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Review Paper

When are payment for ecosystems services suitable for coral reef derived coastal protection?: A review of scientific requirements

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ABSTRACT

Payment for Ecosystem Services (PES) is an emerging tool intended to solve a range of ecosystem management inefficiencies, by linking conservation action to payment. Such schemes have not been tested to our knowledge, for coral reef derived coastal protection, which is a key Ecosystem Service (ES) for many nations bordered by tropical coral reefs. Coral health is deteriorating globally, as are their ES and inadequate finance is identified as a cross cutting factor stymieing management action. In this paper, we assessed the feasibility of PES for coastal protection, with a focus on the scientific requirements. Key PES elements related solely to ecological processes were isolated, the role of coral reefs in protecting beaches reviewed and priority management options for improving reef health synthesized. Outputs indicate that there is adequate scientific knowledge to satisfy a PES. While there is limited ability to prove and quantify causality between management actions and ES delivery, PES criteria can be satisfied with the substitution of a management proxy, rather than payments being conditional on ES measurements. Management, both passive and active, would focus on aintaining reefs that already have a protective function and front stable beaches, above a functioning threshold.

1. Introduction

Payment for Ecosystem Services (PES), a tool for managing ecosystems by providing positive incentives for behavioural changes (Bladon et al., 2016) has been touted as "the next best thing" for filling the conservation financing gap (Fujita et al., 2013; Waylen and Julia Martin-Ortega, 2018). The suitability of this scheme for marine application and in particular coastal protection, which is often rated among the most important services provided by coral reefs, is however unclear (Moberg and Folke, 1999; Burke et al., 2008; Mehvar et al., 2018).

PES is based on Ecosystem Service (ES) science, a relatively new field which seeks to link science, economics, conservation management and economic development (Braat and de Groot, 2012). At its simplest, Ecosystem Services are defined as "benefits people obtain from ecosystems" (Millennium Ecosystem Assessment, 2005). Looking at ecosystems through the lens of services provided to humans, allows for their value (economic and intrinsic) to be clearly highlighted. PES then goes a step further and utilizes quantifications of these services, as a base, to devise payments between buyers and sellers of the service. This is once the agreed upon improvements to the flow of services or management of the ecosystem are provided. The link between conservation action, service flow and payment is therefore made clear (Ingram et al., 2014).

Coastal protection in this paper refers to the ability of coral reefs to protect beaches from erosion by absorbing and dispersing significant quantities of wave energy (Kushner et al., 2011; Storlazzi et al., 2019). This attenuation of wave energy allows for reductions in shoreline erosion, flooding, damage to coastal infrastructure and loss of life. The service can be characterised as the amount of attenuation that can be attributed to the reef or to the increase in wave energy due to reef deterioration. Coral health is declining considerably with both local and global stressors working synergistically to negatively impact the ecosystem (De'ath et al., 2012; Jackson et al., 2014) resulting in a diminishing of the service and its value (Mumby et al., 2014; Weijerman et al., 2018).This trajectory is expected to continue (Maynard et al., 2015) with predictions of increasing climate change induced risks to ecosystems (Pachauri et al., 2014).

In spite of the variety of management measures implemented, such as ecosystem based management, integrated coastal zone management,

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marine spatial planning and watershed management (Mcleod et al., 2019) coral reef health continues to decline and inadequate finance has been identified as a cross-cutting factor, undermining conservation action (Bladon et al., 2016; Gill et al., 2017). Private sector financing mechanisms for coral reef conservation are scarce (Pascal et al., 2018a, 2018b; Meyers et al., 2020). However PES might provide a way for conservation funds to be generated from non– public sources (Wunder et al., 2008; Bos et al., 2015) if coastal protection can meet PES requirements.

With no examples found of PES schemes for coastal protection in the peer reviewed literature, our paper aims to fill this knowledge gap and determine if there is adequate scientific knowledge of the provision of the ecosystem service by coral reefs, to develop a PES scheme. This is with the knowledge that social and financial structures also need to be put in place for a PES system to be implemented, and that the ecological parameters provide the foundation on which other elements (e.g., negotiations of agreements, legal structure and financing) are built.

Our focus for PES development is on Caribbean coral reefs, which are considered globally to be among the most threatened (Gardner et al., 2003). At the same time, these reefs are capable of generating huge amounts of revenue from reef associated tourism, estimated at more than USD$7.9 billion (Spalding et al., 2018). Deriving income from this sector for reef protection therefore, seems a logical course of action.

Our objectives are as follows: (i) define PES and outline the key biophysical elements required to develop a scheme; (ii) compile information on the biophysical elements required from coral reefs to provide the ecosystem service; (iii) outline the management measures on ES delivery and (iv) based on these outputs determine if the science behind both the delivery of the ES and management action is adequate for the development of a PES scheme.

2. Methods

We reviewed peer-reviewed journal articles using the online academic search engine SCOPUS (cut off date July 8th, 2020). No geographical or temporal boundaries were set and key word combinations were searched within the title, abstract and keywords. Only papers and their references relevant to our objectives were assessed and additional papers were consulted as required.

Assessment# 1: Sourcing PES schemes for coastal protection. Key words - “payment for ecosystem services” and “coastal protection” and “coral reefs”. We first carried out this search to gain an overall sense of what has been written about PES mechanisms for coral derived coastal protection. SCOPUS – 42 articles were obtained from the search of which only 4 were directly related to developing PES specifically for coral reefs.

Assessment# 2a: Synthesising the science behind the ability of coral reefs to deliver the coastal protection ES. Key words – “coastal protection” and “coral reefs”. 139 results were obtained from the search of which 104 were eliminated as coral reefs were not central to the discussion and/or the ability of reefs to provide the service was only mentioned but not further assessed. The results of the remaining 35 articles were summarised.

Assessment# 2b: The role of live coral in delivery of the service. We added the key word “live coral” to this search. 7 articles resulted, of which 2 were excluded due to lack of direct relevance (e.g., a focus on sea cages and economic valuations). A further 27 relevant papers were found in references related to reef health (e.g., maintenance of carbonate budgets). 32 articles were summarised.

3. Definition and key PES requirements

PES is a market based approach, designed to provide financing for environmental management (Waylen and Julia Martin-Ortega, 2018). The scheme is based on the principle that those who contribute to producing the service (providers) via effective conservation/management action, should be compensated, while those who benefit from the service (beneficiaries), should pay for it.

There is an ongoing debate about what is actually a PES (Muradian et al., 2010; Vatn, 2010; Moros et al., 2020) and therefore a sliding scale of PES definitions. These range from strongly market based (Wunder, 2005) to an overarching term for approaches that provide positive incentives for management of ecosystems (Engel et al., 2008). With a variety of definitions to choose from, there is also some leeway with which to fit coastal protection to PES requirements. The definition used therefore, can depend on the level of specificity obtained by service provision.

In this paper we used definitions of Wunder as both our maximum (Wunder, 2005) and minimum standards (Wunder, 2015). Wunder is acknowledged as an authority on PES, with 6 articles cited more than 8000 times between 2005 and 2020 according to Google Scholar. His 2005 definition is among the earliest and most heavily utilised (Sommerville et al., 2009) which he revisited in 2015. Wundeŕs (2005) definition, “A voluntary transaction whereby a well-defined ES (or actions likely to secure it) is ‘bought’ from at least one ES provider by at least one buyer, if and only if the payment is conditional on provision,” requires robust science. His 2015 definition, “Voluntary transactions (between service users and service providers) that are conditional on agreed rules of natural resource management for generating offsite services”, however allows for some scientific imprecision (inherent in ecological studies) while not being so broad as to eliminate scientific accountability.

In order for participants to demand and make payments for a service, they should know (as clearly as possible) what is being bought and sold, as well as where and how it is delivered (Forest Trends et al., 2008; Fripp, 2014). Therefore, identifying, quantifying and assessing the service is key. This requires: (i) the selection of suitable indicators which are accepted by the scientific community to have impacts on service flow, and can be replicated via reliable methods, (ii) baselines against which success or failure will be measured and, (iii) the definition of spatial boundaries, so that it is clear where the service originates and where it is being delivered. Ecosystem processes underlying service delivery also require identification, as they are crucial elements in designing conservation action to reduce threats.

Conditionality is considered the “conceptual core” of a PES (Bladon et al., 2016) and is a key element separating it from “business-as-usual” schemes (Wunder, 2013; Ingram et al., 2014). The term refers to the requirement for payments to be made only if the stipulated goals are met. These goals can be either measurements of services (via indicators) or management proxies (accepted by the scientific community to have impacts on service flow). The setting of goals is an important element that should be identified early in the process. In many cases, goals are set based on the degree of technical challenges, such as data collection and the ability to quantify the service, and costs (Sommerville et al., 2009). Proving conditionality is difficult and it is often the un-met criterion of PES (Muradian et al., 2010; Lau, 2013) with issues due in part to reliance on continued monitoring from an established baseline.

Additionality is identified as an advantageous but not critical parameter (Wunder, 2005; Muradian et al., 2010). The term refers to the measurement of an interventioń s impact, relative to no intervention being made and translates therefore to the added benefit of having a PES (Tacconi, 2012). Additionality is another difficult parameter to measure, and requires not only the establishment of baselines, but also counterfactual analysis. Such examination allows for comparison of the impact of a scenario in which there was no PES, to a PES situation, in order to prove increased benefits (Wunder, 2005).

In theory, payments are triggered by evidence of service provision or improvement to the service. However, in reality, such results-based payments are difficult to assess and compounded by time lags between intervention and results, and between monitoring and verification. An alternative is the use of a management proxy, with payments being based on evidence of changes to harmful practices or the implementation of actions proven to assist conservation (Atmodjo et al., 2017). The proxy essentially provides an escape clause, especially for situations where service provision cannot be quantified, whether though lack of data, resources or process.

All PES schemes provide incentives to those who own (or are responsible for) specific areas to maintain, restore or enhance ecosystem services (Moros et al., 2020). The rationale behind development of the scheme can however determine which elements are most important, for example in those instances where PES is used to reward conservation action (environmental stewardship), additionality is not paramount (Swallow et al., 2009).

Key requirements for PES schemes are summarised in Table 1.

4. Synthesis of knowledge on the ecological processes of coral reefs involved in coastal protection

Coastal Protection is a complex ES that depends on coral reefs, acting in concert with a number of other biotic and abiotic factors (Fig. 1) (Burke et al., 2008; Elliff and Silva, 2017).

Coral reefs are comprised of thin veneers of live corals, growing in decadal timeframes, on top of massive depositions of their calcium carbonate skeletons (Kuffner and Toth, 2016). Both living and non-living sections are important to coastal protection, as is described in Sections 4.1 to 4.3. Reefs attenuate wave energy, reducing their height, energy and velocity as they move from deeper to shallower waters (Gourlay and Colleter, 2005; Lowe et al., 2007). Morphology across entire reef profiles affects the process (Yao et al., 2019).

Fringing reefs are often responsible for coastal protection (van Zanten et al., 2014) and are also most heavily affected by anthropogenic impacts (Mumby et al., 2014). Their morphology is typically characterized by a forereef slope that terminates at a shallow reef crest and a relatively horizontal reef flat which continues to the coast (Fig. 2) (Yao et al., 2019). Variations in species, geology and hydrodynamic conditions result in high variability between different coral reefs and therefore their effectiveness at providing the service (Quataert et al., 2015).

4.1. Provision of the coastal protection ecosystem service

Both anecdotal and scientific data support the fact that coral reefs protect shorelines (Wells and Ravilious, 2006; Principe et al., 2012; Ferrario et al., 2014). A total of 139 papers were found in the SCOPUS database on this topic, and while only 35 actually assessed the role of the reef in service provision, causality was demonstrated between coral reefs and wave attenuation. The role that coral reefs play in actually delivering the beach protection service was described in 4 of the papers assessed (Wielgus et al., 2010; Kushner et al., 2011; Reguero et al., 2018; Zhao et al., 2019), while the role of important reef processes, such as maintaining the calcium carbonate budget, was not made at all.

The ability of coral reefs to protect beaches is site specific and dependent on the factors indicated in Fig. 1 among others. Enabling factors for service delivery differ from site to site, with some coral reefs having no impact on beach erosion (Quataert et al., 2015). When coral reefs do offer protection, the literature is clear that while the ecosystem is often not the sole reason for sheltered coastlines, coral reefs, particularly, fringing reefs, are major contributors to wave attenuation. In some cases, they can dissipate greater than 90% wave energy (Kench and Brander, 2006; Ferrario et al., 2014).

Table 1
Summary table of key PES requirements.

Key PES Elements	Status
ES Identification	Required
ES Quantification	Required only in absence of proxy
ES Spatial Boundaries	Required
Proven Management Measures/ Proxy	Required
Conditionality	Required
Additionality	Desired

Wide, shallow, rugose reefs are reported as most effective at attenuating wave energy, with reef crests reducing > 80% of incident wave energy (Sheppard et al., 2005; Kench and Brander, 2006; Ruiz de Alegria-Arzaburu et al., 2013). Coral reefs attenuate waves primarily via wave breaking and bottom friction. Dissipation occurs first as waves break on the shallowest section of the reef followed by additional energy loss via friction as the bottom of the wave moves along the reef towards the shore (Koch et al., 2009; Costa et al., 2016). Various reef attributes (Fig. 1) impact both the amount of energy dissipated and its spatial extent, however the two primary reef related factors with major roles in wave attenuation are: (i) reef depth for wave breaking and (ii) roughness of the substrate which causes friction (Gallop et al., 2014; Monismith et al., 2015; Harris et al., 2018).

4.2. The role of live coral

The role of the coral reef structure in providing coastal protection is well reported (Section 4.1) however the input of live coral and hence the impact of degrading health to ecosystem service provision has not been as extensively studied (Ruiz de Alegria-Arzaburu et al., 2013; Ferrario et al., 2014). In this review only 7 of the relevant 139 studies spoke to the role of live corals and of these, 5 examined more closely their role in service delivery. While few, these and other related studies, clearly indicate that healthy reefs with high abundances of scleractinian (hard) corals and structural complexity, provide greater coastal protection than degraded reefs. This is in cases of both frequent - daily erosion (Guannel et al., 2016; Zhao et al., 2019) and rare – storm events (Ferrario et al., 2014; van Zanten et al., 2014). The function of live coral for coastal protection can be encompassed within two interconnected processes: carbonate budgets and structural complexity, which are both enhanced by the presence of the framework builders *Acropora* spp. and *Orbicella* spp.

4.2.1. Carbonate budgets: Reef growth and maintenance

Scleractinian corals are a broad taxonomic and morphological group, which form the foundation taxa of coral reefs (Veron, 2004). Their generation of huge amounts of calcium carbonate skeleton is crucial to the provision of coastal protection (Guannel et al., 2016). Reef defence functions can only be maintained naturally, if vertical reef accretion allows for the shallow depths required for wave attenuation (Waterman, 2008; Beetham et al., 2017; Perry et al., 2018). Within this group, specific species play dominant roles as reef builders and therefore in the delivery of coastal protection.

Corals grow by accreting calcium carbonate and at the same time erosion (biological and physical) of the skeleton produces sand. For the structure to be maintained, the carbonate budget must be positive (i.e., the rate of growth must exceed that of erosion) (Ryan et al., 2019). If the system switches to a net negative state however (as is caused by large scale coral mortality for example), net erosion can ensue, resulting in deterioration of the reef structure over time (Perry et al., 2013), flattening of the reef (Alvarez-Filip et al., 2009) and a reduction in the structurés ability to attenuate wave energy (Sheppard et al., 2005).

Hard coral cover is a predictor of carbonate production, with the abundance of historical, framework builders such *Acropora* spp. and *Orbicella* spp being especially important to the maintenance of carbonate budgets (Perry et al., 2018; Estrada-Saldívar et al., 2019). A significant reshaping of Caribbean coral reefs has already taken place, with a decline in reef builders and an influx of "weedy" species such as *Agaricia* spp. and *Porites astreoides* colonising their spaces. Such corals have neither fast growth rates nor the ability to generate the quantities of carbonate required for significant reef building (Pandolfi and Jackson, 2006; Alvarez-Filip et al., 2013; Perry et al., 2013). Major events, such as these have been reported as primary reasons for the shifts in carbonate budgets in the Caribbean, from strongly net positive (5 kg $CaCO_3$ $m^{-2}y^{-1}$)

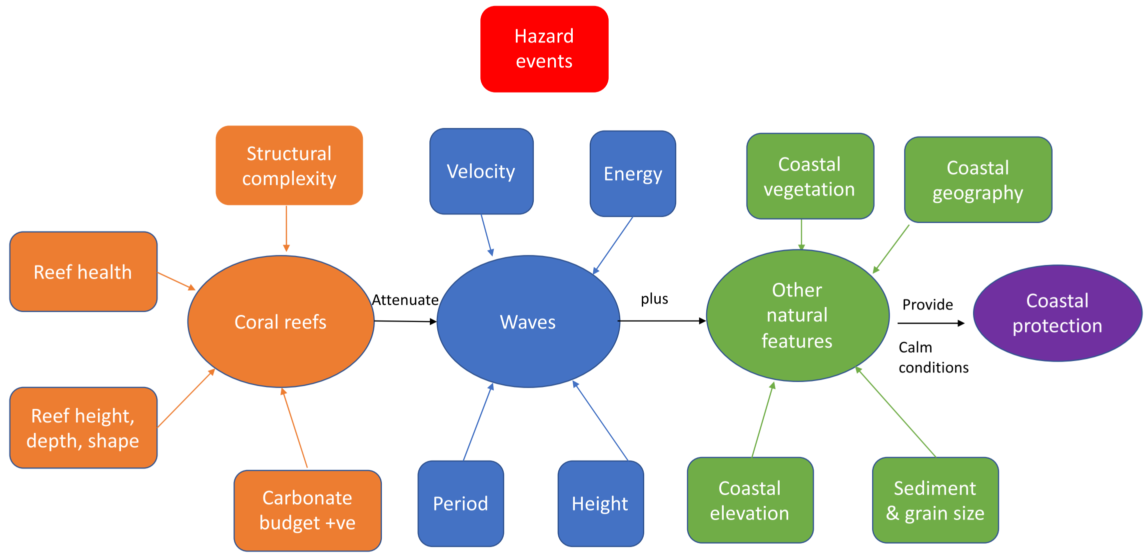

Fig. 1. Natural features that interact to deliver coastal protection. Shapes in orange represent coral reefs and their different ecological parameters. Shapes in blue represent waves and their physical processes. Shapes in green represent other biophysical features that can impact service provision. Red represents natural hazards that impact all natural processes.

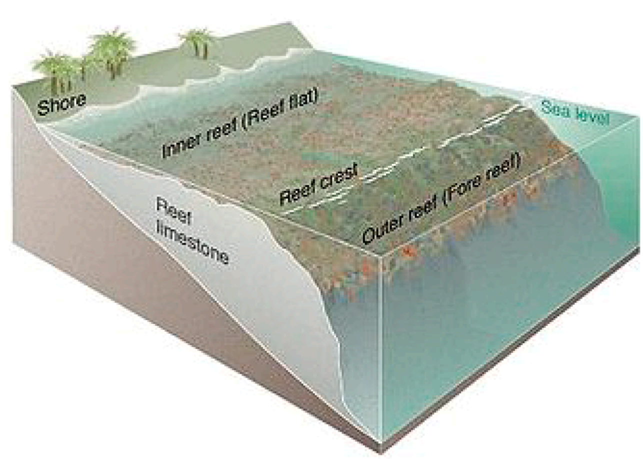

Fig. 2. Cross section of a fringing reef. Image from (U.S. Geological Survey Fact Sheet 025-02). Retrieved September 16th, 2020.

to less so (2.6 kg $CaCO_3$ $m^{-2}y^{-1}$) between the 1960s and 1990s (Kennedy et al., 2013).

Maintenance of the carbonate budget is therefore essential to sustaining reef function, including coastal protection (Lange et al., 2020) and the presence of live coral crucial. Timescales for this deterioration/erosion of reefs are not well understood (World Bank, 2016) and while a figure of 6 mm per year has been suggested as an approximation of the rate by which dead reefs will erode, it is also acknowledged that rates differ dramatically between reefs and even at different locations on the same reef (Hutchings, 1986; Eakin, 1996). For example, on Australia's Great Barrier Reef, dense coral colonies in some inshore areas were lost over 50–100 years (Hoegh-Guldberg et al., 2007), while Uva reef in Panama, was reduced to almost the same level as the surrounding sediment in approximately 15 years (Eakin, 2001). In the Indian Ocean, bioerosion on Chagos reef reduced the structure to rubble within 3 years (Sheppard et al., 2002).

A shift into a net negative phase is reported for situations where live coral cover on Caribbean reefs reaches 10% (Perry et al., 2013). In a more recent study, this concept was refined further to 10% cover of structurally important species only and considered as precautionary threshold (Darling et al., 2019). Nonetheless, these are targets that should be aimed for.

4.2.2. Structural complexity

Structural or topographical complexity of reefs refers to their three-dimensional form or layout on all spatial scales (Zawada and Brock, 2009), and plays an important role in wave dissipation (Lugo-Fernandez et al., 1998). Complexity is primarily defined by the morpho-functional characteristics (size and shape) of dominant and foundation corals (Veron, 2002; Alvarez-Filip et al., 2011). It is measured via rugosity (on one spatial scale) and roughness (range of spatial scales) (Zawada and Brock, 2009) and so both terms are used.

In terms of wave attenuation, live or just dead corals provide roughness, which reduces wave energy (Harris et al., 2018; Reguero et al., 2018). Roughness is influenced by the type and size of substrate, with sand offering little and large, branching coral creating the most friction and therefore the greatest impact on wave attenuation (Sheppard et al., 2005; van Zanten et al., 2014). Reefs can be categorized according to their rugosity index, with the flattest reefs having an index of less than 1.5. Approximately 75% of Caribbean reefs fall into this category, with reefs exhibiting an index higher than 2 (complex reefs) being extremely rare. (Alvarez-Filip et al., 2009).

While structural complexity increases with coral cover and species richness, massive growth forms (e.g., *Orbicella* spp) and fast growers (e.g., *Acropora* spp.) are thought to contribute the most to the structure (Alvarez-Filip et al., 2011; Kuffner and Toth, 2016) and hence to wave dissipation. A significant decline in coral cover and hence structural complexity results in crumbling of the reef and a change from a more varied topographical surface to a flatter surface, with less ability to reduce wave heights (Alvarez-Filip et al., 2009; Osorio-Cano et al., 2019).

4.3. Summary of knowledge on service provision and the role of live coral for coastal protection

The importance of living reefs and therefore reef health to service delivery is shown in Fig. 3. Living reefs contribute to service delivery in the short term via wave attenuation and in the long term via the ability of the reef to grow (accrete) and therefore maintain their structures. Reef degradation affects both carbonate production and structural complexity (Alvarez-Filip et al., 2011; Perry et al., 2013). Reef requirements can therefore be summarised as healthy reefs, dominated by

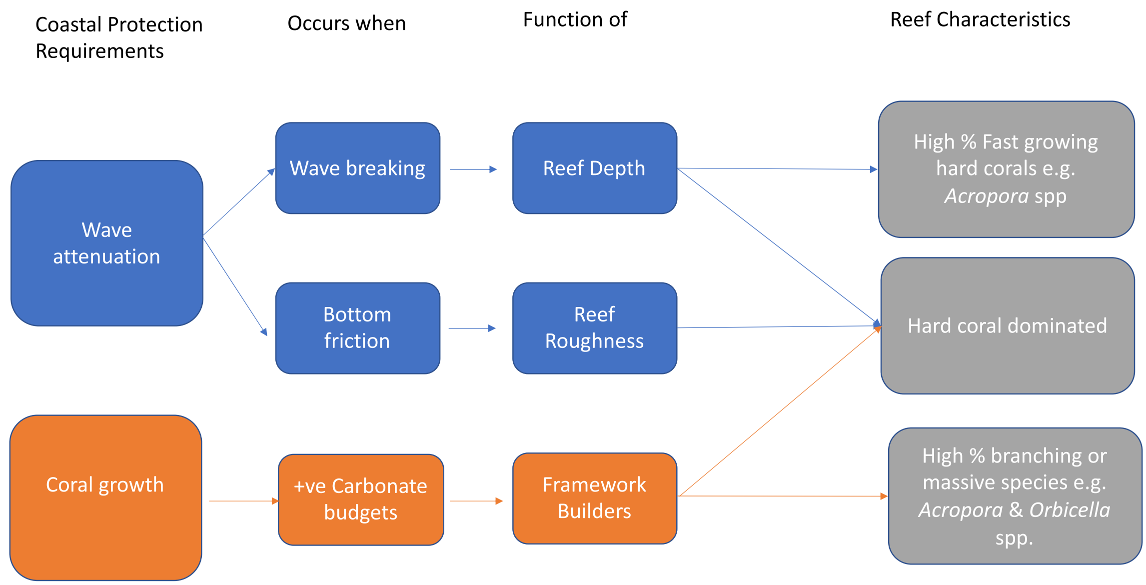

Fig. 3. Summary of key coastal protection requirements and primary coral reef characteristics for their provision.

framework builders. These attributes are interlinked, as the same coral species, (i.e., large framework builders) are largely responsible for both (Graham and Nash, 2013; Harris et al., 2018).

5. Synthesis of knowledge on management for coastal protection

Coral reefs are extremely complex ecosystems with high levels of genetic, species and habitat diversity, leading to a vast variety of interactions involving many different species (Moberg and Folke, 1999; Dikou, 2010). The major stressors are well known, however understanding ecosystem responses to them can be challenging, due to this complexity (Pandolfi, 2015; Steneck et al., 2018). It is however clear that effective reef protection will require management that can mitigate the effects of at least the dominant stressors, under existing and future scenarios of global climate change (Weijerman et al., 2018; Brandl et al., 2019). Coral reefs have been described as the most studied marine ecosystems with scientific consensus on the range of stressors and appropriate management measures (Mumby and Steneck, 2008). The aforementioned authors carried out a comprehensive review of coral reef management and conservation and no attempt was made here to conduct yet another. We used this as our seminal paper, and also referred to relevant references and more current research for specific elements.

Management can be categorised as either passive or active. Passive management operates on the basis of allowing nature to heal itself with limited human interference (e.g., reducing impacts, enforcement) while active management takes the form of human intervention, such as in the growing and planting of coral (Rinkevich, 2008). Knowledge of the way in which the trophic structure, biodiversity resistance (to impacts) and resilience (ability to recover from impact) of coral reefs respond to human impact, is key to determining conservation strategies (Bellwood et al., 2004; Côté and Darling, 2010). Management action must encompass both ecological parameters and social enabling factors, working in tandem to be effective (Gill et al., 2017), however in the paper our focus is on the ecological characteristics.

Coral reefs are impacted by a range of local and global stressors. Local management has however been reported as providing a buffering effect on coral degradation under climate change scenarios (Hughes, 2003; Weijerman et al., 2018; Beatty et al., 2019). Dominant, local stressors on Caribbean reefs, as well as primary impacts and management strategies, are outlined in Table 2. The most significant impact is considered to be macroalgal abundance caused by anthropogenic nutrients, and unsustainable fishing (Jackson et al., 2014; Harborne et al., 2017; Bruno et al., 2019). Excessive levels of macroalgae, are responsible for overgrowing and shading adult corals, inhibiting recruitment of juveniles corals and harbouring disease (Idjadi et al., 2010; Rasher and Hay, 2010), which promote conditions for declining reef health. Key here is that a reef in a negative feedback loop, might be unable to recover even if the disturbance is removed, as processes drive the system towards macroalgal abundance (Mumby et al., 2014). Action aimed at

Table 2
Prominent documented local stressors, impacts and passive management action on Caribbean coral reefs.

Reef Threat	Primary Impact	Management Action
Eutrophication[a]		
Agriculture	Macroalgae	Watershed Management
Sewage	Macroalgae	Treatment Plants, Watershed Management
Hurricanes[b]		
Industry	Outright Mortality	Policy - Water Quality Standards
Coastal Construction	Outright Mortality	Policy - Integrated Coastal Zone Management
Unsustainable Fishing[c]	Macroalgae	MPAs - no take zones
	Outright mortality	
Invasive Species[d]		
Lionfish	Trophic Pathway disruption	Culling
Harmful Orgs. Ballast Water	Disease	Policy - Ballast Water Treatment Protocols
Disease[e]		
of Keystone Species	Macroalgae	
of Corals	Outright Mortality	Removal of diseased colonies Barriers to disease progression Application of chemicals to kill microbes
Physical Damage[f]		
Anchors	Outright Mortality	Permanent Moorings
Divers		ELE*
Hurricanes		Rapid Repair

[*] ELE - Education, Legislation and Enforcement are cross cutting management actions and are only indicated when they are the only action identified.
[a] (Weil and Rogers, 2011; Mumby et al., 2014) [b](Maragos, 1993; Richmond, 1993), [c](Sandin et al., 2008; Jackson et al., 2014), [d](Green et al., 2012; Galil et al., 2019), [e](Hunte and Younglao, 1988; Sutherland and Ritchie, 2004), [f] (Lewis, 1984; Hawkins and Roberts, 1994; Barker and Roberts, 2004).

Table 3
Examples of ecosystem measures and indicators used to quantify coastal protection.

Measures	Indicators	Citation
Biophysical Processes	**Physical**	Harris et al., 2018, Yao et al., 2019
	Wave energy, height, velocity	
	Biological	
	Reef rugosity	Monismith et al., 2015, Lowe et al., 2007
	Fish abundance	Wainger and Boyd, 2009
Ecosystem Service	Beach Erosion	Reguero et al., 2018, Kushner et al., 2011
	Coastal Inundation	Beck et al., 2018, Ferrario et al., 2014
Socioeconomic	Avoided damages	Storlazzi et al., 2019, Van Zanten et al., 2014
	Property values	Burke et al., 2008, Pascal et al., 2016
	Breakwater replacement costs	Ferrario et al., 2014, Beck et al., 2018

reducing macroalgal growth, is therefore one of the crucial factors to be controlled in conservation, and managing herbivores and improving water quality are priority conservation actions (Mumby and Steneck, 2008; Jackson et al., 2014; Steneck et al., 2019).

It is important to note that to date, no conservation or restoration management measures have resulted in reefs recovering to pre-stress state (Mumby et al., 2014; McWilliam et al., 2020). However, strategies aimed at reducing some key stressors have yielded specific successes, that have aided in recovery of reefs, as is outlined in the remaining sections.

5.1. *Passive management: Increase herbivory*

Herbivory is a critical trophic interaction on tropical coral reefs, and declines in herbivore abundance, result in a proliferation of macroalgae (Hunte and Younglao, 1988; Ladd and Shantz, 2020). Increasing the abundance of herbivores leads to reduced macroalgal cover via trophic cascades, or top down control (Hughes et al., 2007; Burkepile and Hay, 2008; Steneck et al., 2019; McClure et al., 2020), which is expected to eventually result in increased coral cover. While this final step has not been strongly made, research by Jackson et al. (2014) recorded a significantly higher abundance of coral on Caribbean reefs with more parrotfish (*Scaridae* spp.). Additionally, research from both Australia (Hughes et al., 2007) and the Bahamas (Mumby and Harborne, 2010) provided some evidence of increasing numbers of herbivores supporting reef health by increasing coral recruitment. While coral reef recovery to pre-stress levels was not observed in these studies, coral recruitment is the first step in such a recovery. It is important to note though, that the dominant recruits reported in the Bahamas were *Porites astreoides* and *Agaricia* spp, which, as was shown in Section 4.2.1. cannot provide the same ecosystem services as framework builders.

Management of herbivores is commonly dealt with within an MPAs framework (Section 5.1.4) and/or via a complete ban of primary herbivores such as parrotfish in Belize (Cox, 2014) and Bermuda (O'Farrell et al., 2016).

5.2. *Passive management: Reducing land based sources of marine pollution*

Activities on land produce excessive flows of nutrients (primarily nitrogen and phosphorous) and sediments onto coral reefs, which have proven detrimental to their health (Bellwood et al., 2004; Burke et al., 2011; Eberhard et al., 2017). Elevated nutrient levels from fertilizer and sewage (both humans and farm animals) promote the growth of macroalgae on small scales, such as the west coast of Barbados (Tomascik and Sander, 1985) and large scales, such as the entire Great Barrier Reef of Australia (De'ath and Fabricius, 2010). Coastal development and riverine run off also produce elevated quantities of sedimentation, which interrupt coral processes (such as feeding), leading to mortality and/or reduced growth (Weber et al., 2012; Bartley et al., 2014). Coral reefs with excessive sedimentation have been shown to have reduced diversity, lower coral abundance and lower accretion rates (Rogers, 1990). Improving water quality around reefs is troublesome, as the sources often originate on land and are some distance away from where impact occurs on coral reefs. However, holistic management frameworks, which incorporate actions taken from the source of pollution to impact sites have been demonstrated to improve water quality around reefs. Land management measures that reduce nutrients and sediments (e.g., agricultural practices and watershed restoration) (Fillols et al., 2020) have resulted in water quality improvements around reef areas, which have been further demonstrated to improve coral cover (Jokiel and Brown, 2004; Fabricius et al., 2014; Shelton III and Richmond, 2016), even if not to pre-stress levels (Wenger et al., 2017).

5.3. *Active management: Reef restoration*

Coral restoration's importance is emerging with the reported inability of passive management measures to restore reefs to pre-stress levels (Rinkevich, 2008). This active management tool focuses on repairing reefs in order to facilitate their recovery and restore ecosystem integrity (Basconi et al., 2020). It often takes the form of coral gardening which is divided into 2 activities: nursery phase for rearing coral recruits and out-planting to the reef (Boström-Einarsson et al., 2018). Of specific interest to coastal protection, is that the fast growing framework builder (*Acropora* spp.) and massive, framework builder (*Orbicella* spp) are the two species primarily selected for restoration, with survival rates of > 60% being reported (Boström-Einarsson et al., 2020).

While there are few reports of active reef restoration resulting in long term ecological recovery (Fox et al., 2019) and concerns raised over survival and fitness of transplants and cost effectiveness of operations (Bayraktarov et al., 2016); there have been many promising advances

Table 4
Summary of the ability of coastal protection to meet PES requirements – ES identification, quantification and geographical boundaries.

PES Requirement	Possible Coastal Protection Parameters	Measurement	Status	Explanatory Notes
ES Identification & Quantification	Wave height	m	**C**	Back reefs wave heights are important indicators of wave energy
	Reef state – 1. Hard coral abundance	% cover	**C**	Health of reef, related to its ability to attenuate wave energy and to accrete
	Reef state – 2. Carbonate budgets	kg $CaCO_3 m^{-2} yr^{-1}$	**C**	Indicator of the rate at which the reef produces and accumulates $CaCO_3$
	Reef state – 3. Parrotfish density	#/m^2	**C**	Keystone species on Caribbean reefs and indicators of overall health
	Beach erosion	m/yr	**C**	Sand lost from the beach due to wave energy
Geographical boundary	Area of beach protected	m	**C**	Index methods, numerical and physical models can indicate spatial boundaries

Key: N – Not Documented, E – Expected result but confirmed by less than 5 studies and C – Confirmed.

made within the last decade, such as selective breeding and assisted evolution (Baums et al., 2010; Drury and Lirman, 2017; Basconi et al., 2020) that are cause for optimism. Restoration of marine ecosystems is still in its nascent phase, however it has been predicted to be the most dominant discipline in environmental science in the 21st century (Hobbs and Harris, 2001), and there have been increased calls for active restoration to be added to the tools of watershed and fisheries management for coral reefs (Rinkevich, 2008; Basconi et al., 2020; Boström-Einarsson et al., 2020). Reef Restoration has therefore been included as a priority conservation action.

5.4. Management proxy: Marine protected areas

Reef management within an Marine Protected Area (MPA)[1] framework, is one of the most extensively used tools for conservation (Toropova et al., 2010; Claudet et al., 2011). MPAs work by managing human activity, within specific boundaries, with the expectation that with reduced impact, recovery will occur (Day et al., 2012). With effective management, which encompasses: adequate compliance, participatory decision making, empowerment and education of local communities (Hughes et al., 2010) MPAs can play important roles in restoring ecosystem structure and function (Mumby and Steneck, 2008; Laffoley et al., 2019).

Well managed MPAs have proven effective in the reduction of many stressors. They are responsible for: reducing unsustainable fishing (Bellwood et al., 2004) leading to increases in the size and biomass of fish species (Johnson and Sandell, 2014; Sciberras et al., 2015; Leenhardt et al., 2017); improving herbivory (Selig and Bruno, 2010; Possingham et al., 2015) and promoting coral recovery after disturbances (Mumby and Harborne, 2010; Perry et al., 2013). Importantly, reefs with protection for parrotfish, have also demonstrated the ability to delay loss of architectural complexity, which is a key component of coastal erosion (Bozec et al., 2015). By reducing some effects of local stressors, MPA's have also been shown to buffer global impacts (e.g., high temperatures). Further to the passive management measures outlined, active management, such as reef restoration, is recommended to be carried out within effectively managed MPAs, where efforts are most likely to result in success (Shaver and Silliman, 2017; Basconi et al., 2020).

MPAs are however not "silver bullets" and even when well-managed, have not always resulted in increases in coral cover (Cox et al., 2017). Most MPAs are susceptible to impacts that originate outside of their boundaries such as anthropogenic pollutants (e.g., sewage, heavy metals), invasive species and disease (Hughes et al., 2010). Mumby and Steneck (2008) in an extensive review of the causes and consequences of reef decline, documented the status of knowledge of the ability of MPAs to meet management goals. They found clear successes in terms of increasing fish populations within the protected spaces, however the expected results of increased coral recruitment and hard coral abundance were less widely documented. Modelling has shown that with an objective of increasing coral cover (as is required for coastal protection) reduction of land based sources of pollution is more effective than on site MPA activities (Weijerman et al., 2018). MPA action therefore would have to encompass land based sources of pollution and be part of a broader programme of Watershed Management within an Integrated Coastal Zone Management (ICZM)[2] (Belfiore et al., 2004; Cicin-Sain and Belfiore, 2005). This is demonstrated on the Great Barrier Reef where an important policy and management focus for marine park managers is on nutrients and pesticides from agricultural practices outside of their boundaries (Kroon et al., 2016).

Increasing coral reef health resilience within an MPA framework that includes watershed management can therefore can be taken as a proxy; a means of ensuring the likelihood of the continuance of the ES by reducing threats to coral reefs (World Bank, 2016; Mcleod et al., 2019).

5.5. Summary of knowledge on management for coastal protection

Management for coastal protection, equates to providing a hard coral reef, dominated by framework builders. Priority measures as well as the primary impact and management frameworks are outlined in Fig. 4.

6. Coastal protection and PES

To date PES has not been extensively considered for coral reef coastal protection. Only 4 peer reviewed articles were found, in which PES for coastal protection was considered, and in none of these was a PES scheme actually developed (Lau, 2013; Castaño-Isaza et al., 2015; Pascal et al., 2016; Elliff and Silva, 2017). One report was identified in which a PES scheme for coastal protection in San Andres, Colombia was considered (Lau, 2012), however this never came to fruition due to the departure of one of the key local stakeholders (T. Agardy 2019, pers. comm.).

The ability of coral reefs to meet each PES requirement (Section 3) is discussed in this section. Requirements are as follows: the ES must be identified, spatial boundaries defined, management measures linked to specific outcomes known and conditionality satisfied. It is important to note that if quantification of the service is not possible, a management proxy can be used to satisfy conditionality. Additionality is desired and is discussed with conditionality, as its basic requirements are the same.

The ability of coral reefs to meet each PES requirement is described and displayed via the use of Tables 5–7. Analyses were based on outputs of the syntheses of knowledge (Sections 4 and 5). Measurement Indicators were categorised as: N – Not Documented (No assessed papers documented the output); E – Expected result, but confirmed by less than 5 studies and C – Confirmed. The Management Proxy status (Section 6.3) was based on knowledge of the ability of the identified system to produce the desired output.

6.1. ES identification, quantification and spatial boundaries

Coastal Protection can be defined and measured by different indicators, which include: the biophysical processes responsible for providing the service, the service itself and the social benefits derived by humans (Guerry et al., 2015). Principie et al. (2012) quantified at least 57 different indicators that can be clearly calculated and replicated (as is required for PES) and a sub set of those deemed most relevant is shown in Table 3.

Indicators of measurement vary in their direct relevance to human well-being (Wainger and Boyd, 2009), with biophysical processes requiring some means of translation into either ecosystem service or socioeconomic measurements, for the benefits to humans to be made clear. Storlazzi et al. (2019) for example used wave processes and reef health indicators (biophysical processes), to determine the protection offered by coral reefs against flooding (ecosystem service measurement). This was then calculated in terms of avoided damages (socioeconomic measurement). Here, we consider only the biophysical elements - processes and the service as outlined in our objectives.

The measurements of indicators for the biophysical processes (e.g., wave heights and hard coral abundance) and the service (beach erosion) are well established in literature (Principe et al., 2012). The height of waves in back reefs for example is indicative of wave energy which will impact the shoreline. The measure is widely used in engineering studies which examine the impact of reefs on shorelines

(Ferrario et al., 2014; Quataert et al., 2015; Guannel et al., 2016;

[1] An all-encompassing term that includes reserves and multiuse zones

[2] ICZM is a resource management system following an integrative, holistic approach and an interactive planning process in addressing the complex management issues in the coastal area Thia-Eng, C. (1993). Essential elements of integrated coastal zone management. Ocean & Coastal Management **21** (1–3), 81–108.

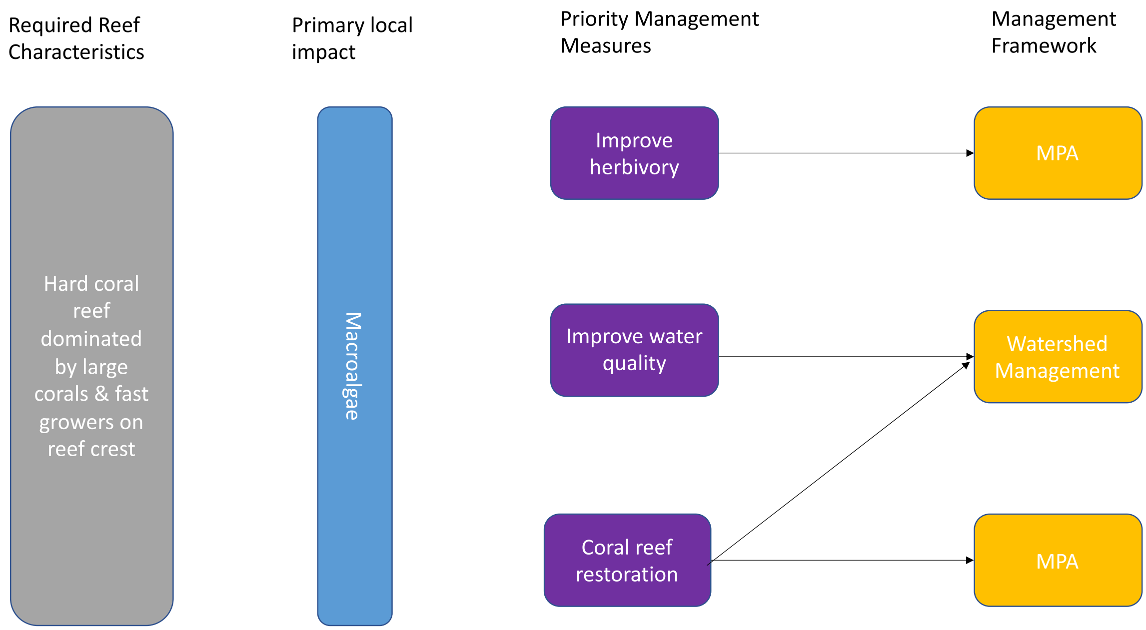

Fig. 4. Summary of primary impact, priority management measures and management frameworks to deliver positive outcomes for coastal protection.

Table 5
Table of management frameworks for priority management action and indications of success for improving coral health.

PES Requirement		Management Framework	Status	Explanatory Notes
Management measures required for increased abundance hard corals & framework builders	Improve herbivory	MPA	E	Increased numbers of parrotfish is documented in MPAs and linked in a few cases to minimal increases in hard coral
	Reef restoration	MPA	E	Methods for gardening *Acropora* spp and *Orbicella* spp have proven successful in increasing hard coral abundance. Issues remain with post-transplantation reproductive ability and survivability.
	Improve water quality	Watershed	E	Reducing terrestrial nutrients is documented in few cases to increase hard coral abundance

Key: N – Not Documented, E – Expected result but confirmed by less than 5 studies and C – Confirmed.

Storlazzi et al., 2019).

Coral reef health is most often indicated by the variable of hard coral cover (Obura et al., 2019). This metric, as well as hard coral species diversity, fish abundance and fish diversity are standard parameters in many established monitoring programmes such as the Global Coral Reef Monitoring Programme (GCRMP) (Hill and Wilkinson, 2004); the Atlantic and Gulf Rapid Reef Assessment (AGRRA) (Lang et al., 2010) and Reef Check (Reef Check Instruction Manual, 2006). Rugosity, the more commonly utilised metric for structural complexity in coral reefs, is less often included in monitoring programmes but there are clear methods established for quantification, ranging from low tech "chain and tape" to remote sensing (Figueira et al., 2015). This parameter, as well as the previously mentioned hard coral and fish metrics, are identified as key indicators in determining reef health (Flower et al., 2017) and as important in the determination of coastal protection (Beck et al., 2018). The carbonate budget is not a commonly measured metric, but could be used as a proxy measurement for maintenance and reef function is carbonate budgets (Lange et al., 2020).

Defining boundaries is not simple for coastal protection, due in part to the diffuse and interconnected nature of water (Bladon et al., 2016). Additionally, benefits are provided and impacts can originate some distance away from the coral reef, which increases this complexity. However, while no standard methods exist for determining where the service is provided, (i.e., what reefs protect what beaches/properties/lives) (van Zanten et al., 2014), the literature is replete with examples of methods used. They include both high and low tech, at different levels of specificity and range from complex simulation models (Reguero et al., 2018; Storlazzi et al., 2019) to a simple reliance on the distance from the reef to the shore (Burke et al., 2008; Silver et al., 2019).

A summary of the ability of coastal protection to meet the first three PES requirements – ES identification, quantification and spatial boundaries is provided in Table 4.

6.2. Management measures linked to specific outcomes and MPA proxy

Restoring coral reefs will not occur with a single management tool. Rather, a combination of marine and terrestrial management actions, as is possible within an ICZM[3] framework, and coupled with a reduction in carbon dioxide emissions (Weijerman et al., 2018). Priority management measures aimed at delivering increased hard coral cover, were identified as, improving herbivory, water quality and reef restoration (Section 5). However, the ability of these measures to increase and sustain hard coral cover populations, while expected, have not been

[3] ICZM is a resource management system that includes a participatory and holistic approach to addressing the complex coastal management issues ibid., Saffache, P. and P. Angelelli (2010). Integrated Coastal Zone Management in small islands: A comparative outline of some islands of the Lesser Antilles. Revista de Gestão Costeira Integrada-Journal of Integrated Coastal Zone Management **10** (3), 255–279.

Table 6
Coastal protection indicators, possible conditions and an assessment of the ability of priority management actions to be able to meet each condition.

Indicators		Possible Conditions	Management Actions				Explanatory Notes
			H	WQ	AR	MPA	
Measurements	1. Physical	(a) Wave heights do not exceed baseline under normal conditions.	N	N	N	N	No documented relationship between passive management actions and condition. Active Restoration is expected to ultimately result in reduced wave heights, but has not been demonstrated.
	2. Biological	(a) Hard coral cover is equal to or exceeds 10%.	E	E	E	E	Few cases documented where management actions result in increases in hard coral cover and not to 10%.
		(b) Framework builder cover is equal to or exceeds 10%.	N	N	E	N	No cases documented where passive management actions result in increases in framework builders and not to 10%. Active Restoration has been documented to increase abundance of *Acropora* spp. (framework builders and fast growers). However, not to 10% and additional concerns re survival and fitness in the long term.
		(c) Carbonate budget is positive.	N	N	N	N	Expected that improving conditions for reef health will result in positive carbonate budgets, but not documented for management actions.
		(d)Rugosity index is equal to or higher than 2.	N	N	N	N	Expected that improving conditions will result in higher rugosity index, but not documented for management actions.
		(e) Parrotfish abundance is equal to or greater than baseline.	C	N	N	C	H & MPA - Measures aimed at reducing fishing pressure of parrotfish both within and out side MPAs have been documented to result in increased abundance of these herbivores. AR & WQ - Improvements to water quality and active restoration have no documented impacts on parrotfish abundance.
	3. Service	(a) Beach widths remain at baseline.	N	N	N	N	Expected that improving conditions for reef health will result in stable beaches, but not documented for management actions.
Proxy	4. MPA	METT management efficiency score of 80% or greater.	C	C	C	C	All management actions carried out will increase METT scores.

Management Actions Key: H – Improving herbivory, WQ – Improving water quality, AR – active restoration, MPA – A variety of management actions carried out and linked to watershed management; N – Not Documented (No papers documented the output), E – Expected result but confirmed by less than 5 studies and C – Confirmed.

convincingly demonstrated in the literature (Table 5).

6.3. Conditionality and additionality

In PES schemes, payments should be based on either service flows or conservation action. If the outcomes identified by the conditions are unmet, then payments are not triggered. This further implies that the effect of conservation action must be observed in the short term and during the life of the PES.

In order for buyers to ensure that the environmental actions they have paid for have: (i) the desired impact (conditionality) and (ii) a beneficial impact greater than what is occurring at present (additionality), there needs to be some form of verification of the sellerś actions and impacts. Therefore, baseline conditions, measured by specific indicators that can be replicated are required (Lau, 2013) and these were identified in Section 6.1. Proving both conditionality and additionality will depend on the indicators chosen and how well these are monitored. The difficulty here appears to be more related to the practical issues (such as expense) involved in establishing baselines and monitoring over time (Bladon et al., 2016), rather than the ability to measure. As the ecological requirements for additionality are the same for those of conditionality, only the latter will be considered in the assessment.

Table 6 lays out some indicators of coastal protection (identified in Section 6.1). For each indicator, examples of conditions and an indication of the ability of priority management actions (Section 5) to deliver these outcomes, are identified. Outputs are based on conclusions/evidence drawn from Sections 4 and 5. For example, the condition for 1a is that wave heights do not exceed baseline measurements. There are no indications in the literature that increasing herbivory (H) will result in this effect, hence not documented (N) is the output.

6.3.1. Conditionality via service flows

For measurements of service flow, the only condition that can be fully met by the management measures outlined is 2e - Parrotfish abundance is equal to or greater than the baseline. Actions aimed at improving herbivory both inside and outside of MPAs have been demonstrated to increase parrotfish abundance. For Conditions 2a and 2b, management actions have been documented to increase hard coral cover and structural builders, however these studies are few and increases have not been recorded to pre-stress levels (or to 10%). Neither physical nor service indicators measurements can meet the conditionality requirement, as none of the aforementioned management actions will have a foreseeable impact on wave heights or beach width in the short term. Therefore, these measures are not suitable for PES.

6.3.2. Conditionality via environmental action/proxy

Using MPAs as a management proxy of environmental action is the strongest means of fulfilling conditionality. A system of monitoring the effectiveness of MPAs such as the Management Effectiveness Tracking Tool (METT) can be used. This tracking tool allows one to monitor progress in improving management effectiveness via a system of scoring (Stolton et al., 2007). Priority management actions and stipulations such as being linked to broader Watershed or ICZM programmes, can be appended.

6.4. Other PES considerations

In addition to meeting the aforementioned requirements, the suitability of PES will be driven by the specific situation. Knowledge on whether the specified beach is eroding or stable is important. For an eroding beach, even if the situation is due to coral decline, no existing management measures can ensure that corals grow to attain the adequate depth and rugosity required to attenuate waves and ensure stable beaches in the short term. In such cases, most hoteliers opt for the more established grey infrastructure such as breakwaters (Silver et al., 2019). For a stable beach, however, where a decline in reef health can cause a concomitant decline in service provision (Kushner et al., 2011; Monismith et al., 2015), payments can be made for better management, aimed at maintaining or improving services from an existing healthy reef (Beck et al., 2018; Iyer et al., 2018). Causality has been demonstrated in the literature between a toolbox of conservation action (as is carried out within MPA frameworks) and some improved reef health parameters and between reef health and greater beach protection. This is therefore a more feasible option. A PES scheme, with a goal of rewarding such environmental stewardship allows for potential PES schemes to be developed.

Social parameters must also be taken into consideration and while not being the focus of this paper, knowledge of potential beneficiaries (buyers) and providers (sellers) of the ES are required to showcase an example of a PES scheme. A variety of stakeholders would benefit from coastal protection, however, for most Caribbean Islands with the importance of tourism and its strong relationship to coastal protection, the most obvious beneficiaries are coastal hoteliers, who desire beach presence as a selling point for their businesses (Uyarra et al., 2005; Pascal et al., 2018a, 2018b), as well as governments. Sellers of the service for the MPA Proxy, will most likely be MPA Managers, who, while not owning the marine space, are designated by governments to manage them.

A PES could be carried out with a hotelier as the buyer and MPA managers as sellers of a range of environmental actions (both ecological and social) that would enhance beach stability (Fig. 5). Payments could be conditional on the measurement of indicators, such as achieving 10% hard coral cover, or delivery of an MPA workplan, or METT scores. The goal must be clear; forestalling potential loss in revenue from beach erosion, by investing in environmental stewardship of reefs that actively protect beaches.

7. Conclusions

PES is a multifaceted approach, with specific ecological requirements, which can be met by the coastal protection ecosystem service.

An analysis of literature, indicates that the scientific knowledge behind the ability of coral reefs to provide coastal protection, is adequate for the development of a PES scheme, under certain conditions. The service is complex, with different factors, both reef and non-reef related, contributing to reducing beach erosion. Not all fringing reefs protect beaches, but some, within specific contexts, are largely responsible for wave attenuation and therefore can play major roles in reducing beach erosion. Studies measuring the impact of coral reef parameters on waves are well reported. However, those demonstrating causality between coral health parameters (e.g., coral cover, rugosity, fish abundance) and the service itself (e.g., decreased erosion) are rarer, possibly due to the complexity of coral reefs and coastal processes (Reguero et al., 2018).

A range of methods exist in the literature to identify, quantify and define geographical boundaries of coastal protection delivery, as required for a PES. Our limited ability to prove causality between management action and ecosystem service delivery causes some difficulty, though not insurmountable, in fulfilling the conditionality requirement. In some very specific cases, such as increasing the number of herbivores, management measures have been shown to be successful for coral reef health and a PES can be designed around this. However, since increasing herbivore abundance has not been strongly demonstrated to have impacts on beach erosion, it is highly unlikely that a PES would be successful. Instead, a PES scheme can be developed with payments triggered by specified conservation action or management proxy.

MPAs cannot solve all issues, however, once effectively managed, they provide a framework within which a range of management measures can be effectively carried out. This toolbox of measures has been demonstrated to improve coral reef health and increase hard coral cover under some conditions. Such interventions therefore remain our best tool at improving overall reef health (Roberts et al., 2017; Sala et al., 2018). MPAs, linked to a broader programme of watershed management and/or under an ICZM programme, are expected to significantly strengthen health outcomes (Cicin-Sain and Belfiore, 2005), and therefore present a sensible management proxy for PES.

It is clear that even with passive conservation measures promoting reef health, some sort of active management would be required to attain the reef depth and rugosity parameters required for reefs that have lost their ability to attenuate wave energy in the short term. In order to achieve optimal outcomes, a combination of active (e.g., reef gardening) and passive (e.g., habitat protection) actions should be utilised (Possingham et al., 2015).

We suggest utilising PES in cases where the beach is stable with a goal of ensuring continued and even improved stability. Consideration of gray-green solutions such as reef augmentation, a concept that integrates nature into the building process (Waterman, 2008) might be a means of creating short term improvements specifically for coastal protection, while allowing for the longer term benefits of overall reef protection. The potential of using PES for such innovative concepts

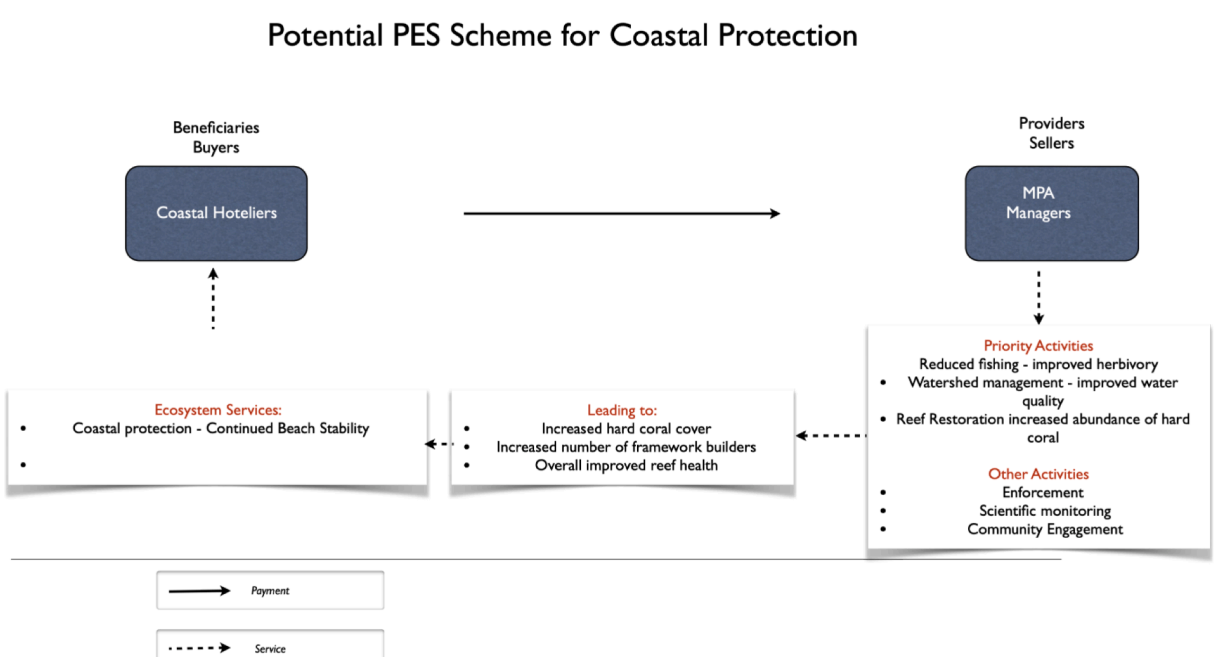

Fig. 5. Potential PES Scheme between buyers (coastal hoteliers) and sellers (MPA managers) of coastal protection.

should be explored further, but it is outside of the scope of this paper. Future research should concentrate on the most effective means of reef augmentation to promote service flow.

It should be noted that average coral cover on Caribbean reefs (reported at the last regional census in 2012) was 14.3% with coral cover declining at 75% of the 88 sites (Jackson et al., 2014). If the trajectory continues, as is expected with the impeding threats of global climate change, the ability of reefs in this region to attenuate wave energy and maintain themselves will be severely diminished. Coral health will therefore become even more important.

PES could play a stronger role in coral reef conservation for this ES, especially if this service was bundled with other more clearly defined and quantified services, such as aesthetics and fish biomass.

Knowledge gaps must be admitted up front to the buyers of the service and the goal must be clear; forestalling potential loss in revenue from beach erosion, by investing in environmental stewardship of reefs that actively protect beaches.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

References

Alvarez-Filip, L., Carricart-Ganivet, J.P., Horta-Puga, G., Iglesias-Prieto, R., 2013. Shifts in coral-assemblage composition do not ensure persistence of reef functionality. Sci. Rep. 3 (1) https://doi.org/10.1038/srep03486.

Alvarez-Filip, L., Dulvy, N.K., Côté, I.M., Watkinson, A.R., Gill, J.A., 2011. Coral identity underpins architectural complexity on Caribbean reefs. Ecol. Appl. 21 (6), 2223–2231. https://doi.org/10.1890/10-1563.1.

Alvarez-Filip, L., Dulvy, N.K., Gill, J.A., Coté, I.M., Watkinson, A.R., 2009. Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. Proc. Royal Soc. B: Biol. Sci. 276 (1669), 3019–3025.

Alvarez-Filip, L., Côté, I.M., Gill, J.A., Watkinson, A.R., Dulvy, N.K., 2011. Region-wide temporal and spatial variation in Caribbean reef architecture: is coral cover the whole story? Glob. Change Biol. 17 (7), 2470–2477.

Atmodjo, E., Lamers, M., Mol, A., 2017. Financing marine conservation tourism: governing entrance fees in Raja Ampat, Indonesia. Marine Policy 78, 181–188. https://doi.org/10.1016/j.marpol.2017.01.023.

Barker, N.H.L., Roberts, C.M., 2004. Scuba diver behaviour and the management of diving impacts on coral reefs. Biol. Conserv. 120 (4), 481–489. https://doi.org/10.1016/j.biocon.2004.03.021.

Bartley, R., Bainbridge, Z.T., Lewis, S.E., Kroon, F.J., Wilkinson, S.N., Brodie, J.E., Silburn, D.M., 2014. Relating sediment impacts on coral reefs to watershed sources, processes and management: a review. Sci. Total Environ. 468–469, 1138–1153. https://doi.org/10.1016/j.scitotenv.2013.09.030.

Basconi, L., Cadier, C., Guerrero-Limón, G., 2020. Challenges in marine restoration ecology: how techniques, assessment metrics, and ecosystem valuation can lead to improved restoration success. YOUMARES 9-The Oceans: Our Research, Our Future, Springer, Cham, pp. 83–99.

Baums, I.B., Johnson, M.E., Devlin-Durante, M.K., Miller, M.W., 2010. Host population genetic structure and zooxanthellae diversity of two reef-building coral species along the Florida Reef Tract and wider Caribbean. Coral Reefs 29 (4), 835–842. https://doi.org/10.1007/s00338-010-0645-y.

Bayraktarov, E., Saunders, M.I., Abdullah, S., Mills, M., Beher, J., Possingham, H.P., Mumby, P.J., Lovelock, C.E., 2016. The cost and feasibility of marine coastal restoration. Ecol. Appl. 26 (4), 1055–1074. https://doi.org/10.1890/15-1077.

Beatty, D.S., Valayil, J.M., Clements, C.S., Ritchie, K.B., Stewart, F.J., Hay, M.E., 2019. Variable effects of local management on coral defenses against a thermally regulated bleaching pathogen. Sci. Adv. 5 (10), eaay1048. https://doi.org/10.1126/sciadv.aay1048.

Beck, M.W., Losada, I.J., Menéndez, P., Reguero, B.G., Díaz-Simal, P., Fernández, F., 2018. The global flood protection savings provided by coral reefs. Nat. Commun. 9 (1), 1–9.

Beetham, E., Kench, P.S., Popinet, S., 2017. Future reef growth can mitigate physical impacts of sea-level rise on atoll islands: sea-level rise impacts on atoll islands. Earth's Future 5 (10), 1002–1014. https://doi.org/10.1002/2017EF000589.

Belfiore, S., Cicin-Sain, B., Ehler, C.N., 2004. Incorporating Marine Protected Areas into Integrated Coastal and Ocean Management: Principles and Guidelines. IUCN Gland and Cambridge UK.

Bellwood, D.R., Hughes, T.P., Folke, C., Nyström, M., 2004. Confronting the coral reef crisis. Nature 429 (6994), 827–833.

Bladon, A.J., Short, K.M., Mohammed, E.Y., Milner-Gulland, E.J., 2016. Payments for ecosystem services in developing world fisheries. Fish Fish. 17 (3), 839–859. https://doi.org/10.1111/faf.12095.

Bos, M., Pressey, R.L., Stoeckl, N., 2015. Marine conservation finance: the need for and scope of an emerging field. Ocean Coast. Manag. 114, 116–128. https://doi.org/10.1016/j.ocecoaman.2015.06.021.

Boström-Einarsson, L., Babcock, R.C., Bayraktarov, E., Ceccarelli, D., Cook, N., Ferse, S.C., Hancock, B., Harrison, P., Hein, M., Shaver, E., 2020. Coral restoration–a systematic review of current methods, successes, failures and future directions. PloS one 15 (1).

Boström-Einarsson, L., Ceccarelli, D., Babcock, R.C., Bayraktarov, E., Cook, N., Harrison, P., Hein, M., Shaver, E., Smith, A., Stewart-Sinclair, P.J., 2018. Coral Restoration in a Changing World - A Global Synthesis of Methods and Techniques. Reef and Rainforest Research Centre Ltd., Cairns.

Bozec, Y.-M., Alvarez-Filip, L., Mumby, P.J., 2015. The dynamics of architectural complexity on coral reefs under climate change. Glob. Change Biol. 21 (1), 223–235. https://doi.org/10.1111/gcb.12698.

Braat, L.C., de Groot, R., 2012. The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst. Serv. 1 (1), 4–15. https://doi.org/10.1016/j.ecoser.2012.07.011.

Brandl, S.J., Rasher, D.B., Côté, I.M., Casey, J.M., Darling, E.S., Lefcheck, J.S., Duffy, J.E., 2019. Coral reef ecosystem functioning: eight core processes and the role of biodiversity. Front. Ecol. Environ. 17 (8), 445–454. https://doi.org/10.1002/fee.2088.

Bruno, J.F., Côté, I.M., Toth, L.T., 2019. Climate change, coral loss, and the curious case of the parrotfish paradigm: why don't marine protected areas improve reef resilience? Annu. Rev. Mar. Sci. 11 (1), 307–334. https://doi.org/10.1146/annurev-marine-010318-095300.

Burke, L., Greenhalgh, S., Prager, D., Cooper, E., 2008. Coastal capital: economic valuation of coral reefs in Tobago and St. Lucia. Working Paper. World Resource Institute, Washington DC.

Burke, L., Reytar, K., Spalding, M., Perry, A., 2011. Reefs at Risk Revisited. World Resources Institute, Washington DC.

Burkepile, D.E., Hay, M.E., 2008. Herbivore species richness and feeding complementarity affect community structure and function on a coral reef. Proc. Natl. Acad. Sci. 105 (42), 16201–16206. https://doi.org/10.1073/pnas.0801946105.

Castaño-Isaza, J., Newball, R., Roach, B., Lau, W.W.Y., 2015. Valuing beaches to develop payment for ecosystem services schemes in Colombia's Seaflower marine protected area. Ecosyst. Serv. 11, 22–31. https://doi.org/10.1016/j.ecoser.2014.10.003.

Cicin-Sain, B., Belfiore, S., 2005. Linking marine protected areas to integrated coastal and ocean management: a review of theory and practice. Ocean Coast. Manag. 48 (11-12), 847–868. https://doi.org/10.1016/j.ocecoaman.2006.01.001.

Claudet, J., Guidetti, P., Mouillot, D., Shears, N.T., Micheli, F., 2011. Ecology-Ecological Effects of Marine Protected Areas: Conservation, Restoration, and Functioning. A Multidisciplinary Approach. Cambridge University Press. Cambridge, Marine Protected Areas, pp. 37–71.

Costa, M.B.S.F., Araújo, M., Araújo, T.C.M., Siegle, E., 2016. Influence of reef geometry on wave attenuation on a Brazilian coral reef. Geomorphology 253, 318–327. https://doi.org/10.1016/j.geomorph.2015.11.001.

Côté, I.M., Darling, E.S., 2010. Rethinking ecosystem resilience in the face of climate change. PLoS Biol. 8 (7).

Cox, C., 2014. Evaluating Strategies for Restoring Parrotfish Populations in Belize. Doctor of Philosophy Dissertation. University of North Carolina, Chapel Hill.

Cox, C., Valdivia, A., McField, M., Castillo, K., Bruno, J.F., 2017. Establishment of marine protected areas alone does not restore coral reef communities in Belize. Mar. Ecol. Prog. Ser. 563, 65–79. https://doi.org/10.3354/meps11984.

Darling, E.S., McClanahan, T.R., Maina, J., Gurney, G.G., Graham, N.A.J., Januchowski-Hartley, F., Cinner, J.E., Mora, C., Hicks, C.C., Maire, E., Puotinen, M., Skirving, W.J., Adjeroud, M., Ahmadia, G., Arthur, R., Bauman, A.G., Beger, M., Berumen, M.L., Bigot, L., Bouwmeester, J., Brenier, A., Bridge, T.C.L., Brown, E., Campbell, S.J., Cannon, S., Cauvin, B., Chen, C.A., Claudet, J., Denis, V., Donner, S., Estradivari, Fadli, N., Feary, D.A., Fenner, D., Fox, H., Franklin, E.C., Friedlander, A., Gilmour, J., Goiran, C., Guest, J., Hobbs, J.-P., Hoey, A.S., Houk, P., Johnson, S., Jupiter, S.D., Kayal, M., Kuo, C.-Y., Lamb, J., Lee, M.A.C., Low, J., Muthiga, N., Muttaqin, E., Nand, Y., Nash, K.L., Nedlic, O., Pandolfi, J.M., Pardede, S., Patankar, V., Penin, L., Ribas-Deulofeu, L., Richards, Z., Roberts, T.E., Rodgers, K.S., Safuan, C.D.M., Sala, E., Shedrawi, G., Sin, T.M., Smallhorn-West, P., Smith, J.E., Sommer, B., Steinberg, P.D., Sutthacheep, M., Tan, C.H.J., Williams, G.J., Wilson, S., Yeemin, T., Bruno, J.F., Fortin, M.-J., Krkosek, M., Mouillot, D., 2019. Social–environmental drivers inform strategic management of coral reefs in the Anthropocene. Nat. Ecol. Evol. 3 (9), 1341–1350. https://doi.org/10.1038/s41559-019-0953-8.

Day, J., Dudley, N., Hockings, M., Holmes, G., Laffoley, D.D.A., Stolton, S., Wells, S.M. (Eds.), 2012. Guidelines for Applying the IUCN Protected Area Management Categories to Marine Protected Areas, 2nd ed. IUCN, Gland, Switzerland.

Ruiz de Alegria-Arzaburu, A., Mariño-Tapia, I., Enriquez, C., Silva, R., González-Leija, M., 2013. The role of fringing coral reefs on beach morphodynamics. Geomorphology 198, 69–83. https://doi.org/10.1016/j.geomorph.2013.05.013.

De'ath, G., Fabricius, K., 2010. Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecol. Appl. 20 (3), 840–850. https://doi.org/10.1890/08-2023.1.

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. Proc. Natl. Acad. Sci. 109 (44), 17995–17999. https://doi.org/10.1073/pnas.1208909109.

Dikou, A., 2010. Ecological processes and contemporary coral reef management. Diversity 2 (5), 717–737.

Drury, C., Lirman, D., 2017. Making biodiversity work for coral reef restoration. Biodiversity 18 (1), 23–25. https://doi.org/10.1080/14888386.2017.1318094.

Eakin, C., 1996. Where have all the carbonates gone? A model comparison of calcium carbonate budgets before and after the 1982–1983 El Nino at Uva Island in the eastern Pacific. Coral Reefs 15 (2), 109–119.
Eakin, C.M., 2001. A tale of two ENSO events: carbonate budgets and the influence of two warming disturbances and intervening variability, Uva Island, Panama. Bull. Mar. Sci. 69 (1), 171–186.
Eberhard, R., Thorburn, P., Rolfe, J., Taylor, B., Ronan, M., Webber, T., Flint, N., Kroon, F., Silburn, M., Bartley, R., 2017. 2017 Scientific Consensus Statement: land use impacts on the Great Barrier Reef water quality and ecosystem condition. Chapter 4: management options and their effectiveness.
Elliff, C.I., Silva, I.R., 2017. Coral reefs as the first line of defense: Shoreline protection in face of climate change. Mar. Environ. Res. 127, 148–154. https://doi.org/10.1016/j.marenvres.2017.03.007.
Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecol. Econ. 65 (4), 663–674. https://doi.org/10.1016/j.ecolecon.2008.03.011.
Estrada-Saldívar, N., Jordán-Dalhgren, E., Rodríguez-Martínez, R.E., Perry, C., Alvarez-Filip, L., 2019. Functional consequences of the long-term decline of reef-building corals in the Caribbean: evidence of across-reef functional convergence. R. Soc. Open Sci. 6 (10), 190298. https://doi.org/10.1098/rsos.190298.
Fabricius, K.E., Logan, M., Weeks, S., Brodie, J., 2014. The effects of river run-off on water clarity across the central Great Barrier Reef. Mar. Pollut. Bull. 84 (1-2), 191–200. https://doi.org/10.1016/j.marpolbul.2014.05.012.
Ferrario, F., Beck, M.W., Storlazzi, C.D., Micheli, F., Shepard, C.C., Airoldi, L., 2014. The effectiveness of coral reefs for coastal hazard risk reduction and adaptation. Nat. Commun. 5 (1), 1–9.
Figueira, W., Ferrari, R., Weatherby, E., Porter, A., Hawes, S., Byrne, M., 2015. Accuracy and precision of habitat structural complexity metrics derived from underwater photogrammetry. Rem. Sens. 7 (12), 16883–16900.
Fillols, E., Davis, A.M., Lewis, S.E., Ward, A., 2020. Combining weed efficacy, economics and environmental considerations for improved herbicide management in the Great Barrier Reef catchment area. Sci. Total Environ. 720, 137481. https://doi.org/10.1016/j.scitotenv.2020.137481.
Flower, J., Ortiz, J.C., Chollett, I., Abdullah, S., Castro-Sanguino, C., Hock, K., Lam, V., Mumby, P.J., 2017. Interpreting coral reef monitoring data: a guide for improved management decisions. Ecol. Ind. 72, 848–869.
Forest Trends, The Katoomba Group, Unep, 2008. Payments for Ecosystem Services: Getting Started—A Primer. Harris Litho, Washington DC.
Fox, H.E., Harris, J.L., Darling, E.S., Ahmadia, G.N., Razak, T.B., 2019. Rebuilding coral reefs: success (and failure) 16 years after low-cost, low-tech restoration. Restor. Ecol. 27 (4), 862–869.
Fripp, E., 2014. Payments for Ecosystem Services (PES): A practical guide to assessing the feasibility of PES projects. CIFOR.
Fujita, R., Lynham, J., Micheli, F., Feinberg, P.G., Bourillón, L., Sáenz-Arroyo, A., Markham, A.C., 2013. Ecomarkets for conservation and sustainable development in the coastal zone. Biol. Rev. 88 (2), 273–286.
Galil, B.S., Mckenzie, C., Bailey, S., Campbell, M., Davidson, I., Drake, L., Hewitt, C., Occhipinti-Ambrogi, A., Piola, R., 2019. ICES AD HOC REPORT 2019.
Gallop, S.L., Young, I.R., Ranasinghe, R., Durrant, T.H., Haigh, I.D., 2014. The large-scale influence of the Great Barrier Reef matrix on wave attenuation. Coral Reefs 33 (4), 1167–1178.
Gardner, T.A., Côté, I.M., Gill, J.A., Grant, A., Watkinson, A.R., 2003. Long-term region-wide declines in Caribbean corals. Science 301 (5635), 958–960.
Gill, D.A., Mascia, M.B., Ahmadia, G.N., Glew, L., Lester, S.E., Barnes, M., Craigie, I., Darling, E.S., Free, C.M., Geldmann, J., 2017. Capacity shortfalls hinder the performance of marine protected areas globally. Nature 543 (7647), 665–669.
Gourlay, M.R., Colleter, G., 2005. Wave-generated flow on coral reefs—an analysis for two-dimensional horizontal reef-tops with steep faces. Coast. Eng. 52 (4), 353–387.
Graham, N., Nash, K., 2013. The importance of structural complexity in coral reef ecosystems. Coral Reefs 32, 315–326.
Green, S.J., Akins, J.L., Maljković, A., Côté, I.M., 2012. Invasive lionfish drive Atlantic coral reef fish declines. PLoS ONE 7 (3).
Guannel, G., Arkema, K., Ruggiero, P., Verutes, G., 2016. The power of three: coral reefs, seagrasses and mangroves protect coastal regions and increase their resilience. PLoS ONE 11 (7), e0158094.
Guerry, A.D., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G.C., Griffin, R., Ruckelshaus, M., Bateman, I.J., Duraiappah, A., Elmqvist, T., 2015. Natural capital and ecosystem services informing decisions: from promise to practice. Proc. Natl. Acad. Sci. 112 (24), 7348–7355.
Harborne, A.R., Rogers, A., Bozec, Y.-M., Mumby, P.J., 2017. Multiple stressors and the functioning of coral reefs.
Harris, D.L., Rovere, A., Casella, E., Power, H., Canavesio, R., Collin, A., Pomeroy, A., Webster, J.M., Parravicini, V., 2018. Coral reef structural complexity provides important coastal protection from waves under rising sea levels. Sci. Adv. 4 (2), eaao4350.
Hawkins, J.P., Roberts, C.M., 1994. The growth of coastal tourism in the red sea: present and future effects on coral reefs. Ambio 23 (8), 503–508.
Hill, J., Wilkinson, C., 2004. Methods for Ecological Monitoring of Coral Reefs.
Hobbs, R.J., Harris, J.A., 2001. Restoration ecology: repairing the earth's ecosystems in the new millennium. Restor. Ecol. 9 (2), 239–246.
Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E., Harvell, C.D., Sale, P.F., Edwards, A.J., Caldeira, K., 2007. Coral reefs under rapid climate change and ocean acidification. Science 318 (5857), 1737–1742.
Hughes, E.A., 2003. Review: climate change, human impacts, and the resilience of coral reefs. Science 301, 929.
Hughes, T.P., Graham, N.A., Jackson, J.B., Mumby, P.J., Steneck, R.S., 2010. Rising to the challenge of sustaining coral reef resilience. Trends Ecol. Evol. 25 (11), 633–642.
Hughes, T.P., Rodrigues, M.J., Bellwood, D.R., Ceccarelli, D., Hoegh-Guldberg, O., Mccook, L., Moltschaniwskyj, N., Pratchett, M.S., Steneck, R.S., Willis, B., 2007. Phase shifts, herbivory, and the resilience of coral reefs to climate change. Curr. Biol. 17 (4), 360–365.
Hunte, W., Younglao, D., 1988. Recruitment and population recovery of Diadema antillarum (Echinodermata; Echinoidea) in Barbados. Mar. Ecol. Prog. Ser. 45, 109–119.
Hutchings, P.A., 1986. Biological destruction of coral reefs. Coral Reefs 4, 239–252.
Idjadi, J.A., Haring, R.N., Precht, W.F., 2010. Recovery of the sea urchin Diadema antillarum promotes scleractinian coral growth and survivorship on shallow Jamaican reefs. Mar. Ecol. Prog. Ser. 403, 91–100.
Ingram, J.C., Wilkie, D., Clements, T., Mcnab, R.B., Nelson, F., Baur, E.H., Sachedina, H. T., Peterson, D.D., Foley, C.a.H., 2014. Evidence of payments for ecosystem services as a mechanism for supporting biodiversity conservation and rural livelihoods. Ecosyst. Serv. 7, 10–21.
Iyer, V., Mathias, K., Meyers, D., Victurine, R., Walsh, M., 2018. Finance Tools for Coral Reef Conservation: A Guide. Conservation Finance Alliance.
Jackson, J., Donovan, M., Cramer, K., Lam, V., 2014. Status and Trends of Caribbean Coral Reefs. Global Coral Reef Monitoring Network, IUCN, Gland, Switzerland, pp. 1970–2012.
Johnson, M., Sandell, J., 2014. Advances in Marine Biology: Marine Managed Areas. Elsevier 416, London.
Jokiel, P.L., Brown, E.K., 2004. Reef Coral Communities at Pila'a Reef: Results of the 2004 resurvey. Hawaii Coral Reef Assessment and Monitoring Program (CRAMP), Hawaii Institute of Marine Biology, PO Box 1346.
Kench, P.S., Brander, R.W., 2006. Wave processes on coral reef flats: implications for reef geomorphology using Australian case studies. J. Coastal Res. 22 (1 (221)), 209–223.
Kennedy, E.V., Perry, C.T., Halloran, P.R., Iglesias-Prieto, R., Schonberg, C.H.L., Wisshak, M., Form, A.U., Carricart-Ganivet, J.P., Fine, M., Eakin, C.E., Mumby, P.J., 2013. Avoiding coral reef functional collapse requires local and global action. Curr. Biol. 23, 912–918.
Koch, E.W., Barbier, E.B., Silliman, B.R., Reed, D.J., Perillo, G.M., Hacker, S.D., Granek, E.F., Primavera, J.H., Muthiga, N., Polasky, S., 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. Front. Ecol. Environ. 7 (1), 29–37.
Kroon, F.J., Thorburn, P., Schaffelke, B., Whitten, S., 2016. Towards protecting the Great Barrier Reef from land-based pollution. Glob. Change Biol. 22 (6), 1985–2002.
Kuffner, I.B., Toth, L.T., 2016. A geological perspective on the degradation and conservation of western Atlantic coral reefs. Conserv. Biol. 30 (4), 706–715.
Kushner, B., Edwards, P., Burke, L., Cooper, E., 2011. Coastal Capital: Jamaica. Coral Reefs, Beach Erosion and Impacts to Tourism in Jamaica. Working Paper. Washington, DC: World Resources Institute.
Ladd, M.C., Shantz, A.A., 2020. Trophic interactions in coral reef restoration: a review. Food Webs 24, e00149.
Laffoley, D., Baxter, J.M., Day, J.C., Wenzel, L., Bueno, P., Zischka, K., 2019. Marine protected areas. World seas, An environmental evaluation, Elsevier, pp. 549–569.
Lang, J., Marks, K., Kramer, P., Kramer, P., Ginsburg, R., 2010. Atlantic and Gulf Rapid Reef Assessment (AGRRA) Protocols Version 5.4. University of Miami, Miami-Florida.
Lange, I.D., Perry, C.T., Alvarez-Filip, L., 2020. Carbonate budgets as indicators of functional reef "health": a critical review of data underpinning census-based methods and current knowledge gaps. Ecol. Ind. 110, 105857.
Lau, W.Y., 2012. San Andres Island, Colombia: A Case Study of Stacking and Bundling in the Marine and Coastal Context. Prepared for the USAID Bundling and Stacking for Maximizing Social, Ecological, and Economic Benefits: A Framing Paper for Discussion at the "Bundling and Stacking Workshop". Washington DC.
Lau, W.Y., 2013. Beyond carbon: conceptualizing payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. Ocean Coast. Manag. 83, 5–14.
Leenhardt, P., Stelzenmüller, V., Pascal, N., Probst, W.N., Aubanel, A., Bambridge, T., Charles, M., Clua, E., Féral, F., Quinquis, B., Salvat, B., Claudet, J., 2017. Exploring social-ecological dynamics of a coral reef resource system using participatory modeling and empirical data. Marine Policy 78, 90–97.
Lewis, J.B., 1984. The Acropora inheritance: a reinterpretation of the development of fringing reefs in Barbados, West Indies. Coral Reefs 3 (3), 117–122.
Lowe, R.J., Falter, J.L., Koseff, J.R., Monismith, S.G., Atkinson, M.J., 2007. Spectral wave flow attenuation within submerged canopies: implications for wave energy dissipation. J. Geophys. Res. Oceans 112 (C5).
Lugo-Fernandez, A., Roberts, H., Wiseman Jr, W., 1998. Tide effects on wave attenuation and wave set-up on a Caribbean coral reef. Estuar. Coast. Shelf Sci. 47 (4), 385–393.
Maragos, J.E., 1993. Impact of coastal construction on coral reefs in the US-affiliated pacific Islands. Coast. Manag. 21 (4), 235–269.
Maynard, J., Van Hooidonk, R., Eakin, C.M., Puotinen, M., Garren, M., Williams, G., Heron, S.F., Lamb, J., Weil, E., Willis, B., 2015. Projections of climate conditions that increase coral disease susceptibility and pathogen abundance and virulence. Nat. Clim. Change 5 (7), 688–694.
Mcclure, E.C., Sievers, K.T., Abesamis, R.A., Hoey, A.S., Alcala, A.C., Russ, G.R., 2020. Higher fish biomass inside than outside marine protected areas despite typhoon impacts in a complex reefscape. Biol. Conserv. 241, 108354.
Mcleod, E., Anthony, K.R.N., Mumby, P.J., Maynard, J.E.A., 2019. The future of resilience-based management in coral reef ecosystems. J. Environ. Manage. 233, 291–301.

Mcwilliam, M., Pratchett, M.S., Hoogenboom, M.O., Hughes, T.P., 2020. Deficits in functional trait diversity following recovery on coral reefs. Proc. Royal Soc. B 287 (1918), 20192628.
Mehvar, S., Filatova, T., Dastgheib, A., De Ruyter Van Steveninck, E., Ranasinghe, R., 2018. Quantifying economic value of coastal ecosystem services: a review. J. Mar. Sci. Eng. 6 (1), 5.
Meyers, D., Alliance, C.F., Bohorquez, J., Cumming, B., Emerton, L., Riva, M., Fund, U.J. S., Victurine, R., 2020. Conservation finance: a framework. Conservation Finance Alliance 1–45.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Wetlands and Water. World Resources Institute.
Moberg, F., Folke, C., 1999. Ecological goods and services of coral reef ecosystems. Ecol. Econ. 29, 215–233.
Monismith, S.G., Rogers, J.S., Koweek, D., Dunbar, R.B., 2015. Frictional wave dissipation on a remarkably rough reef. Geophys. Res. Lett. 42 (10), 4063–4071.
Moros, L., Corbera, E., Vélez, M.A., Flechas, D., 2020. Pragmatic conservation: discourses of payments for ecosystem services in Colombia. Geoforum 108, 169–183.
Mumby, P.J., Flower, J., Chollett, I., Box, S.J., Bozec, Y.-M., Fitzsimmons, C., Forster, J., Gill, D., Griffith-Mumby, R., Oxenford, H.A., 2014. Towards Reef Resilience and Sustainable Livelihoods: A Handbook for Caribbean Coral Reef Managers. University of Exeter.
Mumby, P.J., Harborne, A.R., 2010. Marine reserves enhance the recovery of corals on Caribbean reefs. PLoS ONE 5 (1), e8657.
Mumby, P.J., Steneck, R.S., 2008. Coral reef management and conservation in light of rapidly evolving ecological paradigms. Trends Ecol. Evol. 23 (10), 555–563.
Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: an alternative conceptual framework for understanding payments for environmental services. Ecol. Econ. 69 (6), 1202–1208.
O'farrell, S., Luckhurst, B.E., Box, S.J., Mumby, P.J., 2016. Parrotfish sex ratios recover rapidly in Bermuda following a fishing ban. Coral Reefs 35 (2), 421–425.
Obura, D.O., Aeby, G., Amornthammarong, N., Appeltans, W., Bax, N., Bishop, J., Brainard, R.E., Chan, S., Fletcher, P., Gordon, T.a.C., Gramer, L., Gudka, M., Halas, J., Hendee, J., Hodgson, G., Huang, D., Jankulak, M., Jones, A., Kimura, T., Levy, J., Miloslavich, P., Chou, L.M., Muller-Karger, F., Osuka, K., Samoilys, M., Simpson, S.D., Tun, K., Wongbusarakum, S., 2019. Coral reef monitoring, reef assessment technologies, and ecosystem-based management. Front. Mar. Sci. 6 (580).
Osorio-Cano, J.D., Osorio, A., Peláez-Zapata, D., 2019. Ecosystem management tools to study natural habitats as wave damping structures and coastal protection mechanisms. Ecol. Eng. 130, 282–295.
Pachauri, R., Meyer, L., Plattner, G., Stocker, T., 2014. IPCC, 2014: climate change 2014: synthesis report. In: Contribution of Working Groups I, II and III to the Fifth Assessment Report of the intergovernmental panel on Climate Change, pp. 1–151.
Pandolfi, J.M., 2015. Incorporating uncertainty in predicting the future response of coral reefs to climate change. Annu. Rev. Ecol. Evol. Syst. 46, 281–303.
Pandolfi, J.M., Jackson, J.B., 2006. Ecological persistence interrupted in Caribbean coral reefs. Ecol. Lett. 9 (7), 818–826.
Pascal, N., Allenbach, M., Brathwaite, A., Burke, L., Le Port, G., Clua, E., 2016. Economic valuation of coral reef ecosystem service of coastal protection: a pragmatic approach. Ecosyst. Serv. 21, 72–80.
Pascal, N., Brathwaite, A., Brander, L., Seidl, A., Philip, M., Clua, E., 2018a. Evidence of economic benefits for public investment in MPAs. Ecosyst. Serv. 30, 3–13.
Pascal, N., Brathwaite, A., Bryan, T., Philip, M., Walsh, M., 2018b. Impact investment in marine conservation. Duke Environ. Law Policy Forum 28, 199–220.
Perry, C.T., Alvarez-Filip, L., Graham, N.A., Mumby, P.J., Wilson, S.K., Kench, P.S., Manzello, D.P., Morgan, K.M., Slangen, A.B., Thomson, D.P., 2018. Loss of coral reef growth capacity to track future increases in sea level. Nature 558 (7710), 396–400.
Perry, C.T., Murphy, G.N., Kench, P.S., Smithers, S.G., Edinger, E.N., Steneck, R.S., Mumby, P.J., 2013. Caribbean-wide decline in carbonate production threatens coral reef growth. Nat. Commun. 4, 1402.
Possingham, H.P., Bode, M., Klein, C.J., 2015. Optimal conservation outcomes require both restoration and protection. PLoS Biol. 13 (1), e1002052.
Principe, P., Bradley, P., Yee, S.H., Fisher, W.S., Johnson, E., Allen, P.E., Campbell, D.E., 2012. Quantifying Coral Reef Ecosystem Services. US Environmental Protection Agency, North Carolina.
Quataert, E., Storlazzi, C., Van Rooijen, A., Cheriton, O., Van Dongeren, A., 2015. The influence of coral reefs and climate change on wave-driven flooding of tropical coastlines. Geophys. Res. Lett. 42 (15), 6407–6415.
Rasher, D.B., Hay, M.E., 2010. Chemically rich seaweeds poison corals when not controlled by herbivores. Proc. Natl. Acad. Sci. 107 (21), 9683–9688.
Reef Check Instruction Manual, 2006. A guide to reef check coral reef monitoring. Reef Check.
Reguero, B.G., Beck, M.W., Agostini, V.N., Kramer, P., Hancock, B., 2018. Coral reefs for coastal protection: a new methodological approach and engineering case study in Grenada. J. Environ. Manage. 210, 146–161.
Richmond, R.H., 1993. Coral reefs: present problems and future concerns resulting from anthropogenic disturbance. Am. Zool. 33 (6), 524–536.
Rinkevich, B., 2008. Management of coral reefs: we have gone wrong when neglecting active reef restoration. Mar. Pollut. Bull. 56 (11), 1821–1824.
Roberts, C.M., O'leary, B.C., Mccauley, D.J., Cury, P.M., Duarte, C.M., Lubchenco, J., Pauly, D., Sáenz-Arroyo, A., Sumaila, U.R., Wilson, R.W., 2017. Marine reserves can mitigate and promote adaptation to climate change. Proc. Natl. Acad. Sci. 114 (24), 6167–6175.
Rogers, C.S., 1990. Responses of coral reefs and reef organisms to sedimentation. Mar. Ecol. Progr. Ser. Oldendorf 62 (1), 185–202.
Ryan, E.J., Hanmer, K., Kench, P.S., 2019. Massive corals maintain a positive carbonate budget of a Maldivian upper reef platform despite major bleaching event. Scientific Reports, p. 6515.
Sala, E., Lubchenco, J., Grorud-Colvert, K., Novelli, C., Roberts, C., Sumaila, U.R., 2018. Assessing real progress towards effective ocean protection. Marine Policy 91, 11–13.
Sandin, S.A., Smith, J.E., Demartini, E.E., Dinsdale, E.A., Donner, S.D., Friedlander, A.M., Konotchick, T., Malay, M., Maragos, J.E., Obura, D., 2008. Baselines and degradation of coral reefs in the Northern Line Islands. PLoS ONE 3 (2).
Sciberras, M., Jenkins, S.R., Mant, R., Kaiser, M.J., Hawkins, S.J., Pullin, A.S., 2015. Evaluating the relative conservation value of fully and partially protected marine areas. Fish Fish. 16 (1), 58–77.
Selig, E.R., Bruno, J.F., 2010. A global analysis of the effectiveness of marine protected areas in preventing coral loss. PLoS ONE 5 (2), e9278.
Shaver, E.C., Silliman, B.R., 2017. Time to cash in on positive interactions for coral restoration. PeerJ 5, e3499.
Shelton Iii, A.J., Richmond, R.H., 2016. Watershed restoration as a tool for improving coral reef resilience against climate change and other human impacts. Estuar. Coast. Shelf Sci. 183, 430–437.
Sheppard, C., Dixon, D.J., Gourlay, M., Sheppard, A., Payet, R., 2005. Coral mortality increases wave energy reaching shores protected by reef flats: examples from the Seychelles. Estuar. Coast. Shelf Sci. 64 (2–3), 223–234.
Sheppard, C.R., Spalding, M., Bradshaw, C., Wilson, S., 2002. Erosion vs. recovery of coral reefs after 1998 El Niño: Chagos reefs, Indian Ocean. AMBIO: J. Hum. Environ. 31 (1), 40–48.
Silver, J.M., Arkema, K.K., Griffin, R.M., Lashley, B., Lemay, M., Maldonado, S., Moultrie, S.H., Ruckelshaus, M., Schill, S., Thomas, A., 2019. Advancing coastal risk reduction science and implementation by accounting for climate, ecosystems, and people. Front. Mar. Sci. 6, 556.
Sommerville, M.M., Jones, J.P., Milner-Gulland, E., 2009. A revised conceptual framework for payments for environmental services. Ecol. Soc. 14 (2).
Spalding, M., Longley-Wood, K., Acosta-Morel, M., Cole, A., Wood, S., Haberland, C., Ferdana, Z., 2018. Estimating Reef-Adjacent Tourism Value in the Caribbean.
Steneck, R., Arnold, S.N., Boenish, R., De Leon, R., Mumby, P.J., Rasher, D.B., Wilson, M., 2019. Managing recovery resilience in coral reefs against climate-induced bleaching and hurricanes: a 15 year case study from Bonaire, Dutch Caribbean. Front. Mar. Sci. 6, 265.
Steneck, R.S., Mumby, P.J., Macdonald, C., Rasher, D.B., Stoyle, G., 2018. Attenuating effects of ecosystem management on coral reefs. Sci. Adv. 4 (5), eaao5493.
Stolton, S., Hockings, M., Dudley, N., Mackinnon, K., Whitten, T., Leverington, F., 2007. Management effectiveness tracking tool. Reporting Progress at Protected Area Sites, 2nd ed. WWF, Gland, Switzerland.
Storlazzi, C.D., Reguero, B.G., Cole, A.D., Lowe, E., Shope, J.B., Gibbs, A.E., Nickel, B.A., Mccall, R.T., Van Dongeren, A.R., Beck, M.W., 2019. Rigorously valuing the role of US coral reefs in coastal hazard risk reduction (No.2019-1027). US Geological Survey.
Sutherland, K.P., Ritchie, K.B., 2004. White pox disease of the Caribbean elkhorn coral. Springer, Acropora palmata. Coral Health and Disease, pp. 289–300.
Swallow, B.M., Kallesoe, M.F., Iftikhar, U.A., Van Noordwijk, M., Bracer, C., Scherr, S.J., Raju, K.V., Poats, S.V., Duraiappah, A.K., Ochieng, B.O., 2009. Compensation and rewards for environmental services in the developing world: framing pan-tropical analysis and comparison. Ecol. Soc. 14 (2).
Tacconi, L., 2012. Redefining payments for environmental services. Ecol. Econ. 73, 29–36.
Tomascik, T., Sander, F., 1985. Effects of eutrophication on reef-building corals. Mar. Biol. 87 (2), 143–155.
Toropova, C., Meliane, I., Laffoley, D., Matthews, E., Spalding, M. (Eds.), 2010. Global ocean protection: present status and future possibilities. IUCN Gland Switzerland.
U.S. Geological Survey Fact Sheet 025-02. Retrieved September 16th from https://pubs.usgs.gov/fs/2002/fs025-02.
Uyarra, M.C., Cote, I.M., Gill, J.A., Tinch, R.R., Viner, D., Watkinson, A.R., 2005. Island-specific preferences of tourists for environmental features: implications of climate change for tourism-dependent states. Environ. Conserv. 11–19.
Van Zanten, B.T., Van Beukering, P.J., Wagtendonk, A.J., 2014. Coastal protection by coral reefs: a framework for spatial assessment and economic valuation. Ocean Coast. Manag. 96, 94–103.
Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecol. Econ. 69 (6), 1245–1252.
Veron, J.E.N., 2002. New species described in Corals of the World. Citeseer.
Veron, J.E.N., 2004. Coral survey at selected sites in Arnhem Land. Australian Institute of Marine Science.
Wainger, L.A., Boyd, J.W., 2009. Valuing ecosystem services. Ecosystem-based Manag. Oceans 92–114.
Waterman, R.E., 2008. Integrated Coastal Policy via Building with Nature. Opmeer Drukkerij bv, The Hague.
Waylen, K.A., Julia Martin-Ortega, J., 2018. Surveying views on payments for ecosystem services: implications for environmental management and research. Ecosyst. Serv. 29, 23–30.
Weber, M., De Beer, D., Lott, C., Polerecky, L., Kohls, K., Abed, R.M., Ferdelman, T.G., Fabricius, K.E., 2012. Mechanisms of damage to corals exposed to sedimentation. Proc. Natl. Acad. Sci. 109 (24), E1558–E1567.
Weijerman, M., Veazey, L., Yee, S., Vaché, K., Delevaux, J., Donovan, M.K., Falinski, K., Lecky, J., Oleson, K.L., 2018. Managing local stressors for coral reef condition and ecosystem services delivery under climate scenarios. Front. Mar. Sci. 5, 425.
Weil, E., Rogers, C.S., 2011. Coral reef diseases in the Atlantic-Caribbean. Coral Reefs Ecosyst. Transit. 465–491.

Wells, S., Ravilious, C., 2006. In the front line: shoreline protection and other ecosystem services from mangroves and coral reefs. UNEP/Earthprint.

Wenger, A., Ahmadia, G., Alvarez-Romero, J., Barnes, M., Blythe, J., Brodie, J., Day, J., Fox, H., Gill, D., Gomez, N.A., 2017. Coral reef conservation solution-scape white paper.

Wielgus, J., Cooper, E., Torres, R., Burke, L., 2010. Coastal capital: Dominican Republic. Case studies on the economic value of coastal ecosystems in the Dominican Republic. World Resources Institute, Washington DC, USA.

World Bank, 2016. Managing coasts with natural solutions: guidelines for measuring and valuing the coastal protection services of mangroves and coral reefs. No.103340. Washington, DC.

Wunder, S., 2005. Payments for environmental services: some nuts and bolts. Center for International Forestry Research Occasional Paper No. 42.

Wunder, S., 2013. When payments for environmental services will work for conservation. Conserv. Lett. 6 (4), 230–237.

Wunder, S., 2015. Revisiting the concept of payments for environmental services. Ecol. Econ. 117, 234–243.

Wunder, S., Engel, S., Pagiola, S., 2008. Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. Ecol. Econ. 65 (4), 834–852.

Yao, Y., Zhang, Q., Chen, S., Tang, Z., 2019. Effects of reef morphology variations on wave processes over fringing reefs. Appl. Ocean Res. 82, 52–62.

Zawada, D.G., Brock, J.C., 2009. A multiscale analysis of coral reef topographic complexity using lidar-derived bathymetry. J. Coast. Res. 53, 6–15.

Zhao, M., Zhang, H., Zhong, Y., Jiang, D., Liu, G., Yan, H., Zhang, H., Guo, P., Li, C., Yang, H., Chen, T., Wang, R., 2019. The status of coral reefs and its importance for coastal protection: a case study of Northeastern Hainan Island, South China Sea. Sustainability 11 (16).