaching events in 1987, 1990, 1995, and 1998; and the severe bleaching in 2005 that left extensive patches of dead *Acropora*, however, most affected reefs have been shown to recover after 6 months (Wilkinson & Souter 2008). During the first 25 years of the MPAs designation, extraction of marine resources were prohibited even within traditional fishing grounds of afro-descendants and indigenous communities who had been inhabiting these areas for centuries. However, in the late 1990s a new policy of *Social Participation for Conservation* was introduced in the national parks of Colombia, increasingly involving local communities with environmental education programs and capacity building. These efforts have subsequently weakened however, as conflicts involving coastal development, tourism, and land tenure continue to create tension between the communities and authorities (Durán 2009). Inhabitants of both MPAs have been dependent on natural resources (fishing and tourism) for livelihoods since long before the declaration of protection schemes, so social–ecological dynamics are highly determined by management decisions. These communities are characterized by low income, lack of access to fresh water, sanitation, health and education, and scarce access to other capitals beyond natural capital (Camargo *et al*. 2009), despite their proximity to neighboring cities (Cartagena and Santa Marta).

Conceptual model of the MPAs

We conducted a workshop in Santa Marta in April 2008, with the objective of evaluating and selecting multidisciplinary indicators of ecological health, adaptive capacity and impact intensity. Fifty stakeholders participated, representing local environmental organizations (10), MPA staff (15), conservationists and researchers (18), and local fishermen (7). Workshop participants provided their opinion of indicators after listening to talks and open discussions. Systematization of different opinions expressed was possible through Régnier's Abacus method (Lafourcade & Chapuy 2000): Participants voted in a scale of colors that codified the level of agreement or disagreement. Accordingly only the most voted indicators were included in this study (see supporting information Table S1). Following the workshop, results were integrated with existing models (Berkes *et al*. 2003; Chapin *et al*. 2006, 2009),

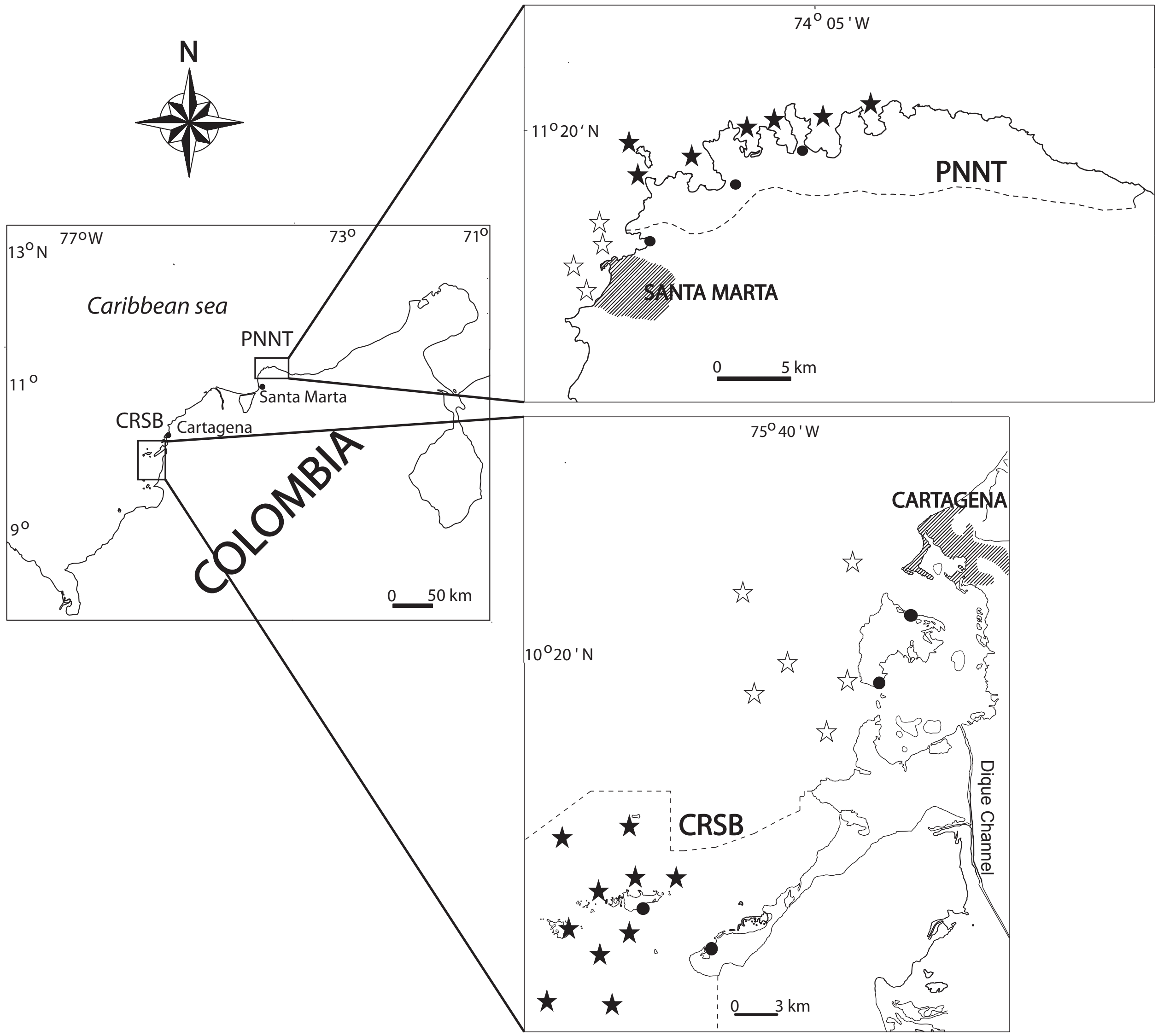

Figure 1 Location of the studied MPAs on the Caribbean coast of Colombia. Solid stars represent biophysical survey sites inside MPAs, open stars represent sites outside MPAs. Local communities are shown as solid circles. Dotted lines indicate the limits of the MPAs.

to generate a conceptual model (Figure 2) to form the framework of our assessment and a set of indicators based on existent methodologies (Walker *et al*. 2006; Obura & Grimsditch 2009; ResilienceAlliance 2010)

We designed measurements and indicators to calculate three indices: (1) an index of ecological health, (2) an index of social adaptive capacity, and (3) an index of local impact intensity. To calculate the indices, we designed a categorical scale of ecological health, social adaptive capacity, and impact intensity, where each measurement was scored according to thresholds selected by validating with local experts and consulting available literature (Table S1). Once determined, each measurement was classified into four levels ranging from very low to high with an associated numeric value (very low = 0, low = 1/3, moderate = 2/3, and high = 1). Mean measurements scores were used to represent an overall score per indicator and impact, which were classified again (very low = 0–0.25, low = 0.26–0.5, moderate = 0.51–0.75, and high = 0.76–1). Values of indices were calculated using a consistently weighted average numeric score of all indicators and normalizing the result to the unit (Tables 1, 2, and 3). Ecological and social indicators with mean scores of *very low* and *low* were considered as

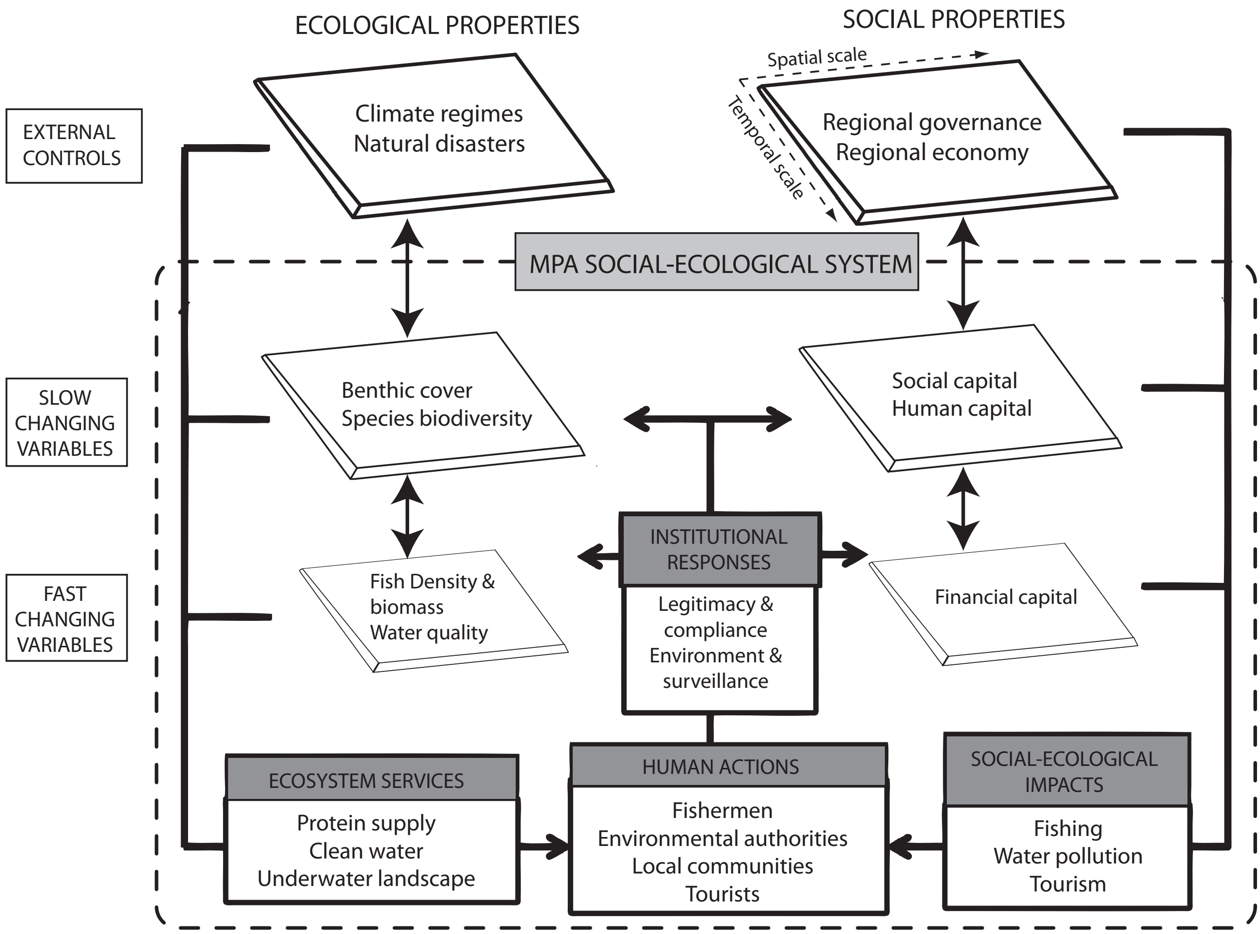

Figure 2 Diagram showing the conceptual model of the studied SES constructed around the local impacts (fishing, water pollution, and tourism) that were perceived as the most relevant in the workshop. At a regional scale, external controls such as climate change, regional economy, and governance affect the MPA. Among slow changing ecological variables are benthic cover and biodiversity. Social capital and human capital are included as slow changing social variables. Slow changing variables influence fast changing variables such as fish biomass and density, water quality, and financial capital. The fluctuations in these variables alter the provision of ecosystem services, affecting the well-being of human actors, who modify the subsystems through institutional responses such as management and governance. In this study slow and fast variables were treated as indicators, however we are aware of the difficulty of capturing the dynamics of the system, so the ecological indicators summarize the ecosystem condition instead (adapted from Chapin (2006).

potential degraders of effectiveness, and *moderate* and *high* considered as enhancers.

Ecological data collection

Data was collected during 2008–2009 at 17 sites located inside the two MPAs (PNNT = 7, CRSB = 10) and 10 sites outside (PNNT = 4, CRSB = 6) (Figure 1). We sampled areas in shallow (3–15 m) highly developed reefs, ensuring geomorphological and environmental similarities. We conducted underwater visual fish censuses by swimming along 2 m × 50 m belt transects (100 m^2) recording all individuals of families important to local livelihoods (Acanthuridae, Lutjanidae, Serranidae, Scaridae, Haemulidae, Carangidae, Scombridae, Sphyraenidae) and estimating total fish length; between 4 and 10 transects were carried out for each site. Fish density was compared between MPA and non-MPA survey sites in each study area (PNNT and CRSB) using a Mann–Whitney U test. Herbivore and predator biomass were estimated following Friedlander *et al*., (2003) and transformed ($\log\times + 1$) to compare MPA and non-MPA survey sites in each study area using ANOVA. Analyses were run in R statistical software.

We estimated relative abundance of coral, crustose coralline algae, microalgal turfs, and frondose macroalgae

Table 1 Indicators and measurements used to determine the ecological health index for two MPAs in the Colombian Caribbean. Results for each measurement and score according to the classification scales are shown by MPA. Please see supporting information Table S2 for the classification values of results on the scale. Indicators with mean scores of *very low* and *low* were considered as potential degraders of effectiveness, and *moderate* and *high* considered as enhancers

Indicator	Result		Score[a]		
Measurement	PNNT	CRSB	PNNT	CRSB	Source[b]
Benthic cover			Low (0.33)	Very low (1.17)	
Coral cover percentage	22%	23.2%	Low	Low	PT
Algae cover percentage	48%	66.8%	Low	Very low	PT
Fish density and biomass			Very low (0)	Very low (0)	
Mann–Whitney test significance (*P*) for differences between fish densities inside and outside MPA	0.25	0.89	Very low	Very low	UVS
ANOVA test significance (*P*) for differences between herbivore fish biomass inside and outside MPA	0.6	0.13	Very low	Very low	UVS
ANOVA test significance (*P*) for differences between predator fish biomass inside and outside MPA	0.64	0.54	Very low	Very low	UVS
Species diversity			Low (0.33)	Low (0.33)	
Fish beta diversity. Shannon Index (*H*)	2.71	2.56	Moderate	Low	UVS
Coral beta diversity. Shannon Index (*H*)	1.6	2.31	Very low	Low	UVS
Water quality			Low (0.33)	Low (0.33)	
Foram Index (FI)	3.42	3.75	Low	Low	FA

[a]Numeric scores for the indicators in parenthesis.
[b]Sources: PT = phototransects, UVS = underwater visual surveys, FA = Foraminifer's analysis.

within 1 m^2 with digital photographs taken from 2 × 50 m belt-transects (100 pictures per transect), one transect per site (López-Angarita *et al*. 2011). We used *ImageJ* software (National Institutes of Health) to determine percentage cover of each group. We used data for the Caribbean (Gardner *et al*. 2003) to calibrate threshold values for the benthic cover indicator (Table S1 and S2). PRIMER software was used to calculate Shannon diversity index values per site for fish and coral; mean values were compared against thresholds selected according to Ramirez-Gonzalez (2006) for fish, and Porter (1972) for coral. The latter was well before many of the dramatic changes that Caribbean reefs have suffered in recent decades (e.g., *Acropora* die-off 1982–1983, *Diadema* die-off 1984, 1998/2005/2010 bleaching events, etc.), hence we believe this serves as an appropriate baseline.

To determine water quality we used data from the FORAM index by Velazquez *et al*. (2011), where FI > 4 indicates an environment conducive to reef growth; 2 < FI < 4 represents the limit for coral growth and unsuitable for recovery; and FI < 2 indicates unfavorable conditions for coral growth (Hallock *et al*. 2003; Uthicke *et al*. 2010)

Social data collection

Various sources of information were used to gather socioeconomic data. The main components of this study were built upon primary information. However, pollution measurements (access to sewage system, sanitary services, and solid waste disposal) were obtained using secondary information from the Colombian 2005 census data (http://www.dane.gov.co/censo/). First, to gauge the perspective of environmental authorities on the indicator of rules legitimacy and compliance, enforcement and surveillance, and human capital, we conducted interviews with 14 officers from CRSB and 12 from PNNT (covering more than half of the staff). We designed and applied a structured survey for tourists visiting the MPAs, totaling 816 for CRSB and 160 in PNNT. Information from tourists was used for rules legitimacy and compliance indicators.

We collected perceptions and behavior patterns of local communities during focus groups using participative rural diagnosis on six groups from local communities living inside or around the MPAs (three in each). A total of 90 persons from PNNT and 66 from CRSB were involved. Workshops were designed to capture particular subsets of information about the communities and their relationship with the ecosystem, as well as to construct an indicator of fishing impact.

Framed Economic Experimental Games (EEGs) with local communities simulated the extraction of common pooled resources, specifically fishing. EEGs mimic the dilemma of resource use, characterized by simultaneously exhibiting nonexclusion and rivalry (Berkes *et al*. 1989;

Table 2 Indicators and measurements used to determine the social adaptive capacity index for two MPAs in the Colombian Caribbean. Results for each measurement and score according to the classification scales are shown by MPA. Please see supporting information Table S3 for the classification values of results on the scale. Indicators with mean scores of *very low* and *low* were considered as potential degraders of effectiveness, and *moderate* and *high* considered as enhancers

Indicator	Result		Score[a]		
Measurement	PNNT	CRSB	PNNT	CRSB	Source[b]
Rules legitimacy and compliance			Moderate (0.52)	Moderate (0.62)	
Proportion of park rangers that consider rules are designed to achieve conservation objectives	58.33%	64.28%	Moderate	Moderate	PRS
Proportion of park rangers that consider local community follow fishing rules	43.75%	38.46%	Low	Low	PRS
Proportion of park rangers that consider environmental authority has legitimacy in the community	70.83%	60.71%	Moderate	Moderate	PRS
Proportion of local community surveyed that has a favorable perception regarding MPA existence	76.66%	93.4%	Moderate	High	LCS
Proportion of the community that agrees that denouncing the noncompliance of formal rules is useful	65.51%	48.93%	Moderate	Low	LCS
Proportion of local community surveyed that declares knowing the rules for the MPA	59.32%	60.85%	Moderate	Moderate	LCS
Proportion of tourists surveyed that declares knowing the rules of entering into a MPA	23%	51.3%	Very low	Moderate	TS
Enforcement and surveillance			Low (0.44)	Moderate (0.56)	
Proportion of community that considers the park has capacity to enforce rules	49.15%	61.27%	Low	Moderate	LCS
Proportion of park rangers that consider the sanctions are easy to implement	33.33%	21.43%	Low	Very low	PRS
Frequency in surveillance routes and coverage of control in MPA	3 days/week	Every day	Moderate	High	PRS
Social capital			Moderate (0.58)	Very low (0.25)	
Potential internal cooperation within communities	32.62%	22.85%	Low	Very low	EEG
Proportion of reduction in extraction as a result of a comanagement rule in EEGs	58.86%	56.11%	Moderate	Moderate	EEG
Proportion of community that consider the natural resources management must be between community and environmental authority	66.1%	22.87%	Moderate	Very low	LCS
Percentage of community that participate in meetings and workshops with park authorities	66.1%	47.23%	Moderate	Low	LCS
Human capital			Moderate (0.67)	Very low (0.17)	
Average years of academic studies	7.78	5.28	Moderate	Low	LCS
Frequency of informal training activities for fishermen	Fortnightly	Monthly	Moderate	Very low	PRS
Financial capital			Low (0.33)	Low (0.33)	
Average household income per month	US$ 399	US$ 226	Low	Low	LCS

[a]Numeric scores for the indicators in parenthesis.

[b]Sources: PRS = park rangers survey, LCS = local community survey, TS = tourists survey, EEG = economic experimental games.

Feeny *et al*. 1990). In EEGs, players must decide how much to harvest (e.g., fishing extraction), and they obtain points that are convertible into money at the end of the game, which tests the individual's response to alternative management schemes for natural resources use (Moreno-Sánchez & Maldonado 2010). We performed games with 235 participants from eight communities located inside or around CRSB, and 60 participants from three communities located close to PNNT. In addition, participants of EEGs completed surveys on their socioeconomic and demographic backgrounds.

To establish thresholds for social measurements, presented as percentages and frequencies of activities, we divided the percentage equally among the four categories (Tables S3 and S4). Income thresholds followed the approach by the World Bank to define poverty (<\$2 per person per day) and extreme poverty (<\$1.25 per person per day). For the indicators of pollution and tourism, we included spatial information using ArcGIS (9.1) to measure distances from sampled coral reefs to stressors. We assumed a distance-based interpretation by which the threat declines with distance to stress

Table 3 Indicators used to determine local impact intensity for two MPAs in the Colombian Caribbean. Results for each measurement and score according to the classification scales are shown by MPA. Please see supporting information Table S4 for the classification values of results on the scale

Impact		Result		Score[a]		
Measurement	Description	PNNT	CRSB	PNNT	CRSB	Source[b]
Fishing impact				Low (0.44)	Moderate (0.56)	
Perception about fishing impact	Percentage of fishermen that consider fishing is highly impacting the ecosystem	43%	38%	Low	Low	LCS
Use of destructive fishing arts	Use of destructive gear	Yes	Yes	High	High	PW
Frequency of herbivores in fishing	Percentage of herbivorous fish in harvest	3%	30.58%	Very low	Low	SI
Pollution impact				Moderate (0.73)	High (0.80)	
Access to sewage system	Percentage of population having access to sewage system	2.5%	5.38%	High	High	SI
Access to sanitary service	Percentage of rural households that have sanitary service connected to a sewage system	1.78%	3.71%	High	High	SI
Adequate solid-waste disposal	Percentage of rural households that report municipality is in charge of waste disposal	56%	14%	Low	High	SI
Distance to urban settlements	Distance to inhabited location	9.37	13.18	Moderate	Low	GIS
Distance to pollution sources	Distance (kilometers) to points of discharge to the sea (waste and sewage) and ports	20.48	23.42	Moderate	Moderate	GIS
Tourism impact				Low (0.33)	Moderate (0.67)	
Distance to tourist places	Distance (kilometers) to tourist places	11.17	5.86	Low	Moderate	GIS

[a]Numeric scores for the impacts and indicators in parenthesis.
[b]Source: LCS = local community survey, PW = participative workshop with community, SI = secondary information, GIS = geographic information system.

sources (Burke & Maidens 2004). Differences in indices between MPAs were analyzed with nonparametric statistics (Mann–Whitney U test).

We combined and adapted the analytical frameworks developed by McClanahan *et al.* (2008, 2009) and Cinner *et al.* (2012) to interpret the indices, and plotted the social adaptive capacity index and ecological health index against the impact intensity index. Plotting indices creates a space with four possible scenarios where particular management actions should be implemented.

Results

Ecological health index

Algae dominated the reef benthos (PNNT = 48%, CRSB = 67%) and coral cover was low in both MPAs (PNNT = 22%, CRSB = 23%) (Table 1). Fish density at sites within the MPAs showed no significant differences compared to populations outside MPAs for PNNT (S = 25378, $Z = -1.14$, $P = 0.25$) or CRSB (S = 84509, $Z = -0.13$, $P = 0.894$). Similarly, no significant reserve effects were seen in biomass of herbivores (PNNT: $F = 0.29$, $P = 0.6$; CRSB: $F = 2.65$, $P = 0.13$) or predators (PNNT: $F = 0.24$, $P = 0.64$; CRSB: $F = 0.39$, $P = 0.54$). Fish biomass and density indicators were scored as very low on the health scale. Shannon diversity for fish was higher in PNNT ($H = 2.71$) than in CRSB ($H = 2.56$), and conversely, coral species diversity was higher in CRSB ($H = 2.318$) than in PNNT ($H = 1.6$). According to the threshold values of Shannon index used in the health scale, fish and coral species diversity in CRSB were low; in PNNT, despite fish being moderate on the scale, coral diversity was very low, making the average overall score low (Table 1). FORAM index had low scores in both MPAs (PNNT = 3.42, CRSB = 3.75) represented by values between 2 and 4. In general, most of the ecological indicators scores were low and very low and potentially degraders of effectiveness. The ecological health index normalized to the unit was 0.25 for PNNT and 0.21 for CRSB.

Social adaptive capacity index

The rules legitimacy and compliance indicator showed a moderate value for both MPAs (score PNNT = 0.52, score CRSB = 0.62) (Table 2). Park rangers perceptions were similar for both MPAs, as more than 50% of them believed rules were well designed in both MPAs, and 60–70% considered the environmental authority had legitimacy in the community, despite around 60% believing that locals did not follow fishing rules. A high proportion of respondents during focus groups with locals showed a favorable perception toward the MPAs (CRSB = 76%, PNNT = 93%) and around 60% declared knowing the rules, but only 65% (CRSB) and 49% (PNNT) said they would denounce rule-breaking by others. A greater proportion of tourists showed awareness of MPA regulation in CRSB (51%) than in PNNT (23%).

The enforcement and surveillance indicator exhibited low values in PNNT (0.44) and moderate in CRSB (0.56). Only 33% of park rangers in PNNT and 21% in CRSB considered sanctions easy to implement, while a greater proportion of communities considered authorities to have enforcement capacity in both parks (60% CRSB vs. 50% PNNT).

Social capital scored moderate in PNNT (0.58) but very low in CRSB (0.25) (Table 2). Both groups scored low for potential internal cooperation between communities (CRSB = 23% vs. PNNT = 33%). Although the proportion of the community in support of comanagement during the EEGs increased to near 60% in both parks (CRSB = 56% vs. PNNT = 59%), communities in CRSB showed more reluctance to accept this type of management scheme (CRSB = 23%, PNNT = 66%). Human capital measured as a frequency of training from authorities and average years of academic studies, was moderate in PNNT, but very low in CRSB. Financial capital was ranked as low in both communities (0.33). When combining these indicators, the social adaptive capacity index normalized to the unit was 0.51 for PNNT and 0.38 for CRSB.

Impact intensity index

The local impact intensity index normalized to the unit was 0.50 for PNNT and 0.67 for CRSB. Fishing impact effect was moderate and slightly higher in CRSB than in PNNT (0.56 vs. 0.44). The use of destructive fishing gears was recognized, and yet less than half of surveyed fishermen accepted that fishing highly impacts the ecosystem (43% PNNT and 38% CRSB). Herbivorous fish represented 31% of the harvest in CRSB and 3% in PNNT (Table 3). The pollution indicator scored moderate and high intensity (PNNT = 0.73, CRSB = 0.80). Reefs were relatively close to urban centers (PNNT = 9.37 km, CRSB = 13.18 km) and points of waste disposal and sewage (PNNT = 20.48 km, CRSB = 23.42 km). In both MPAs, less than 5% of locals had access to sewage system and sanitary service, and collection of solid waste by the municipality was markedly low.

Tourism impact was scored as low in PNNT (0.33) and moderate in CRSB (0.67) (distance to city centers: PNNT = 11.17 km, CRSB = 5.86 km).

Wilcoxon Rank Tests showed no significant differences between the MPAs for both social and ecological indices ($N = 25$, $Z = 0.71$, $P = 0.47$) or for impact intensity index results ($N = 9$, $Z = -0.6$, $P = 0.54$).

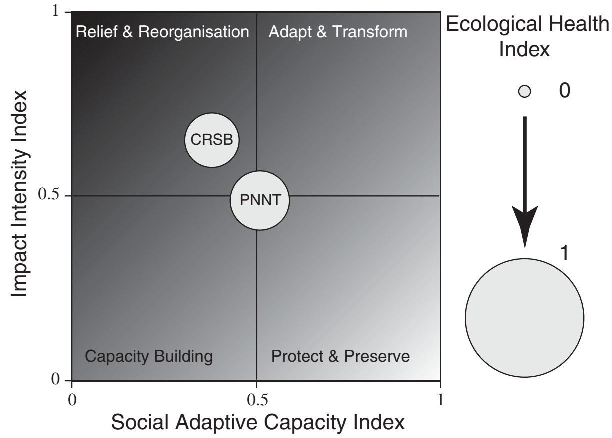

Figure 3 Plot of the social adaptive capacity index against impact intensity index. The third dimension of the analysis, the ecological health index, is represented by the size of the bubble (larger = healthier). This theoretical model used to establish conservation actions was adapted from McClanahan *et al*. (2008) and Cinner *et al*. (2009). Plotting the indices creates a space with four possible scenarios where particular management actions should be implemented. Lighter color represents a more desirable state of the SESs. Both MPAs have low ecological health, however CRSB is in a more vulnerable scenario due to low adaptive capacity and increasing local impact. PNNT has potential to move toward more desirable scenarios.

The plot of the indices showed low ecological health in both MPAs. Low social adaptive capacity and higher impact intensity placed CRSB into the quadrant corresponding to relief and reorganization (Figure 3), following McClanahan *et al*. (2008). PNNT was placed in the center, without falling into one clear strategy.

Discussion

This study represents a contribution to the growing number of models assessing ocean health, as it includes higher resolution data of an understudied region, necessary to improve the power of these evaluations (Halpern *et al*. 2012). Our model was successful in providing a better understanding of the MPA performance incorporating SES complexities and identifying factors that may enhance or erode effectiveness. The ecological subsystem in PNNT and CRSB showed signs of damage as all indicators behaved as degraders: (1) algae dominated the benthos and coral cover was low, (2) fish density and biomass suggested no response to protection, (3) fish and coral diversity was low, and (4) poor water quality provided an environment unsuitable for recovery. These results are unexpected given that the MPAs have been operating for almost 40 years, but are consistent with regional declining trends of fish abundances (Paddack *et al*. 2009) and coral cover (Gardner *et al*. 2003). Other studies in PNNT and CRSB have recorded low densities and biodiversity of fish (Camargo *et al.* 2009), coral cover declining while algae increasing (INVEMAR 2012), and high concentrations of nutrients and sediments (Vivas-Aguas *et al.* 2010) that lead to poorer water quality in areas inside the MPAs than outside (Velásquez *et al.* 2011). Algae-dominated reefs have poor productivity and diversity, which translates into the loss of response diversity and functional redundancy vital for ecosystem recovery (Bellwood *et al.* 2004). Additionally, these reefs exhibit high larval retention by coral populations revealing limited connectivity (Foster *et al.* 2012). Oligotrophic waters and the higher levels of genetic diversity brought by connectivity are essential for maintaining an integral system resilient to disturbances (Hallock *et al.* 2003; Van Oppen & Gates 2006).

In the social domain legitimacy of rules and compliance behaved as enhancers, given the positive attitude of park rangers, local communities and tourists in valuing MPAs and the need for conserving their services provision. However, in practice, enforcing rules seems difficult because of large MPA size and budget constraints of environmental authorities. Social capital was higher in PNNT since communities supported the need for sharing responsibilities between authorities and local stakeholders, whereas CRSB communities were more reluctant as a result of past experiences where strong, top-down policies have discouraged local involvement (Moreno-Sánchez & Maldonado 2010). However, during the EEGs both communities reduced extraction in a hypothetical, comanagement scenario, showing potential for adaptability. The communities are restricted by factors related to human and financial capitals. Low education levels and high poverty trap local communities in activities such as fishing, in turn affecting the adaptive capacity of SES (Cinner 2011). This situation is more severe in the larger MPA (CRSB) where communities are more isolated and have less access to both formal education and training. Potential degraders of effectiveness found in this study require attention from decision makers, as local stakeholders have traditionally been ignored in the process of creating and maintaining MPAs, participatory factors that have proved crucial in compliance and in MPA success (Pollnac *et al.* 2001). The impact of pollution was the highest since most communities did not have sewage systems or sanitary services, and untreated waters were discharged directly to the sea (Vivas-Aguas *et al.* 2010).

The ecological health, adaptive capacity, and impact intensity indices suggested CRSB and PNNT lack institutional and biophysical resources for overcoming current anthropogenic impacts. The interpretation model positioned CRSB in the reorganization and relief area, which according to McClanahan *et al.* (2009) corresponds to regions that do not have resources or ability to adapt to high levels of external impacts and need to strengthen their social networks, find alternative sources of livelihoods, and reduce the high dependence on local reef resources at risk. PNNT was located in the center suggesting it has more potential to move toward desirable scenarios, however given the low ecological health we suggest the conservation action to be reorganized as a precautionary approach. Part of this reorganization should include (1) increasing community participation in MPA decision-making and management processes by building strong linkages and trust between park managers and local communities and other stakeholders for nurturing self-organization (Walker 2006; Ostrom 2009); (2) building community capacity by reinforcing local ecological knowledge through capacity building on specific issues and knowledge sharing (Folke *et al.* 2002; Cinner *et al.* 2009); and (3) promoting ecosystem monitoring of key ecosystem variables through the development of dynamic warning indicators of resilience loss (Folke *et al.* 2002; Littler & Littler 2007), ideally with active participation of locals. These concrete actions would reduce vulnerability by building a more suitable scenario for the conciliation of social and ecological systems.

Alternative schemes different to the traditional top-down approach, such as comanagement of natural resources in MPAs, have been demonstrated to be a more effective way of dealing with the challenge of conserving marine biodiversity (Moreno-Sánchez & Maldonado 2010). In attempting to quantify dynamic SES, we are aware of the limitations in generating ecological inferences and delivering concluding remarks on MPA management effectiveness with low spatial and temporal replication. However, our approach is shown to be a practical tool for environmental authorities to uncover and understand relationships among social and ecological dimensions of MPAs, and identify those variables that should be prioritized to improve effectiveness. Applying methodologies to evaluate MPAs as SES, along with periodic monitoring of ecological and social indicators, will provide comprehensive information to determine which are the major drivers causing an effect on ecological health, and highlight how indicators change in response to management measures, helping to minimize incurred degradation costs to local communities. This signifies a vital step toward adaptive management, allowing managers to assess the SES over time and make decisions based on the current trajectories of their systems.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

The following supplementary material is available for this article:

Table S1. Rationale of the indicators selected for the study. Relevance in the literature, and theoretical justifications for the values used as thresholds to construct the scales are shown. Indicator importance is presented as the mean score obtained applying Régnier's Abacus method. Workshop participants voted in a scale of importance ranging from 0 (less important) to 4 (very important)

Table S2. Measurements used to determine the ecological health index and values used as reference to classify results on the scale

Table S3. Measurements used to determine the social adaptive capacity index, and values used as reference to classify results on the scale

Table S4. Measurements used to determine local impact intensity and values used as reference to classify results on the scale

This material is available as part of the online article from: http://www.blackwell- synergy.com/doi/full/10.1111/j.1755–263X.2008.00002.x

(This link will take you to the article abstract).

References

Bellwood, D., Hughes, T., Nyström, M. & Folke, C. (2004). Confronting the coral reef crisis. *Nature*, **429**, 827-833.

Berkes, F., Colding, J. & Folk, C. (2003). *Navigating social-ecological systems: building resilience for complexity and change*. Cambridge University Press, Cambridge.

Berkes, F., Feeny, D., McCay, B.J. & Acheson, J.M. (1989). The benefits of the commons. *Nature*, **340**, 91-93.

Burke, L. & Maidens, J. (2004). *Reefs at Risk in the Caribbean*. World Resources Institute, Washington.

Burke, L., Reytar, K., Spalding, M. & Perry, A. (2011). *Reefs at risk revisited*. World Resources Institute, Washington.

Camargo, C., Maldonado, J.H., Alvarado, E. *et al*. (2009). Community involvement in management for maintaining coral reef resilience and biodiversity in southern Caribbean marine protected areas. *Biodivers. Conserv.*, **18**, 935-956.

Chapin, F., Kofinas, G. & Folke, C. (2009). *Principles of ecosystem stewardship*. Springer, New York.

Chapin, F., Lovecraft, A., Zavaleta, E. *et al*. (2006). Policy strategies to address sustainability of Alaskan boreal forests in response to a directionally changing climate. *PNAS*, **103**, 16637-16643.

Cinner, J., Fuentes, M. & Randriamahazo, H. (2009). Exploring Social Resilience in Madagascar's Marine Protected Areas. *Ecol. Soc.*, **14**, 41.

Cinner, J.E. (2011). Social-ecological traps in reef fisheries. *Global Environ. Change*, **21**, 835-839.

Cinner, J.E., McClanahan, T.R., Graham, N.A.J. *et al*. (2012). Vulnerability of coastal communities to key impacts of climate change on coral reef fisheries. *Global Environ. Change*, **22**, 12-20.

Costanza, R., Wainger, L., Folke, C. & Mäler, K. (1993). Modeling complex ecological economic systems. *BioScience*, **43**, 545-555.

De'ath, G., Fabricius, K.E., Sweatman, H. & Puotinen, M. (2012). The 27-year decline of coral cover on the Great Barrier Reef and its causes. *PNAS*, **109**, 17995–17999.

Durán, C.A. (2009). Gobernanza en los Parques Nacionales Naturales colombianos. *Revista de Estudios Sociales*, 60-73.

Elmqvist, T., Folke, C., Nystrom, M., Peterson, G., Bengtsson, J., Walker, B. & Norberg, J. (2003). Response diversity, ecosystem change, and resilience. *Front. Ecol. Environ.*, **1**, 488-494.

Feeny, D., Berkes, F., McCay, B. & Acheson, J. (1990). The tragedy of the commons: twenty-two years later. *Human Ecol.*, **18**, 1-19.

Folke, C. (2006). Resilience: the emergence of a perspective for social–ecological systems analyses. *Global Environ. Change*, **16**, 253-267.

Folke, C., Carpenter, S., Elmqvist, T., Gunderson, L., Holling, C. & Walker, B. (2002). Resilience and sustainable development: building adaptive capacity in a world of transformations. *Ambio*, **31**, 437-440.

Folke, C., Carpenter, S., Walker, B. *et al*. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. *Ann. Rev. Ecol., Evol., Systemat.*, **35**, 557-581.

Foster, N., Paris, C., Kool, J. *et al*. (2012). Connectivity of Caribbean coral populations: complementary insights from empirical and modelled gene flow. *Mol. Ecol.*, **21**, 1143-1157.

Friedlander, A.M., Brown, E.K., Jokiel, P.L., Smith, W.R. & Rodgers, K.S. (2003). Effects of habitat, wave exposure, and marine protected area status on coral reef fish assemblages in the Hawaiian archipelago. *Coral Reefs*, **22**, 291-305.

Gardner, T., Côté, I., Gill, J. & Grant, A. (2003). Long-term region-wide declines in Caribbean corals. *Science*, **301**, 958-960.

Hallock, P., Lidz, B., Cockey-Burkhard, E. & Donnelly, K. (2003). Foraminifera as bioindicators in coral reef assessment and monitoring: the FORAM index. *Environ. Monitor. Assess.*, **81**, 221-238.

Halpern, B.S., Longo, C., Hardy, D. *et al.* (2012). An index to assess the health and benefits of the global ocean. *Nature*, **488**, 615-620.

Halpern, B.S., Walbridge, S., Selkoe, K.A. *et al.* (2008). A global map of human impact on marine ecosystems. *Science*, **319**, 948-952.

Hughes, T., Baird, A.H., Bellwood, D.R. *et al.* (2003). Climate change, human impacts, and the resilience of coral reefs. *Science*, **301**, 929-933.

Hughes, T., Bellwood, D., Folke, C., Steneck, R. & Wilson, J. (2005). New paradigms for supporting the resilience of marine ecosystems. *Trends Ecol. Evol.*, **20**, 380-386.

INVEMAR. (2012). *Informe del estado de los ambientes y recursos marinos y costeros en Colombia: Año 2011 (No. 8)*. Serie de Publicaciones Periódicas, Santa Marta.

Jackson, J., Kirby, M., Berger, W. *et al.* (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science*, **293**, 629-638.

Lafourcade, B. & Chapuy, P. (2000). Scenarios and Actors' Strategies: the case of the agri-foodstuff sector. *Technol. Forecast. Social Change*, **65**, 67-80.

Levin, P.S., Fogarty, M.J., Murawski, S.A. & Fluharty, D. (2009). Integrated ecosystem assessments: developing the scientific basis for ecosystem-based management of the ocean. *PLOS Biol.*, **7**, 23-28.

Littler, M.M. & Littler, D.S. (2007). Assessment of coral reefs using herbivory/nutrient assays and indicator groups of benthic primary producers: a critical synthesis, proposed protocols, and critique of management strategies. *Aqua. Conserv.: Mar. Freshw. Ecosyst.*, **17**, 195-215.

Liu, J., Dietz, T., Carpenter, S.R. *et al.* (2007). Complexity of coupled human and natural systems. *Science*, **317**, 1513-1516.

López-Angarita, J., Moreno-Sanchez, R., Restrepo, M., Alvarado, E., Maldonado, J. & Sánchez, J. (2011). Resiliencia socioecológica en el parque nacional natural corales del rosario y san bernando. Pages 364-376 in E. Zarza-González, editor. *El entorno ambiental del pnn corales del rosario y de san bernardo*. Parques Nacionales Naturales de Colombia—Banco Mundial Global Environment Facility (GEF)—Patromionio Natural, Cartagena de Indias.

Mangel, M. & Levin, P.S. (2005). Regime, phase and paradigm shifts: making community ecology the basic science for fisheries. *Philos. Transact. R. Soc. B: Biol. Sci.*, **360**, 95-105.

McClanahan, T.R., Cinner, J., Graham, N. *et al.* (2009). Identifying reefs of hope and hopeful actions: contextualizing environmental, ecological, and social parameters to respond effectively to climate change. *Conserv. Biol.*, **23**, 662-671.

McClanahan, T.R., Cinner, J., Maina, J. *et al.* (2008). Conservation action in a changing climate. *Conserv. Lett.*, **1**, 53-59.

Mora, C., Andréfouët, S., Costello, M. *et al.* (2006). Coral reefs and the global network of marine protected areas. *Science*, **312**, 1750-1751.

Moreno-Sánchez, R. & Maldonado, J. (2010). Evaluating the role of co-management in improving governance of marine protected areas: an experimental approach in the Colombian Caribbean. *Ecol. Econ*., **69**, 2557–2567.

Norström, A., Nyström, M., Lokrantz, J. & Folke, C. (2009). Alternative states on coral reefs: beyond coral–macroalgal phase shifts. *Mar. Ecol. Prog. Ser.*, **376**, 295-306.

Nyström, M., Folke, C. & Moberg, F. (2000). Coral reef disturbance and resilience in a human-dominated environment. *Trends Ecol. Evol.*, **15**, 413-417.

Obura, D. & Grimsditch, G. (2009). *Resilience Assessment of coral reefs—assessment protocol for coral reefs, focusing on coral bleaching and thermal stress*. IUCN working group on Climate Change and Coral Reefs, IUCN, Gland, Switzerland.

Ostrom, E. (2007). Challenges and growth: the development of the interdisciplinary field of institutional analysis. *J. Institut. Econ.*, **3**, 239-264.

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science*, **325**, 419-422.

Paddack, M.J., Reynolds, J.D., Aguilar, C. *et al.* (2009). Recent region-wide declines in caribbean reef fish abundance. *Curr. Biol.*, **19**, 590-595.

Pollnac, R., Christie, P., Cinner, J.E. *et al.* (2010). Marine reserves as linked social-ecological systems. *PNAS*, **107**, 18262-18265.

Pollnac, R.B., Crawford, B.R. & Gorospe, M.L. (2001). Discovering factors that influence the success of community-based marine protected areas in the Visayas, Philippines. *Ocean Coast. Manage.*, **44**, 683-710.

Porter, J.W. (1972). Patterns of species diversity in Caribbean reef corals. *Ecology*, 745-748.

Ramirez González, A. (2006). *Ecología. Métodos de muestreo y análisis de poblaciones y comunidades*. Editorial Pontificia Universidad Javeriana, Bogotá.

ResilienceAlliance. (2010). *Assessing resilience in social-ecological systems: Workbook for practitioners. Version 2.0.* (No. http://www.resalliance.org/3871.php). Resilience Alliance.

Uthicke, S., Thompson, A. & Schaffelke, B. (2010). Effectiveness of benthic foraminiferal and coral assemblages as water quality indicators on inshore reefs of the Great Barrier Reef, Australia. *Coral Reefs*, **29**, 209-225.

Van Oppen, M.J.H. & Gates, R. (2006). Conservation genetics and the resilience of reef-building corals. *Mol. Ecol.*, **15**, 3863-3883.

Velásquez, J., López Angarita, J. & Sánchez, J.A. (2011). Evaluation of the FORAM index in a case of conservation. *Biodivers. Conserv.*, **20**, 3591-3603.

Vivas-Aguas, L., Tosic, M., Sánchez, J. *et al*. (2010). *Diagnóstico y evaluación de la calidad ambiental marina en el Caribe y Pacífico Colombiano*. Informe Técnico 2010. INVEMAR, Santa Marta.

Walker, B., Gunderson, L., Kinzig, A., Folke, C., Carpenter, S. & Schultz, L. (2006). A handful of heuristics and some propositions for understanding resilience in social-ecological systems. *Ecology and Society*, **11**, 13.

Wilkinson, C. & Souter, D. (2008). *Status of Caribbean coral reefs after bleaching and hurricanes in 2005*. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre, Townsville.

