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Protection côtière : Approche écologique et financière pour améliorer la santé des récifs coralliens et le service écosystémique associé

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THÈSE DE DOCTORAT
DE L'UNIVERSITÉ PSL

Préparée à l'École Pratique des Hautes Études

**Protection côtière: approche écologiques et financières
pour améliorer la santé des récifs coralliens et le service
écosystémique associé**

Ecological and financial considerations for improving coral reef health, to enhance
ecosystem service flow

Soutenue par

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Coastal protection:
Ecological and economic considerations
for improving coral reef health, to
enhance ecosystem service flow

PhD Thesis

Doctorat PSL

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Abstract

Coastal Protection is an extremely important coral reef Ecosystem Service (ES) especially with the impending and increasing negative impacts of Global Climate Change (GCC) on coastal and marine ecosystems. Coastal protection has however not received the attention of the more visible ES such as aesthetics and provision of food via fisheries; perhaps due to challenges in clearly determining the role of the coral reef and living coral, in providing the service. Challenges notwithstanding, however, the contribution of reefs to service provision has been demonstrated in the science literature.

Coral reefs are declining on a global scale from local and global impacts, exacerbated by inadequate and unsustainable funding for their management. It is imperative that we determine the most suitable methods for improving reef health, in a world where the situation is both dire; with unprecedented levels of coral mortality and time sensitive; with an estimated less than 50 years remaining to act. A critical part of any solution is how to address the financial gap, at a time when traditional financing methods are failing and insufficient finance has been identified as a major obstacle to conservation success.

The aim of this thesis therefore is to investigate solutions for improving coastal protection and to explore the role of the private sector in financing interventions. We cross the disciplines of science, engineering and economy in our quest for solutions.

Chapter 1 Provides a general introduction and background for the thesis, by presenting the core elements of coastal protection, management and conservation finance, including the rationale for selections made.

Chapter 2 - **Evidence of economic benefits for public investment in MPAs**, sets the “scene” for the following chapters, by proposing a framework for the ecological and economic analysis of coral reef ecosystem services. In this paper we conducted Cost Benefit Analyses on coral reef management within Marine Protected Areas of 2 island nations. The objective was to provide evidence of the overall economic benefits of improvements made to 5 different ES. The coastal protection ES was identified as one of the most important for real estate owners, with the tourism industry benefitting from beach protection against erosion. The paper provided methods for the ecological quantification and economic valuation of the different ES. It also highlighted the knowledge gaps and needs for improvements in the methodology for the coastal protection ES valuation.

Chapter 3 - **Economic valuation of coral reef ecosystem service of coastal protection: A pragmatic approach**, provides a more precise valuation methodology for coastal protection provided by coral reefs, specifically for countries with low data availability. The method was found useful for policy advocacy and the identification of ES beneficiaries, but not deemed suitable for precise analysis such as identifying the protective role of living corals for the protection of a specific coast or beach. Queries re the role of living coral raised in this chapter were examined in Chapter 4.

Chapter 4 - **When are Payment for Ecosystems Services suitable for Coral Reef derived coastal protection?: a review of scientific requirements** reviews coral reef provision of coastal protection and determines its feasibility for a Payment for Ecosystem Services (PES). In the process, the role of living coral was analyzed and management actions identified that could improve reef health for service provision. This paper identifies coral restoration as a key intervention for coastal protection, highlighting the scientific knowledge gaps and the different risks of the interventions. It also identifies the primary beneficiaries and provides the economic rationale for Chapters 5 and 6.

Chapter 5 - **Impact Investment in marine conservation**, further explores innovative means of financing marine conservation via impact investments and blended finance. Having identified that it makes sense to invest in MPAs in Chapter 1 to increase the delivery of coastal protection (among other ES), and having identified ES beneficiaries with payment capacities of coastal protection in Chapter 4 we suggested a blended financing mechanism including the private sector.

Chapter 6 - **Coral reef restoration for coastal protection: Crafting technical and financial solutions**, utilizes outputs from Chapters 4 and 5 (taking into account scientific knowledge gaps and limits) further develops ecological and economic solutions for coastal protection. In this paper, we demonstrate the positive impact of restoring reefs, provide options for doing so, and show the additionality obtained from using such Nature-based Solutions over traditional gray infrastructure for mitigating coastal erosion.

Chapter 7 - **Private capital to improve tropical coastal protection: time for a boost**, synthesizes the outputs of the preceding chapters and concludes that while restoration is not a perfect solution, it is perhaps our best chance at improving reef health for coastal protection. The burden of financing such solutions – which vary considerably in cost - should not rest solely with governments, and should be shared with the private sector, especially those who directly benefit from coastal protection.

Outputs from the research, show that in spite of ecological complexities associated with coastal protection, private investment in coral restoration is promising as a means of contributing to service flow. Advances in ecology and engineering are steadily allowing for the role of coral reefs in coastal protection to be identified and quantified, while NbS interventions are being refined and discovered, to improve reef health and increase wave attenuation. At the same time, the private investment sector is undergoing a “sea change” with investment priorities shifting from a strong reliance on financial returns, to an acceptance of lower financial returns, once environmental and social returns are also achieved. Mechanisms exist in both the environmental and financial realms, to reduce risk and promote investment of NbS.

Specific interventions both in terms of NbS and financing mechanisms have been identified in the thesis. As the technical solutions advance, there also needs to be improved dialogue between the investment and conservation worlds. Money is available in the private sector to help with the conservation of coral reefs for coastal protection and other ecosystem services. We are on the right trajectory, with a common desire to improve our environmental impact. In this UN Decade on

Ecosystem Restoration, it is the right time to fully engage the private sector in investing in ecosystem protection.

Résumé Français

Protection côtière: approche écologiques et financières pour améliorer la santé des récifs coralliens et le service écosystémique associé.

Les récifs coralliens déclinent à l'échelle mondiale en raison d'impacts locaux et mondiaux, exacerbés par des financements inadéquats pour leur gestion.

La protection côtière est un service écosystémique (SE) important rendu par les récifs coralliens, en particulier avec des impacts négatifs croissants du changement climatique mondial. Ce SE n'a cependant pas retenu la même attention que d'autres SE, peut-être en raison de difficultés à déterminer clairement le rôle du récif corallien dans la fourniture du service. Malgré ces défis, la contribution des récifs à la prestation de services a pu être démontrée dans la littérature scientifique.

Nous sommes dans un contexte où la situation est préoccupante avec des niveaux sans précédent de mortalité corallienne et qui semblent s'accélérer. Afin de lutter contre le déclin des coraux, il est impératif que nous déterminions les méthodes les plus adaptées pour améliorer la santé des récifs. Un élément essentiel de toute solution est aussi de savoir comment la financer, à un moment où les méthodes de financement traditionnelles échouent et où le manque de financement a été

identifiée comme un obstacle majeur au succès de la conservation de la nature (affectant particulièrement les pays en développement).

Cette thèse étudie des solutions pour mieux comprendre, quantifier et potentiellement optimiser le SE de protection du littoral rendus par les récifs coralliens. Le travail explore aussi le rôle du secteur privé, comme bénéficiaire du SE, dans le financement des interventions.

L'objectif de cette thèse est d'étudier des solutions pour améliorer la protection côtière et d'explorer le rôle du secteur privé dans le financement des interventions. Nous croisons les disciplines de la science, de l'ingénierie et de l'économie dans notre quête de solutions. Les questions abordées se répartissaient en 2 catégories : i. Solutions écologiques et ii. Solutions économiques. Les solutions écologiques ont été déclinées en interventions de gestion susceptibles de servir les flux de services de protection du littoral, tandis que Solutions économiques ont consisté à trouver et à évaluer les structures de financement adaptées à la fois à la protection du littoral et à l'investissement privé.

La thèse est divisée en 4 sections spécifiques, chacune après le chapitre d'introduction, comprenant 1 ou 2 articles.. Les résultats de chaque section s'appuient les uns sur les autres pour fournir des solutions. Les sections ont des titres généraux, qui sont les suivants :

- i. Introduction
- ii. Informer et convaincre - Chapitres 2 et 3
- iii. Rapport qualité/prix - Chapitres 4 et 5

iv. Que payons-nous ? Chapitre 6

v. La voie à suivre - Chapitre 7

Les 7 chapitres suivent l'ordre suivant :

Chapitre 1 - Fournit une introduction générale et le contexte de la thèse, en présentant les éléments essentiels de la protection, de la gestion et du financement de la conservation du littoral, y compris la justification des sélections effectuées.

Chapitre 2 – Article: « Bénéfices économiques des investissements publics dans les aires marines protégées (AMPs) ». L'article plante le « décor » des chapitres suivants, en proposant un cadre pour l'analyse écologique et économique des services écosystémiques des récifs coralliens. Dans cet article, nous avons mené des analyses coûts-avantages sur la gestion des récifs coralliens dans les aires marines protégées de 2 nations insulaires. La protection du littoral a été identifiée comme l'un des SE les plus importants pour les propriétaires immobiliers et l'industrie du tourisme.

Résumé: Les AMPs améliorent certains services écosystémiques (SEs) fournis par les récifs coralliens. Des évaluations claires et solides de ces impacts peuvent aider à améliorer le soutien des parties prenantes et à mieux informer les décideurs. Conformément à cet objectif, des analyses coûts-avantages (ACA) des AMPs dans 2 contextes différents ont été analysées : une AMP communautaire à faible pression touristique à Vanuatu, et une AMP gérée par le gouvernement avec une pression touristique relativement élevée, à Saint-Martin. Des évaluations ont été faites sur six SEs : la biomasse des poissons, la beauté des paysages, la protection contre l'érosion

côtière, les valeurs de legs et d'existence, le capital social et la séquestration du CO₂, qui ont été quantifiés par différentes approches qui comprenaient la pêche expérimentale, les enquêtes et le transfert des bénéfiques. Les coûts d'exploitation totaux pour chaque AMP ont été collectés et le rapport avantages-coûts et le retour sur investissement basés sur des projections actualisées sur 25 ans ont été calculés. Des analyses de sensibilité ont été réalisées sur les impacts des AMP, et les taux d'actualisation (5, 7 et 10%).

Les indicateurs de retours sur investissement public ont tous montré des résultats positifs, l'impact sur le SE de l'éco-tourisme étant le plus important estimé pour toutes les AMP, soulignant l'importance de cette relation. L'étude a également démontré une sensibilité relativement élevée des résultats à différents niveaux d'impacts sur les SEs, ce qui souligne la nécessité de réduire les lacunes dans les connaissances scientifiques.

Chapitre 3 - Article: « Évaluation économique du service écosystémique des récifs coralliens de protection côtière : une approche pragmatique ». L'article fournit une méthodologie d'évaluation plus précise de la protection côtière fournie par les récifs coralliens, en particulier pour les pays où les données sont peu disponibles.

Résumé: L'identification et l'évaluation économique de SEs deviennent des composantes importantes pour la gestion des récifs coralliens. Dans de nombreux contextes, la protection des actifs humains contre les inondations côtières est l'un de SEs les plus importants fournis par les récifs coralliens. Les méthodes utilisées pour caractériser ce SE doivent pouvoir s'adapter aux situations où les données sont peu

disponibles, sans sacrifier la robustesse. Dans cet article, nous suggérons une approche qui utilise l'opinion d'experts et ne nécessite pas de quantités importantes de données. Notre objectif principal est de trouver un équilibre entre des modèles simples et complexes qui peuvent être utilisés dans un environnement où les données sont rares, pour produire une évaluation économique de l'ES des récifs coralliens de protection contre les inondations côtières. L'approche comporte trois étapes : (i) identifier les zones géographiques et les actifs à risque, (ii) identifier le rôle contributif des récifs coralliens dans la protection des côtes et, (iii) évaluer les coûts annuels de réparation des actifs à travers l'approche du coût des dommages évités .

La méthode proposée paraît appropriée pour le plaidoyer auprès des décideurs politiques, mais semble être moins efficace pour les approches à petite échelle, telles que celles requises pour les négociations sur le paiement des SE ou la planification spatiale marine.

Chapitre 4 - Article: « Sous quelles conditions le paiement pour les services écosystémiques convient-il à la protection côtière dérivée des récifs coralliens ? : un examen des exigences scientifiques». L'article examine la fourniture par les récifs coralliens de la protection côtière et détermine sa faisabilité pour un paiement pour les services écosystémiques (PSE). Dans le processus, le rôle du corail vivant a été analysé et des actions de gestion identifiées qui pourraient améliorer la santé des récifs pour la fourniture de services. Ce document identifie la restauration des coraux comme une intervention clé pour la protection côtière, identifie les principaux bénéficiaires et fournit la justification économique des chapitres 5 et 6.

Résumé - Le paiement pour les services écosystémiques (PSE) est un outil émergent destiné à résoudre une série d'inefficacités de la gestion des écosystèmes en liant l'action de conservation au paiement. De tels programmes n'ont pas été testés à notre connaissance, pour la protection côtière dérivée des récifs coralliens, qui est un service écosystémique (SE) clé pour de nombreux pays bordés par des récifs coralliens tropicaux. La santé des coraux à l'échelle mondiale se détériore, tout comme leur SEs et un financement inadéquat est identifié comme un facteur transversal qui entrave l'action de gestion. Dans cet article, nous avons évalué la faisabilité des PSE pour la protection côtière, en mettant l'accent sur les exigences scientifiques. Les éléments clés des PSE liés uniquement aux processus écologiques ont été isolés, le rôle des récifs coralliens dans la protection des plages a été examiné et les options de gestion prioritaires pour l'amélioration de la santé des récifs ont été synthétisées.

Les résultats indiquent qu'il reste beaucoup de manques de connaissance, mais, que cependant il existe des connaissances scientifiques adéquates pour satisfaire un PSE. Bien qu'il y ait une capacité limitée à prouver et à quantifier la causalité entre les actions de gestion et la prestation de SE, les critères de PSE peuvent être satisfaits avec la substitution d'un proxy de gestion, plutôt que les paiements étant conditionnés aux mesures de SE. La gestion, à la fois passive et active, se concentrerait sur le maintien des récifs qui ont déjà une fonction de protection et des plages avant stables, au-dessus d'un seuil de fonctionnement.

Chapitre 5 - Article: « Investissement avec impact dans la conservation marine » .
L'article, explore plus en détail les moyens innovants de financer la conservation marine via des investissements d'impact et des financements mixtes.

Résumé : Les AMPs sont des exemples prometteurs de solutions fondées sur la nature qui peuvent protéger la diversité tout en fournissant des services écosystémiques. Cependant, le financement insuffisant pour une gestion efficace et l'expansion des AMP reste un défi qui affecte particulièrement les pays en développement. Au cours des dix dernières années, une communauté d'investisseurs à la recherche d'impacts sociaux et environnementaux positifs en plus de rendements financiers, est intervenue pour aider à combler ce déficit de financement de la conservation marine. Une approche innovante de gestion collaborative a été récemment mise en place en République dominicaine pour l'une des plus grandes AMPs des Caraïbes. Des solutions de financement mixte associant financement catalytique, de développement et d'impact ont été utilisées pour couvrir les besoins en capital initiaux. Les revenus des AMP sont générés pour la gestion des AMP et les retours des investisseurs, via une gamme d'outils de financement durable, y compris les droits d'entrée par les visiteurs. La solution offre des résultats intéressants qui utilisent des financements catalytiques et de développement pour mobiliser des financements à impact commercial dans les AMP. Du point de vue du gouvernement, l'approche fournit des preuves empiriques de la manière dont le financement non public peut faire partie des options de financement pour le réseau d'AMP d'un pays, réduisant ainsi la charge financière sur les budgets publics. La replicabilité de l'approche semble cependant limitée par le nombre d'AMP avec des modèles de revenus adéquats.

Chapitre 6 - Article: « Restauration des récifs coralliens pour la protection côtière : Élaboration de solutions techniques et financières, développement de solutions écologiques et économiques pour la protection côtière ». Dans cet article, nous démontrons l'impact positif de la restauration des récifs, proposons des options et montrons l'additionnalité obtenue en comparant ces solutions (basées sur la nature) avec des infrastructures grises traditionnelles.

Résumé - L'érosion côtière, aggravée par la mortalité des récifs coralliens, est un problème majeur pour les petits États insulaires en développement. Les infrastructures traditionnelles « grises », telles que les récifs artificiels, financées par les budgets publics, ont été utilisées pour lutter contre la disparition des plages . Nous avons examiné si trois solutions basées sur la nature (NbS) pouvaient produire des résultats positifs pour la protection côtière et encourager le partage des coûts pour la conservation des récifs, avec des bénéficiaires privés.

Ces trois solutions basées sur la nature (NbS) sont :

- (i) restauration du corail (solution « verte »)
- (ii) restauration du corail + infrastructure calcaire (solution hybride verte-grise)
et,
- (iii) restauration + infrastructure béton imprimée en 3D (solution hybride verte-grise)

Nous avons modélisé l'impact de la restauration sur l'atténuation des vagues sur deux récifs de la Barbade et simulé les coûts initiaux et de maintenance sur une période de 25 ans. Toutes les solutions offrent une additionnalité par rapport à l'infrastructure

grise seule, en particulier pour atténuer les effets de l'élévation attendue du niveau de la mer.

La restauration reste la moins coûteuse avec cependant le risque d'échec le plus élevé car l'ingénierie écologique n'est pas encore suffisamment consolidée.

Les solutions hybrides sont moins risquées que la restauration seule car, par l'infrastructure grise, elles offrent une atténuation immédiate des vagues, ainsi que des services complémentaires tels qu'une attractivité accrue due à la présence de poissons de récif. Leurs coûts sont cependant entre +80% et +450% supérieurs aux solutions grises seules.

Bien que cela puisse initialement dissuader l'utilisation des SfN, le financement mixte et les paiements pour les services écosystémiques, dans certains cas, pourraient offrir une solution à cette situation. En effet les gouvernements et les bénéficiaires privés pourraient bénéficier de nouvelles options pour le financement initial et pour partager les coûts, avec en fin de compte la possibilité d'atténuer les effets de l'élévation attendue du niveau de la mer .

Chapitre 7 – Article « Des capitaux privés pour améliorer la protection des côtes tropicales : le moment d'un coup de pouce? » , synthétise les résultats des chapitres précédents et conclut que la restauration des récifs financée par des capitaux privés est peut-être une opportunité d'améliorer la santé des récifs pour la protection des côtes.

Résumé : La protection côtière, un important service écosystémique des récifs coralliens, est menacée par l'augmentation de la mortalité corallienne, exacerbée par le changement climatique mondial. Les solutions basées sur la nature sous la forme de restauration des coraux, bien qu'imparfaites, peuvent aider à reconstruire la structure des récifs et à améliorer le flux de service pour certains sites. Avec un manque de financement existant entre ce qui est requis pour la conservation et ce qui est accessible, les investisseurs privés devraient jouer un rôle plus important dans ces activités de restauration. D'autant plus que les hôteliers côtiers, en particulier, bénéficient de la stabilité des plages et ont également un potentiel de génération de revenus supplémentaires avec des récifs sains. Les solutions de financement mixte, en particulier, sont particulièrement adaptées à la restauration mixte basées sur des infrastructures grises-vertes (récifs en béton et restauration corallienne). De leur côté, les paiements pour les services écosystémiques semblent plus adaptés au programme seul de restauration corallienne. Les praticiens de la conservation et de la finance doivent s'engager davantage et comprendre les mondes de chacun, afin que ces sources privées soient efficacement recherchées et utilisées.

Les résultats de la recherche montrent qu'en dépit des complexités écologiques associées à la protection côtière, l'investissement privé dans la restauration des coraux est prometteur en tant que moyen de contribuer au flux de service. Les progrès de l'écologie et de l'ingénierie permettent régulièrement d'identifier et de quantifier le rôle des récifs coralliens dans la protection côtière, tandis que les interventions NbS sont affinées et découvertes, pour améliorer la santé des récifs et augmenter l'atténuation des vagues. Dans le même temps, le secteur de l'investissement privé subit un « changement radical », les priorités d'investissement

passant d'une forte dépendance à l'égard des rendements financiers à une acceptation de rendements financiers inférieurs, une fois que les rendements environnementaux et sociaux sont également atteints. Des mécanismes existent dans les domaines environnementaux et financiers pour réduire les risques et promouvoir l'investissement dans le NbS.

Des interventions spécifiques tant en termes de SNB que de mécanismes de financement ont été identifiées dans la thèse. Au fur et à mesure que les solutions techniques avancent, il faut également améliorer le dialogue entre les mondes de l'investissement et de la conservation. L'argent est disponible dans le secteur privé pour aider à la conservation des récifs coralliens pour la protection des côtes et d'autres services écosystémiques. Nous sommes sur la bonne trajectoire, avec une volonté commune d'améliorer notre impact environnemental. En cette Décennie des Nations Unies pour la restauration des écosystèmes, c'est le bon moment pour engager pleinement le secteur privé à investir dans la protection des écosystèmes.

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Chapter 1. General Introduction

Coral reefs are among the most biodiverse and economically important ecosystems in the world (Hoegh-Guldberg et al., 2019) providing an array of Ecosystem Services (ES), on which humans, especially those who live on small islands, depend. Of these services, coastal protection is among the most important for island nations, due to the ability of coral reefs to reduce risk of current and impending Global Climate Change (GCC) impacts. Reefs are however in jeopardy, (Vercelloni et al., 2020, Kleypas et al., 2021), with reports of over 50% coral being lost within the past 30 years (NASEM, 2019). In the Caribbean, 90% of coral reefs are predicted to fall into the category of “threatened” by 2030 and 100% by 2050 (Burke et al., 2011). “Business as usual” is failing, both in terms of management interventions (Rinkevich, 2008) and conservation finance (Iyer et al., 2018).

Effective and sustainable means must be found to restore reef health and finance the interventions quickly, as it is estimated that there are less than 50 years remaining in which we can mitigate and adapt to a changing climate (Kleypas et al., 2021). Predictions (very high confidence) have been made of increasing risks to “people, assets, economies and ecosystems” due to ocean warming, storms and sea level rise, among other extreme events (IPCC, 2014, IPCC, 2021). The risk of flooding and erosion is already increasing due to coastal storms (Wong et al., 2014) and tropical Small Island Developing States (SIDS) are among the most vulnerable to Climate Change impacts due to limited land space, dependence on tourism and fisheries and exposure to hazards (Nurse et al., 2014). With the ability to reduce coastal inundation and minimize beach erosion, coastal protection, becomes an increasingly important service for disaster management.

Taking into consideration the major benefits received by the private sector from coral reef services (Brander et al., 2007, van Beukering et al., 2007), beneficiaries, could and should be fully involved in protecting the ecosystems on their businesses are built. Sensible financing mechanisms therefore have to be developed in order to ensure returns are made from ES. To date only relatively tentative steps have been made in conservation by this sector (Huwyler et al., 2014).

This chapter is organized as follows: Context and Background are presented in section 1.1, Aim & Structure of thesis in 1.2 and Summary of chapters in 1.3.

1.1 Background

The core element of this thesis is the coral reef, an extremely complex ecosystem, with high levels of diversity among species and habitats and an array of interactions involved in service provision (Dikou, 2010, Brandl et al., 2019). The structural foundation of these massive, biogenic structures, is created by hard (scleractinian) corals achieved via biomineralization - the deposition of vast quantities of Calcium carbonate skeleton (Weis and Allemand, 2009). The resulting porous, wave resisting, limestone framework, provides habitats for a profusion of life, each with varied roles in maintaining a healthy, functioning coral reef. Fish in particular, (with different life histories and ecologies) have many functional roles on reefs and therefore support a dizzying variety of processes on the reefs (Dikou 2010). Herbivores, are especially important as they, facilitate coral growth and settlement by grazing on algae , which would otherwise overgrow, outcompete and/or provide a haven for deleterious bacteria (Jackson et al., 2001, Mumby et al., 2016). Without the hard corals, however, there would be no structure to support the other organisms and thus, all ecosystem services provided, are directly attributed to these reef builders (e.g. Johannes, 1975;

Done et al., 1996). Increasing hard coral abundance, will therefore be our management focus.

1.1.1 Coastal Protection

This service was selected, due to its importance in mitigating current (e.g. increasing intensity and frequency of storms) and predicted (e.g. Sea Level Rise) GCC impacts (Wells and Ravilious, 2006, World Bank, 2016, Hein et al., 2020) coupled with the relative lack of studies compared to fisheries and aesthetics (Principe et al., 2012). Coastal protection is not as obvious as such coral reef ES, where the service is provided in close proximity to the ecosystem itself. Wave breaking is easily observed on the reef, but the ability of reefs to attenuate or absorb wave energy, is unseen and generally not acknowledged, at least until reef deterioration results in beach erosion (Zepeda-Centeno et al., 2018).

Coral reefs are natural wave dampeners, and the first line of defense against flooding and erosion (Reguero et al., 2018). In so doing, they protect human lives and property, beach related economic interests (Barbier et al., 2011) and act as risk reduction and adaptation strategies (Reguero et al., 2018). The service is complex and reliant on a number of elements including: other marine ecosystems, seasons, tides, bathymetry and oceanographic conditions, as well as reef type, rugosity, coral cover and depth. Reefs however have been reported to play key roles and global estimates have projected that damage from floods would double and the costs from frequent storms would triple, without their presence (Beck et al 2018).

1.1.2 Management

In order to determine effective management strategies for improving reef health to protect coasts, an understanding is required of their ecology and the underlying processes required for healthy reefs. The ability to identify how the ecosystem responds to negative impacts is challenging (Steneck et al., 2018), however, coral reefs are reported to be the most studied marine ecosystems and in spite of their complexity, scientists have a good understanding of major issues and management solutions (Mumby and Steneck, 2008). Determining which management measures can aid in the provision of coastal protection, will be key to unlocking the finance required to do so.

Marine Protected Areas (MPAs), are sections of ocean where governments have placed limits on human activities to protect marine habitats (Bohorquez et al., 2019). While not perfect (Bruno et al., 2019) MPAs have been recognized as the best framework in which effective management can be carried out (Mumby and Harborne, 2010, Perry et al., 2015, Laffoley et al., 2019, Mcleod et al., 2019). MPAs are also important in terms of financing conservation. The sea is a common resource available to all, to use, and to harm. There is less incentive for the private sector to invest in conservation action in a space without clear boundaries, or control (World Bank, 2012). MPAs however, are legally instruments designated by governments, with identified management authorities, and within such frameworks, a type of ownership and responsibility is achieved.

Many of the actions associated with coral reef management can be described as passive management, which rely on mitigating negative impacts (Rinkevich, 2008).

However these measures have not had overwhelming success and it is unlikely that coral reefs will recover without human intervention – active management – (Hobbs and Cramer, 2008, Rinkevich, 2014). Active techniques and in particular, reef restoration have been suggested as ways to complement existing management interventions (Zepeda-Centeno et al., 2018, Hein et al., 2020). Such management interventions can be categorized as Nature-based Solutions, which improve ecosystem health while delivering tangible and sustainable benefits for communities (IUCN, 2020). Both active and passive management interventions will be examined.

1.1.3 The Financing gap

Financing gaps have been identified as major obstacles to the management of all biodiversity (Milnea and Christie, 2005) and hence the flow of ES. Globally, marine conservation is severely underfunded (Lennox, 2012) with the situation being worse in the tropics (Balmford and Whitten, 2003, Gill et al., 2017). Studies have shown that the total funding for protected areas and biodiversity conservation must be increased dramatically to achieve the targets set at national or international levels (Brunner et al., 2004, Johansen and Vestvik, 2020). As the biodiversity slide continues, the needs for resource management are increasing in a world fraught with increasing challenges such as Global Climate Change. New revenue streams are needed, which must be sustainable and steady; hence the call from the Convention of Biological Diversity (CBD) and Aichi target #20 to urgently explore new and innovative financial mechanisms at all levels of marine conservation management. According to the Secretariat of the Convention of Biological Diversity (CBD) (Convention on Biological Diversity, 2011), long- term, reliable sources of market financing for biodiversity

conservation must be established and strengthened in order to increase funding to support the three objectives of the Convention¹.

While there is no global definition of innovative finance, it is taken to mean a move away from traditional finance, to the emerging market based schemes (Agardi and Pascal 2014). In the marine environment, traditional finance is often provided by governments, Official Development Assistance or Philanthropic individuals (Lennox, 2012) which have still resulted in a large financing gap (Clark et al., 2013, Salamon, 2014). Government funds in particular are almost completely dependent on allocations from the national budget, an arrangement that is often tenuous, with the budget or objectives being affected by changes in government policies and priorities. Oftentimes the duration of government funding is influenced by political cycles of 1-5 years (Claudiu-Gabriel and Claudiu, 2002) which is typically too short a time for conservation objectives to be met (Gurney et al., 2014). Grants, are necessary but often are not sufficient to meet conservation objectives (Bos et al 2014).

Innovative finance mechanisms vary in their approach, amount of revenue generated, timing of funds release and whether an incentive for behavioural change is provided, however they all rely on an understanding of the benefits that flow from nature to humans and an ability to market them (Balmford and Whitten, 2003, Agardy et al., 2014). Many mechanisms are rooted in the idea that those who benefit from the services provided by nature, should participate in its conservation. However, to date,

¹ the conservation of biodiversity; sustainable use of the components of biodiversity; and sharing the **benefits** arising from the commercial and other utilisation of genetic resources in a fair and equitable way <https://www.cbd.int/2011-2020/about/goals>

compared to their terrestrial counterparts, the marine environment has received little attention (Ruckelshaus et al., 2015) and more research is required to develop appropriate mechanisms for this environment (Bos et al., 2015).

We assessed and selected a suite of non-traditional finance tools in “Finance Tools For Coral Reef Conservation: A Guide² (Iyer et al., 2018) for suitability as well as Marine Conservation Agreements (The Nature Conservancy and Conservation International, 2012) and Blended finance (Küblböck and Grohs, 2019). Tools selected: Conservation Trust Funds (CTF), Biodiversity Offsets (BO), Payment for Ecosystem Services (PES) Impact Investment (II) and Blended Finance (BF) (Table 1) range from those in which no revenue is generated, to those that generate revenue by providing incentives (positive³ and negative⁴) and those that provide incentives without generating revenue, with their associated investors and investees. They further: i. are designed to provide positive impact; ii. funding duration does not rely on government political cycles for success; iii *primary* decision makers are limited to the private sector partner and the conservation authority – green taxes for example would require extensive government involvement and the concomitant complexities involved in such an engagement.

² Led by the Conservation Finance Alliance “the leading professional association for conservation finance experts and practitioners” <https://www.conservationfinancealliance.org/>

³ Rewards for marine conservation e.g. payments for conservation actions

⁴ Punishments for negative impacts e.g. biodiversity offsets

Table 1: Non- Traditional Finance Tools

MECHANISM	DESCRIPTION
Blended finance (BF)	Capital of different risk levels combined to achieve the objectives of all, whether they be environmental, social or financial (Küblböck and Grohs, 2019)
Biodiversity Offsets (BO)	Used in infrastructural projects to allow for no net loss (or gain) of biodiversity (BBOP Forest Trends 2019)
Conservation Trust Funds (CTF)	Private, legally independent institutions that provide sustainable financing for biodiversity conservation. Utilising resources from different sources (international donors, governments and the private sectors) into a fund which then provides grants to stakeholders for conservation action. (Spergel and Mikitin 2013)
Impact Investment (II)	Investments for positive change, designed to generate "positive, measurable social and environmental impact, alongside a financial return" (GIIN n.d.)
Marine Conservation Agreements (MCA)	Any formal or informal contractual arrangement that aims to achieve ocean or coastal conservation goals in which one or more parties (usually right-holders) voluntarily commit to taking certain actions, refraining from certain actions, or transferring certain rights and responsibilities in exchange for one or more other parties (usually conservation-oriented entities) voluntarily committing to deliver explicit (direct or indirect) economic incentives (TNC & CI 2012)
Payment for Ecosystem Services (PES)	A system which provides incentives for behavioural change where those who contribute to producing the service are compensated by those who benefit from it (Lau 2013)

From this list, PES, Impact Investments & Blended finance were selected for further investigation, as they:

1. Provide uninterrupted funding - unlike CTF's delivery of grants to stakeholders for specific action. This sporadic funding is appropriate for specific management objectives but not for overall management which could involve payment to staff for example.
2. Are not reliant on a specific damage event, as in the case of BOs which are activated only when there is a damage event .
3. Provide financial incentive for improvements. MCAs are similar to PES in that ES beneficiaries pay those with management responsibilities for conservation of the

service, however, unlike PES, these payments are not conditional on success, simply that management is provided

4. Novel elements – for PES there is conditionality (pt.3 and described in Chapter 4) and impact investment- blended finance, introduces a relatively new concept of debt, for conservation finance (described in chapter 7).

1.2 Aim and Structure of the Thesis

Using state-of-the-art scientific knowledge about the ES of coastal protection provided by coral reefs, as well as a review of the different interventions to enhance this ES in different locations, this thesis attempts to fill in some of the knowledge gaps and address enabling factors and challenges to implementing solutions for coastal protection.

The aim of this thesis is to **investigate solutions both ecological and economic for improving coral reef health, for protecting coasts**. In so doing, the authors will explore and provides insights around two primary queries: 1. What management interventions are most effective at improving the flow of the ecosystem service of coastal protection and 2. What financing structures are likely to attract private investment and via what mechanisms.

In so doing, we will determine:

1. The role of the coral reef in service delivery (Chapters 3, 4, 6), which encompasses the:
 - mechanisms via which reefs deliver the service
 - importance of live coral

- the identification of coral reefs responsible for service provision

2. Priority management interventions to improve the flow of the coastal protection ecosystem service (Chapters 2, 4)

3. Identification of beneficiaries (chapter 2, 5) and financing structures likely to attract private investment (Chapters 5, 6).

Chapter 2 “sets the scene” and shows that investment in the management of coral reef ES (CRES) can provide returns and for whom. Chapter 3 is a supporting paper which provides a methodology for valuing coastal protection, expected to be a cross-cutting theme of the thesis and raises elements for further investigation. Chapter 4 reviews what is known about coral reef provision of coastal protection and determines its feasibility for a Payment for Ecosystem Services (PES) scheme. Chapter 5 further explores non-public means of financing marine conservation via impact investments and blended finance. Chapter 6 utilizes outputs from Chapters 4 and 5 and develops ecological and financial solutions for coastal protection, while Chapter 7 synthesizes the outputs of the preceding chapters.

1.3 Chapter Summaries

Chapter 2 Evidence of economic benefits for public investment in MPAs

The paper examines the economic benefits and costs of investment in the management of coral reefs. The study was conducted in Vanuatu and St. Martin, chosen to be representative of small island states. MPAs are accepted as important management tools. The study analyzed 6 coral reef ecosystem services including

coastal protection and identified the main benefits, costs and beneficiaries of the management intervention. Cost Benefit Analysis (CBA) was used as a means of demonstrating the economic sense of improving ES through establishing MPAs. The importance of coastal protection and its beneficiaries were identified (especially for St Martin where real estate development is more consolidated). Further, the potential and limits of valuation for designing impact investment solutions and Payment for Ecosystem Services were discussed.

Chapter 3: Economic valuation of coral reef ecosystem service of coastal protection:

A pragmatic approach

This paper provides a method for quantifying the ES of coastal protection and an economic valuation in data poor environments (as is the case in many island nations), without sacrificing robustness. The method utilized expert opinion and involved three steps: identifying the at risk areas, isolating the actual role that reefs play in coastal protection and valuing the loss of services via the avoided damage approach. Study areas ranged from small islands such as Tobago in the Caribbean to the Marine Bohol Triangle in the Philippines. The authors concluded that while this method is suitable for policy advocacy at a scale of an island nation, due to a lack of precision, it would be less effective for fine tuning instruments such as Payment for ES.

Chapter 4. When are Payment for Ecosystems Services suitable for Coral Reef derived coastal protection?: a review of scientific requirements

An intensive review of PES was carried out to determine its suitability for coastal protection taking into account ecological science knowledge gaps. In so doing, key PES elements were isolated and the role of the reef in service delivery, importance of

living coral and priority management interventions explored. Outputs suggested that there is adequate scientific knowledge for PES to be utilized. Further living coral is important, in the present, and future, for on-going reef maintenance as well as their ability to keep up with expected SLR. Both passive and active management interventions can be financed via PES.

Chapter 5. Impact Investment in marine conservation

An innovative blended finance solution, mixing private, philanthropic and public sources implemented for marine conservation (through an MPA) in the Dominican Republic) was investigated. The enabling factors required to unlock private capital for marine conservation are described and discussed. Outputs demonstrate that blended finance can be used to improve reef health when a specific legal, financial and governance framework is in place. This broader analysis set the stage for designing instruments with the private sector for the ES of coastal protection.

Chapter 6. Coral reef restoration for coastal protection: Crafting technical and financial solutions

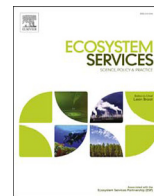
Three types of coral reef restoration - 1 green and 2 gray-green (hybrid) interventions were investigated on a conceptual level to determine if they could improve service flow for coastal protection. We modelled the impact of restoration of 2 reefs on service flow for coastal protection. Numerical models were run on fringing reefs off Barbados, (chosen as representative of small islands). Up-front and maintenance costs were simulated over a 25 year period for each solution. Outputs show that all 3 solutions contribute to maintaining beaches and deliver increased benefits to specific private sector beneficiaries, especially in terms of Sea Level Rise, when compared to

fully grey solutions. Further, PES seems suitable for financing the fully green solution, while impacts investments, blended with grants can finance the costlier hybrid interventions.

Chapter 7. Private capital to improve tropical coastal protection: time for a boost.

This Commentary paper synthesized the outputs of the preceding papers. The authors concluded that while restoration is not perfect, it is perhaps our best chance at improving reef health for coastal protection (no conditions? E.g. On previous reasons of coral reef loss?). The burden of financing such solutions – which vary considerably in cost – should not rest solely with governments, but must also include the private sector, who directly benefit from coastal protection. We suggested the aforementioned financing mechanisms (Chapter 6) to allow the flow of finance from private investors to conservation practitioners. We further suggested improved communication to improve awareness between both groups, that could lead to a “sea change” in thinking about conservation finance.

Chapter 2: Evidence of economic benefits for public investment in MPAs



Evidence of economic benefits for public investment in MPAs

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ABSTRACT

MPAs enhance some of the Ecosystem Services (ES) provided by coral reefs and clear, robust valuations of these impacts may help to improve stakeholder support and better inform decision-makers. Pursuant to this goal, Cost-Benefit Analyses (CBA) of MPAs in 2 different contexts were analysed: a community based MPA with low tourism pressure in Vanuatu, and a government managed MPA with relatively high tourism pressure, in Saint Martin. Assessments were made on six ES: fish biomass, scenic beauty, protection against coastal erosion, bequest and existence values, social capital and CO₂ sequestration, which were quantified via different approaches that included experimental fishery, surveys and benefit transfer. Total operating costs for each MPA were collected and the benefit-cost ratio and return on investment based on 25-year discounted projections computed. Sensitivity analyses were conducted on MPA impacts, and discount rates (5%, 7% and 10%). The investment indicators all showed positive results with the impact on the tourism ES being the largest estimated for all MPAs, highlighting the importance of this relationship. The study also demonstrated a relatively high sensitivity of the results to different levels of impacts on ES, which highlights the need for reducing scientific knowledge gaps.

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1. Introduction

Marine protected areas (MPAs) are clearly defined geographical spaces which are recognised and managed through legal or other effective means, to achieve long-term conservation of nature with associated ecosystem services and cultural values (IUCN-WCPA, 2008). MPAs are considered to be effective tools in managing coral reef ecosystems for ecological objectives (Salm et al., 2000, Halpern, 2003). The ecological benefits of MPAs on fish and invertebrate resources inside the boundaries of reserves, as well as in the surrounding areas, are now well established (Sale et al., 2005). Several international policy fora have called for the global expansion of MPAs. The predominant statements are from the Durban Action Plan and the Convention on Biological Diversity (CBD) Aichi Target 11. More recently, the 'Promise of Sydney' statement issued by the 2014 World Parks Congress recommends that we "urgently increase the ocean area that is effectively and equitably managed in ecologically representative and well-connected sys-

tems of MPAs or other effective conservation measures", with an ultimate aim of creating a fully sustainable ocean, with at least 30% set aside for no extraction. While some progress has been made towards meeting these targets, considerably more needs to be done in order to ensure the expanded geographic coverage as well as effectiveness and ecological representativeness of MPAs (Ban et al., 2014, Dunn et al., 2014, Fox et al., 2014).

The economic benefits of investing in MPAs are derived from improvements in the quality of the natural environment, which strengthens the capacity of coastal and marine ecosystems to produce ecosystem services for local populations, businesses and the global community (TEEB, 2010a). On the cost side, the start-up and management costs of MPAs are among the major hurdles in gaining support for the initiation of MPAs worldwide (Balmford et al., 2004, Mora et al., 2006). The Cost Benefit Analysis (CBA) approach is a way to inform and convince stakeholders of the potential returns from investing in marine protection (Gowdy, 2007).

Determining the CBA of marine conservation projects such as MPAs however, is far from being a consolidated approach. On one hand, the approach faces similar issues as ES valuation exercises (Laurans et al., 2013) in terms of the methodological

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limitations of obtaining the required information on welfare derived from ES. In addition, CBA has to take into account major uncertainties regarding MPA impacts on fishery yields, tourism revenues, coastal protection and other ES (Boersma and Parrish, 1999, Wilkie et al., 2006).

We analyze the methods and results of two existing CBAs of MPAs. The first is a network of community-based MPAs in the Vanuatu Islands (South Pacific) while the second is a government managed MPA in Saint Martin (French West Indies). These cases represent two different geographic and institutional contexts. Sensitivity analyses are carried out on the parameters of each CBA to improve our understanding of the robustness of the results. We examine the usefulness of CBAs of MPAs for decision-making and discuss how this tool can be further developed.

2. Material and methods

2.1. Description of the study sites

The analysis examines two MPAs, one in Vanuatu and the other in Saint Martin. The MPA situated in Efate, Vanuatu (South-West Pacific) is comprised of a group of five small MPAs, which are independent from each other, but linked via a network. The MPA in Saint Martin (French West Indies) comprises one large MPA (Fig. 1). The main socio-ecological characteristics of the sites are described in Table 1. Each of the MPA sites fulfills the following criteria: (i) proximity of economic activities (mainly fishing, tourism and human settlements) (ii) active management and adequate enforcement of the MPA (by communities or government) for at least five years and, (iii) the reserve covers at least 10% of the fishing grounds.

2.2. Description of the selected MPA impacts

Six ES categories were chosen for analysis: (i) subsistence and commercial reef fishing (ES1, also named as ‘fishery’) (ii) scenic beauty and emblematic species for tourism (scuba diving, recreational boating, snorkeling tours, charters, etc.) and associated activities (accommodation, food, etc.) (ES2, also named as ‘tourism’), (iii) protection against coastal floods (ES3, also named as

‘coastal protection’), (iv) non-use values of the ecosystems (ES4, also named as ‘bequest value’), (v) social capital impacts, defined as the ability to encourage collective action through external assistance (ES5 can be considered as one of the many multidimensional aspects of the concept of ‘social capital’) and (vi) CO2 sequestration by mangroves (ES6, also named as ‘carbon sequestration’).

When several methods were available, preference was given to those producing the most conservative values (Beukering et al., 2007).

2.3. Quantification and valuation of ES

Quantification and valuation of these six categories of ES requires a variety of measurement methods (Table 2). The valuation methods were developed following recent guidance produced by The Economics of Ecosystems and Biodiversity (TEEB) (TEEB, 2010b) and other manuals (Beukering et al., 2007, Balmford et al., 2008, Defra, 2008). Although the methodology sought to be homogenous across sites, some modifications were needed to better suit the unique ecological and socioeconomic conditions of each scenario.

For ES1 in Vanuatu, the producer surplus (the difference between the amount the producer is willing to supply goods for and the actual amount received by him) for commercial fisheries was calculated from data collected from fishing logbooks over the course of 2 years ($n = 96$ h of spearfishing), interviews with fishermen ($n = 25$), experimental Catch Per Unit Effort (CPUE) (45 h of net-fishing) and household and income expense surveys in two villages ($n = 12$ households). Estimates of CPUE (e.g. kg of fish captured per hour of standard fishing effort for net and spearfishing), annual fishing effort (h.net^{-1} or $\text{h.spearfishing}^{-1}$), protein content and fishery revenues were then produced. The value for subsistence fishing was produced by converting the amount of fish caught to the monetary value of an equivalent amount of animal protein from a different source (e.g. chicken or pork). In Saint Martin, data were collected from fishers and cooperatives’ surveys ($n = 11$) conducted in 2012. The surveys allowed for the quantification of fishing frequency per gear type, target species and catch per trip, as well as investment and operational costs of the activity. For both sites, the fisheries are small scale with fishers

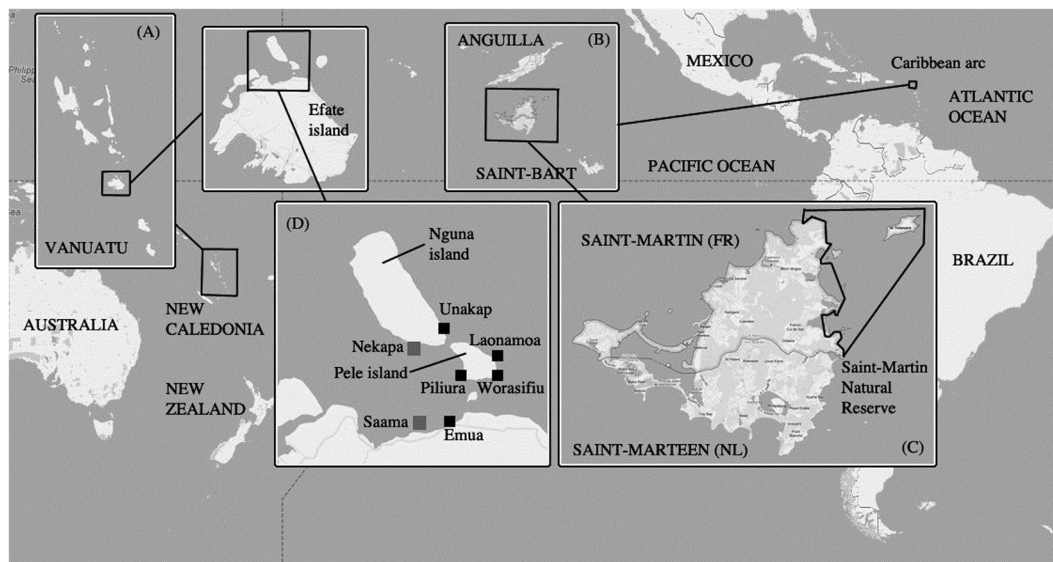


Fig. 1. Location of the study sites. NB: (A) Vanuatu, South Pacific; (B) North of Caribbean arc, composed of Anguilla, Saint-Barthélemy and Saint Martin island; (C) this latter island is divided in two parts with a French-speaking one in the North and a Dutch-speaking one in the South; the Saint Martin Natural Reserve (28 km²) is situated in the North East of the island. (D) The five MPAs in Vanuatu (< 1 km²) (represented by black squares) and the two control sites (represented by grey squares) are situated around the islands of Nguna, Pele and North Efate.

Table 1
General description of the six MPAs and control sites involved in the study.

	MPA sites							Control sites		Sources of data
	Emua	Piliura	Unakap	Laonamo	Worasifiu	Vanuatu	St Martin	Nekapa	Saama	
Resident population	240	110	90	250	50	740	45,000	110	130	VNSO (2009); INSEE (2010)
Demographic pressure on reef (hab/km ² of reef)	157	102	71	188	104	124	21,226	92	144	VNSO (2009); INSEE (2010)
Human density (hab.km ⁻²)	300	275	450	417	167		840	275	217	VNSO (2008); INSEE (2010)
Monthly average household incomes (monetary and non monetary) (PPP US\$, 2010)	335	261	272	294	294	291	671	306	318	Description in the text
Monthly average non monetary incomes (% total incomes)	31%	40%	40%	36%	36%		N.A.	36%	31%	VNSO (2008)
Fishing ground (km ²)	1,5	1,1	1,3	1,3	0,5	6	50,0	1,2	0,9	Direct observations
Tourism infrastructure (number of beds) in MPA perimeter	5	–	8	14	5	128	16,500	–	–	Direct observations; INSEE (2010)
Fishing pressure index	3,1	3	1,35	3,75	3,65	2,97	4,7	3,05	3,25	Description in the text
MPA creation date	2005	2003	2003	2003	2003	2003–2005	2001	–	–	Description in the text
MPA size (km ²)	0,24	0,13	0,12	0,14	0,13	0,75	27,96	–	–	

NB: The synthetic fishing pressure index was developed to compare the fishery effort potential between MPA sites and the control sites. The index is a score based on several characteristics of fishing effort standardized per km² of fishing ground for each site (number of gears, storage devices, “regular” fishermen and transport infrastructure).

Table 2
Description of the six main selected ES and MPA impacts involved in the study.

Ecosystem service	Valuation method	MPA contributing factor	Spatial perimeter	Estimated contributing factor as a% of ES value. Sensitivity analysis values in ()	
				Vanuatu	St Martin
Subsistence and commercial fishing	ES 1	Producer surplus	MPA impact on fishing productivity	Spillover and larval dispersion zone	25% (6–25–33%)
Tourism and associated expenses	ES 2	Producer surplus	Contribution in tourism expenses	MPA	70% (50–70–90%) 7% (5–7–10%)
Coastal protection	ES 3	Avoided damage costs	Impact on wave energy absorption mechanisms	MPA coastline	5% (3–5–10%)
Bequest value	ES 4	Transfer benefit	Impact on ecosystem existence	MPA	35% (20–35–50%) Not Available
Social capital	ES 5	Direct expenditures	Proportion of grants linked to the MPA	MPA	100%
Carbon sequestration	ES 6	Market value of carbon credits	MPA impact on avoided habitat conversion	MPA	10% (5–10–15%)

carrying out their activity in areas close to the MPAs. There is also a very limited fishing effort from “outsiders” in the studied fishing grounds.

For ES2, in both countries, the producer surplus corresponded to the value of the selected tourism activities directly related to the processes produced by coral reef ecosystems. The data collection method, however differed in each country. In Vanuatu, expenditure details and occupancy rates data were collected through interviews ($n = 45$), which were conducted on a monthly basis during 6 months (June to November). The majority of the tourism professionals were located in the interviewed study zone. Interviews with other professionals included all the diving clubs and most of the tour operators. Data were completed with official tourism statistics (Vanuatu National Statistics Office, 2008). In Saint Martin, expenditure estimates were based on business declarations of the number of their clients sent every month to the MPA by diving clubs and day tour operators ($n = 38$). These declarations were used to calculate the monthly fees collected from users. The associated average expenses on accommodation, food, local transport and souvenirs were estimated through existing tourism statistics (INSEE, 2008). Intermediary costs for the primary tourism businesses (service provider and accommodation sectors) were col-

lected via professional interviews and compared with costs from other studies. Data on recreational boating expenditures and added value were estimated based on available information of boat use frequencies in the MPA (Rastoin, 2011) and completed by interview ($n = 8$) with the main boating associations (sport fishing, harbours, yacht clubs) and professionals (sailing charters, brokers, maintenance operators).

For ES3 (coastal protection), the damage costs avoided by the presence of the protected ecosystems were assessed in a similar manner in both countries. The coastal areas potentially at risk from wave damage were identified, then the contribution of reefs and mangroves to the protection of vulnerable areas was estimated and finally the potential impacts on residential buildings and infrastructure were quantified and monetized, using the expected likelihood of a damaging event. Quantification of ES values utilised a bio-physical model developed for the valuation of ES in low-data availability situations (Burke et al., 2008; Pascal et al., 2016). The model allows us to define the coastal protection index (low-medium-high) of each segment of coastline with seven physical characteristics: coastal geomorphology, exposure of the coast, wave energy (usually the maximum wave height), frequency of storms, characteristics of coral reefs, coastal vegetation (man-

groves, wetlands, etc.) elevation and slope of the coast. Avoided damage valuation was assessed on potential residential and infrastructure damages. In vulnerable zones, the potential impacts were valued using geographic analysis. Urban and rural areas were differentiated and mean real estate prices collected from available data sources (home owners, real estate agencies, the internet).

For ES4, the benefit transfer method was used in Vanuatu based on existing studies that estimated bequest and option values of coral reefs for villages in Fiji (O'Garra, 2012). The socio-economic context of this study displayed similarities with Vanuatu, including customary management of marine tenure, small villages, small fringing reefs, and an emergent market economy in which subsistence, barter and market exchanges are mixed. The economic values were obtained from villagers through willingness to contribute (in hours/week) towards conservation of their marine zone, in order to maintain some of the ES (presented as option and bequest values by the author, even though it might be argued that the value also includes direct uses). In the case of Saint Martin, a review of selected non-use valuation studies in the Caribbean (Park et al., 2000, Rudd et al., 2001, Raboteur and Rodes, 2006, Parsons and Thur, 2007, Wielgus et al., 2009) failed to provide the conditions required for results to be used in a benefit transfer protocol (Rosenberger and Loomis, 2001, Wilson and Hoehn, 2006), and so these estimates were excluded.

In this context, social capital (ES5) refers to the capacity of the community to be represented in external events (Putnam, 2006, Barnes-Mauthe et al., 2015) and we chose to focus on one aspect of this capital, which is the ability to attract external assistance through grants or technical support. In both countries, data were collected from the census of all grants and assistance related to marine ecosystem management. Based on historical observations of past external assistance, a periodic flow (every 7 years) of support, after the creation of the MPA was applied.

ES6 (carbon sequestration) is primarily provided by mangroves and seagrass ecosystems (Sifleet et al., 2011). Results obtained were based on the methodology developed by several authors (Duarte and Middleburg, 2005, Bouillon et al., 2009, Murray et al., 2010, Nicholas Institute for Environmental Policy Solutions, 2011). Following the additional approach (Peters-Stanley et al., 2013), calculations were based on the volume of CO₂ eq. annually avoided from being released into the atmosphere, by maintaining ecosystems in their current state. Following our general preference for conservative values and safe-minimum estimates, we chose the value of market carbon credits rather than using the social cost of carbon (US EPA (U.S. Environmental Protection Agency) 2013). The prices of carbon credits were estimated from market data (Forest Trends Ecosystem Marketplace and Bloomberg New Energy Finance, 2014). Due to the limited presence of mangrove and seagrass habitats in the Vanuatu sites, this aspect of the valuation was only conducted in Saint Martin.

2.4. Valuation methods of impacts of MPAs on ES

The valuations of both MPAs focused on the identification of the factor of the MPA to the value of the ES, in order to isolate the MPA's effects. The control-impact approach has been proposed by several authors as a way to address this issue (Smith, 2002, Halpern, 2003). In control (non-MPA) and impact sites (MPA), the following studies were conducted: (i) a comparative assessment of the ecological factors using the Medium Scale Approach (Clua et al., 2006), specifically designed for commercial reef fisheries, (ii) a comparative assessment of fishing effort and (iii) a comparative assessment of tourism attractiveness resulting from objective indicators of site accessibility (e.g. distance to airport, infrastructure, price) for visitors. In Vanuatu, two villages served as control sites (Table 1). In Saint Martin however, it was not possible to

locate any valid control sites due to the relatively small area located outside of the MPA boundaries (the MPA covers approximately half of the island). Hereafter, we describe the details of the calculations for the MPA impact on ES.

An additional challenge of MPA valuation is whether the potential benefits of the MPA impacts will be experienced by the local community, the national private sector, or by international entities. This knowledge can help make the case for public or private investment. To achieve this a certain understanding of the spatial boundary of the MPA impacts is needed. This task can be difficult considering knowledge gaps of larval dispersion, species migration, and other factors related to the interconnectedness of marine ecosystems. In addition, the areas where beneficiaries are found may not coincide with the location of the impacts. A good example is the leakage effect for tourism benefits (Garrigós-Simón et al., 2015) showing the volume of tourism revenues that does not remain in the destination economy. To address this issue there needs to be not only an ecological understanding of the site but a socioeconomic understanding as well. This task has been confronted through the use of simplified assumptions and proxies described thereafter.

The MPAs' impacts on ES1 were calculated through the identification of the effect of MPAs on CPUEs for main gear type. The results were obtained from an experimental fishery and logbook data collection effort conducted in Vanuatu and described previously. Vanuatu's MPA sites demonstrate higher fish productivity than non-MPA locations. The estimated increase in catch productivity is between 15% and 33% (kg.h⁻¹) higher for gillnet and 6–22% (kg.h⁻¹) for speargun harvest techniques (Pascal, 2013). We assumed that this productivity enhancement represents a proxy of the MPA contributive factor on fishery added value (ES1). A direct gradient analysis (redundancy analysis: RDA and partial RDA) on data (Legendre and Anderson, 1999) was conducted to differentiate between the explanatory variables linked to the MPA (e.g. existence, distance from border) and those linked to the context (e.g. moon cycle, tide and habitat variables including depth, coral reef description and substrate category). The absence of "outsider" fisheries and the prevalence of subsistence fishing yield the assumption that the benefits are experienced by the local communities.

The effect of MPAs on tourism is reflected in the role of the reserve in visitor destination choice. Estimates were based on specific questions in individual surveys, conducted with end-users (Vanuatu $n = 34$ (Pascal, 2013), Saint Martin, $n = 110$ (Malterre P et al., 2011)). The respondents were asked to answer specific questions about the importance of the MPA and ecological attributes expected to be enhanced by the MPA (i.e. beauty of the sea scapes and the diversity and abundance of fish) in their destination choice. These questions provide information about a variable "MPA_factor". It takes a value between 0 and 1 following the proportion of time devoted to activities/MPA visit for the duration of the stay, the main motivation for travel and the importance of these activities in the choice of destination. factors, expressed as a percentage of the added value of tourism expenses (ES2), varied between 10% (Saint Martin) and 70% (for some sites in Vanuatu). They were both completed with studies of tourism Advertising Image Analysis (AIA) (Hajkowicz et al., 2005) based on a minimum of 100 images and identifying the weight of images or references made specifically to the MPA or the marine biodiversity in regard to other categories of tourism attributes (cultural and people, ecosystems and landscapes, beaches, other forms of entertainment). They highlighted that marine ecosystem attributes and MPA images represent approximately 20% of all the images in both sites. In Vanuatu sites, there is little international investment so the benefits are primarily received locally and by the national private sector. In St. Martin, part of the tourism sector is owned by

international investors (Tourism and Transport Consult, 2005) but, to our knowledge, no leakage calculations have been assessed.

The determination of the contribution of the MPA to the production of ES3 must take into account that the ES is more dependent on physical than biological factors (Pascal et al., 2016). As MPAs strengthened mainly the biological factors, with little evidence of influence on the physical factors driving the production of the ES (Halpern, 2003) and without any supporting literature, a low factor (5%) of the value of the ES, was proposed. Private and public infrastructure at both sites experience protection from coastal processes due to ES3.

Regarding bequest value (ES4) that people may ascribe to the coral reef, none of the available studies assessed were sufficiently precise to determine the role played by the MPA in the valuation. Therefore, based on existing evidence of MPA impacts on habitats and biodiversity (Sale et al., 2005), we applied a contributing factor of 35% on the ES valuation.

The effect of the MPA on the ES5 is calculated as the proportion of the grants or assistance received due to its existence. The role was assessed through interviews with grant recipients (village committees, NGOs or MPA managers).

The impacts of the MPA on ES6 were based on the existing literature and expert opinions, in order to assess the level of conversion or degradation threats on the mangrove habitats in Saint Martin. Based on an annual loss surface of habitat avoided by the existence of the MPA regulations (3–25 ha.y⁻¹ of mangrove habitat non destroyed), the factor of the MPA was estimated at 10%.

The benefits for ES4, ES5, and ES6 can be far-reaching (Balmford et al., 2008). They can be experienced by the local, national and international communities in terms of the existence value, conservation of biodiversity, and the removal of carbon from the atmosphere.

2.5. Costs of MPAs

Different kinds of MPA costs were distinguished: direct operational costs, initial investment and opportunity costs. The direct operational costs include administration, employment, monitoring and enforcement. The investment costs refer to the establishment costs, transaction costs, and material assets such as boats, buoys, signboards, etc. The opportunity costs consider all the losses of potential earnings such as losses in the fishery revenues, longer displacement time for fishermen or loss from the time spent in the MPA management. The jobs created by the MPAs (in management, monitoring and enforcement) were considered here as a cost, considering that in the absence of the MPA, workers would have obtained another similar level of job elsewhere. Other opportunity costs linked to the MPA establishment for tourism (e.g. new advertising material, relocation of tours, etc.) are very limited in both islands and have not been incorporated in the study. For investment costs, normal rules of accountancy were applied to calculate annual amortizing costs (i.e. based on life expectancy of the assets).

2.6. Economic and financial analysis

The *ex-post* Return on Investment (RoI) was calculated following standard methods (Campbell and Brown, 2003). It is based on the observed historic cash flows of impacts on ES1 and ES2 and compared to the direct and investment costs (without amortizing). RoI was calculated on discounted values for the period from the creation of each MPA, to the study date (10 years for Saint Martin and 6–8 years for Vanuatu).

In addition to the *ex-post* study, an *ex-ante* CBA was undertaken based on the present values of projections of the impacts (Wielgus et al., 2008). Projections are made under simplifying

assumptions incorporated into a scenario that takes into account the potential for development of tourism and fisheries in each country. Following existing studies (Wielgus et al., 2008), the projections cover a period of 25 years after the creation of the MPA. This range was proposed in similar studies to reflect the ecological responses of ecosystems to the tested scenario (Balmford et al., 2008). The period starts from the first payment of investment costs and/or the beginning of enforcement activities (2001 for Saint Martin and 2002/2004 for Vanuatu). A Benefit Cost Ratio (BCR) is calculated on the flows of average values of impacts on all ES and costs. We apply a discount rate of 10% in computing present values (Ehrlich, 2008) even though lower discount rates have been argued to be more appropriate when assessing investment in nature (TEEB, 2010b) in order to have the most disadvantageous situation possible (Weitzman, 2001). Using this rate strengthens any positive conclusions (BCR > 1) that are reached.

2.7. Sensitivity analysis

Sensitivity analyses were carried out to better understand the robustness of our estimates (Cesar and Chong, 2006). Analyses focused on the different estimates of the MPA factors for each of the 6 ES (Table 3). The chosen method is through an “extreme case” approach that allows, through non-intensive techniques, the identification of those inputs that generate the most sensitivity (Boardman et al., 1996). The levels presented are based on minimum and maximum estimates of the factor, whereas the best estimate represents the average value. Sensitivity analyses were conducted for the BCR and RoI results.

An additional sensitivity analysis was carried out on the best estimate of the BCR for different discount rates (5%, 7%, 10%) (Pearce et al., 2006).

3. RESULTS

3.1. Impacts on ES

The mean economic impacts have been estimated at US\$ 44,200 (+/–19,800) per year per km² for the five MPAs in Vanuatu and US\$ 29,800 per year per km² for the MPA in Saint Martin (Table 3). For both case studies, the effects of the MPA on tourism, represent the greatest proportion of the total effect (60–70%). Tourism effects were followed by effects on the fishery added value (25%) in Vanuatu and coastal protection (23%) in Saint Martin.

On a *per surface* area basis, the MPA impacts on the tourism sector were similar across sites (Table 3). ES2 associated with the MPA generates a value of US\$ 21,446 in 2010 in Saint Martin, shared among the private nautical sector (15%), scuba diving (15%), day charters (10%), boat rental (10%) and glass-bottom boats (10%). In 2009 in Vanuatu, the average ES2 was assessed at US\$ 33,000 with a MPA impact (US\$ 25,375) shared between day tours (60%) and guest-houses (40%). The differences in the contributions of MPA impacts in the ES2 values (8% and 70% for Saint Martin and Vanuatu respectively) highlight the different motivations and expenditure behaviours between the specific nature tourism niche in Vanuatu and a mass tourism destination in Saint Martin. Complementary results about other MPA impacts per ES are given in Table 3.

3.2. Financial and economic results

The total direct costs without amortizing (Table 4) are US\$ 1,800 per year and US\$ 300,000 per year for Vanuatu and Saint Martin, respectively. However, on a *per area* basis, these direct costs are quite similar (US\$ 12,100 vs 10,700 per year per km²).

Table 3
Synthesis of the main results in terms of ES valuation and MPA impacts.

	2009 and 2010 data, US\$.y ⁻¹ .km ⁻² of MPA			
	ES valuation		MPA impacts	
	Vanuatu (mean)	St Martin	Vanuatu (mean)	St Martin
Subsistence and commercial fishery (ES1)	42.846	837	10.863	235
Tourism (ES2)	32.614	329.290	25.375	21.446
Coastal protection (ES3)	34.319	83.586	3.339	6.822
Bequest value (ES4)	8.745	N.A.	2.993	N.A.
Social capital (ES5)	1.675	589	1.648	589
CO2 sequestration (ES6)	N.A.	6.842	N.A.	683
TOTAL MPA impacts	120.199	421.144	44.218	29.775

Table 4
Synthesis of the main results in terms of MPA costs.

	MPA costs (US\$.y ⁻¹ .km ⁻² of MPA)	
	Vanuatu (mean)	St Martin
Direct costs without amortizing	-12.110	-10.740
Initial investment and material assets	-56.231	-18.912
Amortizing costs	-8.033	-2.128
Opportunity costs	0	-2.514
TOTAL MPA annual costs with amortizing costs	-20.143	-15.382

The BCR is calculated as the expected net benefits of the MPA over a 25-year project life and evaluated at a 10% discount rate (Table 5). Using these parameters, we calculate a BCR between 2 and 4.1. The longer the MPA life, the higher the ratio, due to the initial establishment costs being spread over more years. It also reflects the potential of future development of the tourism and fisheries sectors in Vanuatu.

The positive BCR for all the studied MPAs demonstrates that investment in marine reserves, in addition to conservation of coral reefs, is an effective means of investing in economic development.

The results of the RoI are based on the present values of observed impacts from the date of the MPA creation up to 2009–2010 (Table 6). The mean RoI of Vanuatu MPAs is 0.8 (+/-0.6) after 6–8 years of activity. For Saint Martin, the RoI is 0.6 after 10 years of activity. The positive results of RoI mean that the benefits exceed the costs after several years of activity and that public investment should be considered. It confirms the previous statement, that investment in MPAs can produce benefits which are greater than the costs, even if based only on the historic fishery and tourism business benefits (ES1 and ES2).

Table 5
Synthesis of the main results for BCR and present values of projected MPA impacts under scenario 2 (25 years).

Present values ($t = 10\%$) - 25y projections	MPA impacts (US\$.km ⁻² of MPA)	
	Vanuatu (mean)	St Martin
Subsistence and commercial fishery (ES1)	73.378	1.511
Tourism (ES2)	188.587	144.188
Coastal protection (ES3)	15.136	42.637
Bequest value (ES4)	21.968	0
Social capital (ES5)	15.666	3.679
CO2 sequestration (ES6)	0	4.271
TOTAL MPA impacts (1)	314.734	196.286
TOTAL MPA economic costs (2)	-76.165	-92.363
BCR (1)/(2)	4,1	2,1

Table 6
Synthesis of the main results for RoI ratios and present values of observed MPA impacts since the MPA creation up to 2009–2010.

Present values ($t = 10\%$) - since creation of MPA	MPA impacts (US\$.km ⁻² of MPA)	
	Vanuatu (mean)	St Martin
Subsistence and commercial fishery (ES1)	15.115	729
Tourism (ES2)	34.013	66.471
TOTAL MPA financial impacts (1)	49.128	67.200
TOTAL MPA financial costs (2)	-27.932	-41.945
RoI ((1)-(2))/(2)	0,8	0,6

3.3. Sensitivity analysis

The results of the analysis show a relative sensitivity of the BCR and RoI results to different levels of MPA factors (Fig. 2) with a 2-fold difference between minimum and maximum levels. In particular, in cases where all factors are set to their minimum value, the Vanuatu RoI becomes negative and the Saint Martin RoI ratio is close to zero after 6 years of activity, meaning that costs have exceeded or equalized the benefits. This result reflects (i) that more time is necessary to compensate for the costs of MPA and/or (ii) the investment was too high with respect to the economic benefits.

The results of the analysis on discount rates show a low sensitivity (i.e. less than 20% difference between minimum and maximum) to the different levels on the 25-year projections (Fig. 3).

4. Discussion

4.1. Methodological approaches and reliability

This study provides some important information for decision-making on marine conservation and the need for extensive empirical studies on MPA impacts. Credible evaluations of common conservation instruments are rare (Miteva et al., 2012) and calls for rigorous evaluation of protected areas (PAs) have been made repeatedly (Vincent, 2010). It has been observed that assessments of tropical marine parks are often inaccurate as they value the resource protected and not the protection provided (Pendleton, 1995). In spite of such warnings, economic valuations of the added value, provided by management are still scarce, especially compared to the more static and descriptive studies that estimate Total Economic Values (Wielgus et al., 2010).

Our analysis has presented valuations carried out by utilising a mix of data sources, including experimental fishery with control-impact approach, empirical observations from surveys, transfer benefit and expert opinions. Even though the study demonstrates that investment in marine reserves, in addition to conserving coral reefs, is an effective means of encouraging local economic develop-

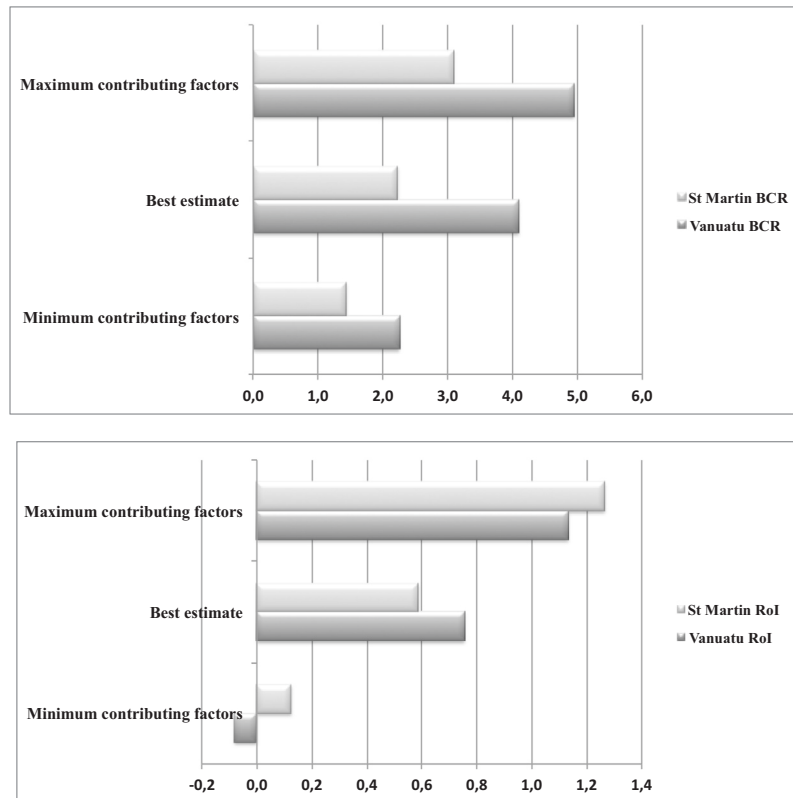


Fig. 2. Sensitivity analysis of BCR and RoI to main uncertainties on MPA factor.

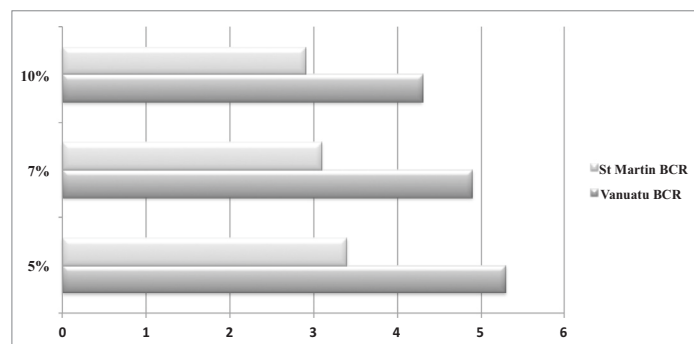


Fig. 3. Sensitivity analysis of BCR based on three discount rates ($t = 10\%$, 7% and 5%).

ment, the benefit-cost ratios have proven to be sensitive to the primary uncertainties regarding the extent to which MPAs impact ES provision. This raises the requirement for (i) filling scientific gaps regarding impacts of MPAs on ES, with multidisciplinary inputs, in order to provide robust information for decision-making and, (ii) future studies to include more systematically designed sensitivity analyses. As noted by François and Pascal (2012), in their review of 17 CBAs on MPAs on coral reef ecosystems, most CBAs neglect to provide any sensitivity analyses on the results. The analysts should employ the use of non-intensive sensitivity analysis techniques, in order to identify those inputs that generate the greatest sensitivity. Several such methods are available (EPA 2009), including extreme case (min-max) analyses, Monte Carlo and variance-based methods.

Additionally, for most of the ES, the relationship with coral reef ecosystems is non-linear (Jennings and Polunin, 1995, Halpern, 2003, Barbier et al., 2008, Ban et al., 2011), which means that a

marginal increase in the area or quality of the ecosystem will not necessarily translate into a proportional increase in services. In particular, the ecosystem is likely to respond very differently at extremely small sizes (Halpern, 2003) or when it is close to disappearance (threshold effects) (Ban et al., 2011). These relationships are unlikely to be linear, although due to the difficulty in estimating these values, this has been the overwhelming assumption in the literature reviewed. Dynamic links between investment and ecosystem services are far from being well-known and almost all of the reviewed studies have focused on average values. The marginal values of ES that depend directly on the ecosystem's productivity can only be estimated if an ecological function which provides the change in service, exists in the literature. Different authors (Cesar, 1996, Cesar and Beukering, 2002, Gunawardena and Rowan, 2005) have estimated the marginal change in the value of fisheries by looking at the increase in production from a marginal change in the ecosystem and the consequent change in pro-

ducer surplus. ES that are valued by estimating a demand curve (stated and revealed preferences, hedonic methods) can also be marginally measured, but these methods are more demanding, both methodologically and in terms of data.

Our study has provided some clear inputs for this production function, by demonstrating empirical MPA impacts on fishery ES. Results are derived from a relatively time consuming regime with a program of experimental fishery, control-impact approach and RCA statistical analysis. Results showed an increase in productivity for the principal gear, estimated to vary from 6% to 33% increase in the CPUE in Vanuatu, similar to findings of the few other available studies (Russ and Alcalá, 1998). Even if variation is substantial (five fold between minimum and maximum CPUE increase) and more empirical studies are necessary, these results should provide a base for other MPA studies through benefit transfer, or through calibration of the many existing bio-economic fishery models. Other observed effects were not valued, such as: (i) less variability on fish harvest per trip (Cote et al., 2001) and, (ii) larger-sized fish caught for villages with MPAs (Halpern and Warner, 2002).

For tourism, results were derived primarily from surveys with end-users to determine the role of the MPA and marine attributes enhanced by MPA in their choice destination. The range of factors was high (between 10% and 70%) and reflects the variability of the tourism context faced by MPAs. The ecological attributes and/or the presence of the MPA will have different weights in the motivation of tourists to visit the sites. In Vanuatu, the tourism context is a small-scale accommodation sector attracting primarily a “nature- tourism” segment, for which MPAs represent an important attraction. In Saint Martin, with a mass tourism sector based mainly on the model of the 3Ss (Sea, Sand and Sun), the MPA and/or ecological attributes of the site (e.g. scenic beauty and the presence of emblematic species) do not appear to be an important aspect of tourism attractiveness. In both cases, the MPA impact on the tourism ES was the most important in monetary terms. This importance highlights the weight of this relationship between MPAs and tourism linked to nature.

Even though it is important to bear in mind the difficulty of isolating any factor in the selection process of the destination by tourists (Tourism and Transport Consult, 2005, Parry and McElroy, 2009), the approach of determining the MPA factor through end-user surveys has provided some clear insight about this MPA impact.

4.2. Spatial boundary of analysis

The spatial boundary has rarely been discussed systematically and specifically in the reviewed CBAs (François and Pascal, 2012). Most studies define and delimit the ecosystem being studied, and some explicitly limit their analysis to the local communities and areas directly interacting with the ecosystem (“local” economy), but these limits generally do not coincide with the area impacted by the ecosystem processes, nor the economic jurisdiction (Balmford et al., 2008). Several authors concur that spatial understanding of ES is a key variable in the implementation of the majority of economic instruments (Balmford et al., 2008). Indeed, identification of areas where the processes required to generate ES are central to accurately improve their management. For example, the larval dispersal and migration areas of key species for fishing, or as dive attractions are often not in the same location as the fishing grounds and dive sites (Kinlan et al., 2005). Similarly, the site where the added value of fishing and tourism is generated, may be different from the area where the benefit is found (residence tax, pre-paid package tourism, etc.) (Burke et al., 2008). However, appropriate policy formation based upon spatially explicit economic values is rife with theoretical and empirical limitations. Considering the complexity of these processes (variability

and importance) and the technical challenge in identifying the flows of dispersion of marine species (Sale et al., 2005), economists will need to use proxies and make simplified assumptions in order to most efficiently affect individual human behavior toward social objectives.

4.3. Advocacy

The results of this study have provided information which can be used by decision-makers regarding investment in marine reserves as a conservation and development tool for local populations. Our findings are complementary with the few peer-reviewed papers on the subject of CBAs in the domain of coral reef economics. One study using a similar approach, was conducted by White et al. (2000) in Olango island, Cebu (Philippines) and showed a very strong justification on the part of local and national government and private sector groups to invest in the management of reefs to generate current annual net revenue ranging from US\$ 38,300–63,400 per km². Another similar approach was provided by Cesar and Chong (2006), who assessed the economic interest in establishing a MPA in Portland Bight Area (Jamaica). The authors found a total (incremental) benefit estimated around US\$ 40.8 million (in the pessimistic tourism case), hence justifying the US\$ 19.2 million costs involved in the MPA implementation over 25 years. Calculating the financial benefits based on the size of the Jamaican MPA, shows a rough figure of US\$ 21,700 per year per km². Though slightly greater, our key-figures ranging from US\$ 30,000 (Saint Martin) and US\$ 44,000 (Vanuatu) per km², are still consistent with all these findings.

Most other CBAs for coral reef management are found in the grey literature. For example, studies in South Africa and Vietnam found Net Present Values of MPAs, estimated at US\$ 10.4 million (Turpie et al., 2006) and US\$ 18.9 million (Nam and Beukering, 2013), respectively. However, these results are difficult to link with our single MPA figures. Other CBA reports significantly differ from our objectives as they focus on analysing other management options. For example, Fahrudin (2003) looked at the economic effect of reducing coral threats (cyanide and coral collection) in Indonesia, while Beukering et al. (2007) analysed the management options of six MPAs in Hawaii, and Clarke et al. (2010) assessed the cost and benefits of a zoning extension of a marine park in Australia.

It may be important to complement the CBA with more specific indicators about effectiveness for nature conservation (Tallis et al., 2010). In our case we relied on areas of protected reef as an indicator of conservation, but other studies have demonstrated that MPAs are not a “silver bullet” for all marine conservation issues (Boersma and Parrish, 1999).

To our knowledge, the juxtaposition of measures of local financial returns (RoI) and of broader economic measures of project net benefits (BCR) to bridge the gap between incentives for local stewardship on one hand and total economic value on the other, has been used infrequently in the environmental economics scientific literature. By comparing it with the BCR, the indicator RoI provides less complete information, not including benefits absent from markets (ES3 to 5) as well as opportunity costs. The RoI was intended to highlight the cash flows for the local economy, in addition to fisheries and tourism, which represent both concrete sources of local cash incomes and implicit income through harvested foodstuff. Focusing on flows of observable impacts and avoiding the projections bias, it may complement the BCR with results that are easier to understand or accept. Financial decision-makers (e.g. public finance), accustomed to the CBA and RoI tools (Arrow et al., 1996) may feel more comfortable basing their budget allocation tradeoffs on both results. As highlighted by other authors (de Wit et al., 2012) it is important to prove that public investments in

natural assets provide yields with adequate returns, as does investment in other infrastructure and services, such as housing and education.

In parallel, the BCR and RoI financial approach may bring concrete information about potential returns for private sources of financing for conservation (Parker et al., 2012). In the case of the impact investment funds (Achleitner et al., 2011) or the Socially Responsible Investment (SRI) funds (Van den Bossche et al., 2010), where capital is invested with the explicit expectation of financial, social and environmental returns, the Internal Rate of Return (IRR) and the RoI are common indicators for investment selection (Littlefield, 2011). Many requirements for viable and effective investment of this type are still necessary (conservation market institutions, quantifying benefits, business models, accounting and auditing standards) (Miteva et al., 2012) and this work is an initial step in this new field.

5. Conclusion

The study has provided some insights into the trade-offs between a pressing need for informed decision making on the environment and the requirement for extensive empirical studies on MPA impacts. As we have shown, CBAs have demonstrated both promising results and significant limitations. Results can be sensitive to the uncertainties and knowledge gaps regarding MPA impacts on ES, however CBAs of environmental investments (such as MPAs) are complex exercises and methods must still be consolidated. There is a need to improve the ecological knowledge base in relation to CBAs, such as the requirement to address the links between investment, ecosystem processes and ecosystem services. This study has proposed several approaches for the valuation of these links with the ES of fisheries and tourism. It also showed evidence that the selected MPAs produced benefits for local stakeholders through their impacts on ES; commercial and subsistence fishermen (through impacts on ES1), tourism businesses (ES2), landowners (ES3), local populations (ES4 and ES5) and the international community (ES6) all benefited from the establishment and management of these MPAs. The evidence of local benefits and their distribution patterns has been recognized as a way of increasing support from local populations (Whittingham et al., 2003). For multilateral agencies and public finance decision-makers, this study also demonstrated that investing in MPAs can be effective both for the conservation of nature and to fulfil local economic development objectives. Financial and economic analyses of ES may enable benchmarks with other investments in infrastructure or public services and change the false idea that investing in nature produces low returns (de Wit et al., 2012). If adequately communicated, these results can allow for more informed decisions regarding budget allocations. Such nature based solutions for economic development challenges, provide excellent examples of the conditions under which investments in nature's wealth, also result in the increase in the wealth of nations.

In addition to strengthening support from policy makers, these results also show a potential basis for the establishment of exploratory marine PES, by helping in the identification of the beneficiaries of improved marine ES (Engel et al., 2008). Clear values of ES and impacts of investment can serve to "fine tune" the instrument in negotiations between providers and buyers (The Katoomba Group and Marketplace, 2010). In parallel, the private financing of conservation with innovative tools such as impact investing funds (Littlefield, 2011), where capital is invested with the explicit expectation of economic, social and environmental returns, may be attracted by some aspects of the benefit-cost ratios for MPAs, since it could make the business case for coral reef conservation.

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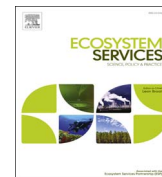
Chapter 3: Economic valuation of coral reef ecosystem service of coastal protection: A pragmatic approach



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Economic valuation of coral reef ecosystem service of coastal protection: A pragmatic approach

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ABSTRACT

The identification and economic valuation of ecosystem services (ES) are becoming important components of coral reef management. In many contexts, protection of human assets against coastal floods is one of the most important ES provided by coral reefs. The methods utilized to characterize this ES should be able to accommodate situations with low data availability, without sacrificing robustness. In this paper, we suggest such an approach that utilizes expert opinion and does not require copious amounts of data. Our primary objective is to find a balance between simple and complex models that can be used in a data scarce environment, to produce an economic valuation of the coral reef ES of protection against coastal floods. The approach has three steps: (i) identify geographic zones and assets at risk, (ii) identify the contributing role of coral reefs in the protection of coasts and, (iii) value the annual repair costs of assets through the avoided damage cost approach. The proposed method seems appropriate for advocacy with policy makers, but appears to be less effective for small scale approaches, such as those required for Payment for ES negotiations or marine spatial planning.

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1. Introduction

Economic valuations of coral reef ecosystem services (ES) have been undertaken to address several objectives, such as raising environmental awareness among decision-makers and the public, evaluating the costs and benefits gained from different levels of investment in coastal management, incorporating present and future values of both negative and positive impacts via a common metric and fine tuning economic instruments (Burke et al., 2008; Laurans et al., 2013). In addition, valuations can form part of monitoring routines which utilize quantitative indicators to inform management ("we manage better what we can measure") (Beukering et al., 2007; David et al., 2007; Pascal et al., 2012) and used in the implementation of tools such as 'Payment for ES' (PES) (Huwyler et al., 2014). To date, the most common use of economic valuations of marine ES has been for advocacy with policy makers (Waite et al., 2014). In these studies, the priority has been to transmit accurate information on the value of a large range of ES

and their beneficiaries, without striving for the highest level of confidence ("science to inform policy"). For other studies, such as those focusing on monitoring or the design of a PES, the level of precision required is much higher (Sale et al., 2014).

Recent work on the economic valuation of coral reef ES shows that it is important to concentrate valuation efforts on three main ES, which under the Millennium Ecosystem Assessment's classification (MEA, 2003) fall under the headings of provisioning, cultural and regulating (Hilmi et al., 2014; Laurans et al., 2013). The first two ES (provisioning and cultural) refer to respectively: (i) fish biomass production; coral reef associated fisheries provide an important source of protein and a basis for livelihoods and (ii) scenic beauty for recreational tourism; coastal recreation and tourism activities generate significant economic value depending on the quality and availability of specific marine ecosystem attributes. This article will focus on coastal protection provided by coral reefs, which falls under the third heading of regulating services.

Coral reefs form barriers which buffer coastal zones from severe weather events and in so doing, protect human lives, coastal properties and economic activities (Barbier et al., 2011). Several

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studies show that reefs reduce ocean swells, which results in wave transformations and rapid attenuation of wave energy (Brander et al., 2004; Kench and Brander, 2009; Lugo-Fernandez et al., 1998). The fringing reef can absorb a large part of this force (Brander et al., 2004; Roberts et al., 1992) up to 90% at low tide (Lugo-Fernandez et al., 1998). Given the expected increase of extreme climatic event occurrences, as well as sea level rise due to climate change (IPCC, 2013), it becomes increasingly important to better understand, communicate and manage this ecosystem service of coral reefs.

Another service related to coastal protection, is the contribution of reefs to the regulation of erosion and sedimentation, which are critical to maintaining beaches and coastlines. Coastlines where coral reefs are located receive coral or algal derived sediments from this ecosystem via various hydrodynamic processes (waves, currents, tides). Accumulation on the coastline of those sediments (sedimentation) is the source of beach formation (Huang et al., 2007).

The ES of coastal protection is therefore relevant both for the protection of human lives and economic activities associated with beaches. Economic valuation of shoreline protection provided by coral reefs is a useful tool in highlighting the importance of this service to decision-makers (Sale et al., 2014).

Characterization (with or without economic valuation) of the coastal protection ES however is extremely challenging, especially in a data scarce environment. Over 64% of coral reef ecosystems lay in developing countries (Bryant et al., 2011) which are often characterised by high population densities in the nearshore and limited government resources for coastal management. The development of ES valuation methodologies adapted to these contexts is a requirement if the full potential of ES valuations is to be realized.

Many factors contribute to make the valuation of coastal protection ES complex. The typology of the reef, its depth and the type of wave that impact it, result in great variability in the assessment of coastal protection (Kench and Brander, 2009). In addition, the role of coral reefs in coastal protection is mixed with other factors which drive the process, such as: bathymetry, currents, geomorphology, and biological cover (Burke, 2004). Some of these factors (e.g. biotic components such as live coral coverage) require complex methods and add another layer of difficulty (Barbier et al., 2008; Cooper et al., 2009; Ferrario et al., 2014; Gourlay and Colleter, 2005; Van Zanten et al., 2014; Yee et al., 2014) This might explain why, to our knowledge, relatively few economic valuation studies (Laurans et al., 2013; Van Zanten et al., 2014) have focused on isolating the role that reefs play in coastal protection. However, taking into consideration the importance of their role for advocacy for policy makers, it is important to improve these valuations, in order to transmit the most accurate values of this ES.

The aim of this article is to propose a relatively easy methodology that will allow for an assessment of the value of coastal protection provided by coral reefs, within a data scarce environment. The methodology has been designed primarily for policy advocacy.

We will discuss if improved characterization of the ES (with or without valuation) can contribute to better management of the coastal ecosystems, as well as being the initial step of a payment scheme with beneficiaries of the ES as suggested by Engel et al. (2008).

2. Material and methods

To fulfill the objective of this paper, five studies were used to illustrate the valuation of coastal protection provided by coral reefs (Table 1). Chosen study areas have many similarities: (i) insular

Table 1
Main characteristics of case studies – ADC stands for Avoided Damage Costs method and RC stands for Replacement Cost method.

Ocean/Regional Seas	Country/Island or specific area	Objectives of the study	Scale of the study	Valuing method	Method particularities	Sources
North West Pacific Ocean/Philippines Sea	Northern Mariana/Is. Saipan	Valuation of total value of ecosystem services provided by coral reefs	Island	ADC	Comparison of damages inflicted by marine flooding based on a reference situation (current geomorphology of reefs) and an hypothetical one (absence of reefs).	Beukering et al. (2006)
North West Pacific Ocean/Philippines Sea	Philippines/Marine Bohol Triangle	Economic valuation of ecosystem services provided by four marine ecosystems (coral reefs, mangroves, beaches or intertidal areas, and marine waters)	Specific area	RC	Only mangroves are considered as acting for coastal protection (the role of reefs is not accounted)	Samonte-Tan et al. (2007)
North West Atlantic Ocean/South East Caribbean Sea	Trinidad&Tobago, Belize, St. Lucia	Economic valuation of coral reefs in the area	Island	ADC	Use of Coastal Protection Index	Burke et al. (2008), Cooper et al. (2009)
South West Pacific Ocean	New-Caledonia	Economic valuation of ecosystem services of coral reefs at the scale of the area	Island	ADC	Method applied in a low-data context	Pascal (2010)
North West Atlantic Ocean/North East Caribbean Sea	Virgin Islands/Saint-Croix	Comparison of existing methods for quantifying shoreline protection at island scale	Island	N/A	Measurement of relative contribution of different habitat types to wave energy attenuation. Use of biophysical features of the reef for characterization of the coastal protection	Van Zanten et al. (2014); Yee et al. (2014)

territories (entire single islands such as Tobago, New-Caledonia), Saint-Croix (Virgin Islands), Saipan (Northern Mariana Islands) or a set of several islands such as Marine Bohol Triangle (Philippines) and, (ii) a GDP superior to the world average¹ (excluding the Bohol Marine Triangle, located in Philippines) (Central Intelligence Agency, 2014). They were chosen to reflect the diversity of existing methodological approaches, and provide an optimal background for supporting our choices presented in the Results section.

In the majority of case studies analyzed, methodologies used to value coastal protection were based on three primary steps: (i) determination of areas and assets exposed to risk, (ii) quantification of reef contribution to coastal protection and (iii) economic valuation. These three steps can be implemented independently of each other, producing important heterogeneity in methodological approaches.

In the next section, we will present the three methodological steps, including a case study that is highly data (and time) demanding and another which is less data dependent.

2.1. First step: determination of areas and assets exposed to risk

The area at risk is defined as the shoreline located under the maximum wave height. To identify such on the island of Saipan (Commonwealth of the Northern Marina Islands) (Beukering et al., 2006) the authors used a comparison of two models predicting maximum average wave height during an extreme climatic event along the entire shoreline. One model utilized the existing situation, with reef, while in the other hypothetical case, the reef was excluded. After determining the zone at risk for flooding, both in the presence and absence of reef, the authors then determined the infrastructure and buildings that might be impacted. A similar approach was used in Cooper et al. (2009).

In other studies, only the scenario with existing reef coverage was used (Burke et al., 2008; Pascal, 2010; Van Zanten et al., 2014) and simpler historical data on 'average wave height' during extreme climate events were utilized instead of predictive models to determine the zones at risk. Van Zanten et al. (2014) incorporated local oceanographic data for assessing the maximum average height of waves, while Burke (2008) used local data based on observations from a storm atlas (online source from Organization of American States) and Pascal (2010) an average regional maximum height calculated for tropical environments by meteorological departments (online sources from Meteo France and NOAA).

2.2. Second step: quantifying the role of coral reefs in coastal protection

Reefs can dissipate wave energy either by waves breaking on the physical structure or via reef friction (biophysical) (Van Zanten et al., 2014; Yee et al., 2014). The physical function has been well documented (Hardy et al., 1990; Hearn, 1999; Massel and Brinkman, 1999; Roberts et al., 1992; Young, 1989), but less emphasis has been placed on the friction or biophysical aspects (Beaumont et al., 2007; Done et al., 1996; Duarte, 2000). Both processes have been utilized in methods to assess the value of the coral reef to coastal protection.

Yee et al. (2014) and Van Zanten et al. (2014) used the biophysical approach to demonstrate the role of coral reefs in decreasing wave heights, and dissipating wave energy. These studies showed the importance of friction which is linked to percentages of living coral coverage. A wave model taking into account coral

cover, wave characteristics, water depth and storm characteristics, was applied by Van Zanten et al. (2014). The wave model was used to determine the amount of energy dissipated via four reef types (shallow high density, shallow low density, deep high density and deep low density reefs). It was then possible to assess the contribution of reefs to the coastal protection service.

Yee et al. (2014) confirmed that models including friction are better at explaining wave energy dissipation over the reef. Shepard et al. (2002), introduced a method to measure attenuation percentage in wave energy and wave height due to the presence of the reef. The spatial scale was a critical aspect. In this case, the studied area was divided into 10 × 10 meter-cell grids to reflect the complexity of the shoreline (coral cover, reef depth etc). The scale and data required by the biophysical models however, makes them difficult to establish for large areas of tens of kilometers (as might be the case for the valuation of an island or sub-national areas).

Burke et al. (2008), Cooper et al. (2009) and Pascal (2010) developed a coastal protection index (CPI), which incorporates the contribution of the reef to coastal protection (the reef-contributing factor). This index is the sum of scores, attributed to several factors that contribute to coastal protection and represents the 'degree of protection' to the shoreline, which includes the contributory role of the reef. The number of factors required to calculate the CPI were intentionally limited to 6 in order to avoid extensive data collection and they partially reflect the complexity of the coastal protection service. The calibration of the factors is a critical component in the determination of the CPI.

In Van Beukering et al.'s (2006) study, the contributory role of the reef was assessed by comparing the values of avoided damages cost with and without the presence of the coral reef. This method relied heavily on the application of a theoretical model, where the quality of results was directly linked to the availability of the input data.

2.3. Third step: economic valuation of the ES

The literature provides two main methods for the economic valuation of the coastal protection service: avoided damages and replacement costs.

Van Zanten et al. (2014) used economic data from an existing land use valuation methodology estimating "the maximum damage for flood events per land use class per hectare in the Netherlands". The Dutch values were then translated from euro to dollar² and adapted to the USVI context. Then, the percentages of wave energy dissipation and friction established from the previous step were applied to the four types of reef.

Burke et al. (2008) and Pascal (2010) used average property values (derived from real estate market prices and construction costs), prorated by the reef-contributing factor previously calculated. While Cooper et al. (2009) used location-specific average property values, adjusted by settlement and proximity to the coast. The avoided damages costs method used in Beukering et al.'s (2006) study considered the mean value of property at risk, and then estimated a percentage of 'flooding damage' to the property (in that case, 5% of the entire property value was impacted by flooding). Burke et al. (2008) and Cooper et al. (2009) used the share of coastal protection ascribed to the coral reefs as a prorating factor.

For replacement costs, (Samonte-Tan et al., 2007) used replacement costs of the construction and maintenance of an artificial breakwater (seawall or dyke) to assess the value of coastal protection provided by mangroves in the Bohol Marine Triangle (BMT). Although the method is applied in this case for the

¹ This material highlights the difficulty in finding case studies from developing countries.

² The exchange rate used is: €1=\$1.32 (2007, source: www.xe.com).

mangroves, it behaves similarly for coral reefs (Brander et al., 2012). The average value of constructing protective seawalls or dykes in the area was estimated at \$352/Ha. This approach did not take into account the potential differences of construction and maintenance costs between different types of seawalls and a wide range of seawalls exists, varying in material and orientation.

2.4. Summary of economic values

Results of economic valuations have been summarised (see Table 3) for informative and comparative purposes.

3. Results

3.1. General model

The model proposed hereafter is based on those developed by Burke et al. (2008) and Pascal (2010). Both served as a starting point and elements of each have been utilized in developing this new method. The general model is divided in four main steps: (i) Identify coastal areas potentially at risk during coastal flooding events, (ii) determine the CPI and the contribution of coral reefs to coastal protection and, (iii) quantify and value the avoided damages, including repair costs and loss of use through two sub-steps: (a) characterization of the assets exposed to risk (three main categories), (b) valuation of the total repair costs of direct and indirect tangible damages and, the last step (iv) estimation of the annual probability of flood event per impact category.

3.2. Step 1: zones at risk mapping

Identifying assets under flood risk is the first step of the economic valuation. The main challenge is to clearly identify the areas at risk, using low precision maps and satellite data. We proposed that areas at risk be designated as the coastal lands between 0 and 6 m above the high tide sea level (as storm surge will have maximum impact if it coincides with high tide). This level comes from models of extreme wave height, such as Slosh (Sea, Lake and Overland Surges from Hurricanes) as well as from data on historic maximum heights of non-tsunami waves during the last 25 years (main sources: NOAA, Météo France and Vanuatu Meteorological Office), showing that the maximum wave height surging on shorelines rarely exceeds 6 m.

A projection of this height to determine the potential impacts of flood events can be made easily with GIS topographic data. Several databases might be used, such as Aerial photography:

PlanetObserver (satellite images), Google Earth plug-in layer on Qgis software, and/or GIS data from Pacific Catastrophe Risk Assessment and Financing (PCRAFI, 2011). The areas vulnerable to the flood impacts has been limited to 1 km inland following the experiences developed by Cooper et al. (2009), Das and Vincent (2009), Kench and Brander (2009).

3.3. Step 2: Coastal Protection Index (CPI)

The Institute of Marine Affairs (IMA) the World Resources Institute (WRI) and the University Of New Caledonia (UNC) developed a Coastal Protection Index (CPI), based on a set of geomorphological and physical factors. This CPI showed in numerical form, the level of protection provided from intense wave action (Burke et al., 2008) as well as the contribution of coral reef to this protection.

The decision was made to divide the study zone into shoreline segments of 10s–100 km depending on the similarity of one or several of the first 3 factors of the Table 2 (i.e. geomorphology, coastal exposure and reef morphology).

For each segment of the coastline, the CPI incorporates up to seven factors (Table 2) depending on available data. For each factor, a number between 1 (no protection provided) and 5 (maximum protection) is assessed. The average of all factors is then calculated to produce a unique index value, the CPI. The relative contribution of reefs to the CPI can then be calculated.

3.4. Step 3: economic valuation

For the valuation of avoided damages costs, it is necessary to determine the different categories of damages from flood events (Messner et al., 2007). Due to the difficulty of assigning an economic value to intangible damages (loss of life, increased vulnerability of survivors), we chose to focus the analysis on tangible damages. For those that are direct and tangible (i.e. damage on physical assets), three categories of land use are distinguished: (i) buildings (ii) infrastructure and (iii) crops. For those that are indirect (i.e. loss of industrial production, traffic disruption and emergency costs), emergency costs is the most appropriate choice in terms of availability of data. Other indirect tangible damages such as loss of industrial and agricultural production, as well as loss of tourism revenue due to the temporary closure of hotels are very important, but precise data are generally difficult to find.

Following the methodology developed in several reports (Bolt et al., 2005; Messner et al., 2007; Pagiola, 2004; PCRAFI, 2011), we assess the total repair costs of damages based on average construction (and/or crop appraisal values as well as standard

Table 2
Coastal Protection Index.

	Very strong	Strong	Medium	Low	None
	5	4	3	2	1
Geomorphology	Rocky shore	Mix of rocks/sediments/mangroves	Mangroves	Sediments	Beaches
Coastal exposure	Protected bay	Semi-protected bay	Artificial reefs	Low protected bay or coast	No protection
Reef morphology, area and distance to the coast	Continuous barrier (> 80%) close to the coast (< 1 km)	Continuous barrier (> 50%), patch reef, close to the coast (< 1 km)	Fringing reefs (width > 100 m)	Coral formation discontinuous	No reefs
Inner slope, crest width	Very favorable conditions (gentle slope, large crest width)	Favorable conditions (slope, large crest width)	Favorable conditions (at least one component)	Reduced favorable conditions (strong slope, reduced crest width)	None
Platform slope	6–10%	2.5–6%	1.1–2.5%	0.4–1.1%	< 0.4%
Main depth (1 km away from the coast)	< 2 m	< 5 m	> 5 m	> 10 m	> 30 m
Presence of other ecosystems	Mangroves, seagrass > 75% shoreline	Mangroves, seagrass 50–75% shoreline	Mangroves, seagrass 25–50% shoreline	Mangroves, seagrass < 25% shoreline	None

emergency costs), and then apply a standard depth-damage function to determine the value of damages. The standard was established to determine the value of damages, assessing the average percentage of destruction of the total asset during a high flooding event (> 5 m high). This standard percentage represents destruction of 65% of the total asset, which corresponds to the average of several studies (California Natural Resources Agency, 2012).

3.5. Other aspects

3.5.1. Assets at risk

Assets might be residential, commercial, public infrastructure (such as airports, ports, power plants, dams, major roads, and bridges), rural and industrial buildings and crops. A database containing a description of the assets with the number of residential houses, the floor area of residential, commercial and public infrastructure and simplified structural characteristics has to be created for each segment. When detailed data are available, information on single properties (also called object-oriented data) will allow for a property-by-property approach (Penning-Rowsell et al., 2003). In other cases, properties will have to be aggregated in areas that are fairly homogeneous (also called aggregated land use data). Land use data might be available from secondary sources (e.g. urban planning). Data can be completed with a GIS-based population database to identify the population at risk in each segment. Data field verification, if planned, will have to be located in areas that have more variety in building type and usage.

3.5.2. Repairing costs

The direct damages for all assets (except crops) are valued through estimates of benchmark costs needed to repair or replace the damaged assets. The values (generally expressed in \$US per m² of building floor area) are collected from a variety of sources, including construction companies, government reports, interviews with local real estate experts, and historical disaster reports. In tropical areas, three types of structures should be considered to reflect costs of different structures: single-story timber, masonry/concrete, and traditional-style buildings. For crops, the damages are derived from loss in annual gross profit. Data might come from local governments or from official annual rental values of croplands.

3.6. Step 4: probability of the hazard event

The final step consists of calculating the annual probability of the hazard event (storm, cyclones) to determine valuation of the ES on a yearly basis. Tropical cyclone activity and intensity are variable on the intraseasonal, interannual, interdecadal and

multidecadal timescales (CSIRO and Australian Bureau of Meteorology, 2007). The probability of events is calculated from existing models for the study region. Tracks of historical tropical cyclones are analyzed from a catalog of historical storms assembled from the dataset of the International Best Tracks Archive for Climate Stewardship project, the NOAA Historical Hurricane Tracks, the Joint Typhoon Warning Center (JTWC), the Australia Bureau of Meteorology (BoM), the France Météo and other country Meteorological Service.

3.7. Formula

(Fig. 1).

3.8. Approach of uncertainties

The model is based on an important number of factors, with uncertainties. Estimates of the minimum and maximum values for the reef contributive index, the areas at risk (e.g. number of residential houses) and the repair costs are proposed to show the range in which the “real” value most likely lies.

3.9. Summary of economic valuations

The valuation outputs in international dollars reflect the variability of coral reefs and the economic context of the sites. This variability is very clear both in absolute values and in values per km² ranging from 14 to 12,250 US\$/ha. In terms of values per capita, results seem less variable, from 14 to 615 US\$/capita (Table 3).

4. Discussion

The proposed methodology aims to be adaptable for a wide range of contexts, in particular low-data availability settings. As highlighted previously, other methods are more precise, but are also more data demanding and require calibration of complex models. Our method is appropriate for many situations, using local parameters that are easily available in most countries. It is being implemented so far in very different geographical contexts (Colombia, French Polynesia, Fiji, Honduras, Martinique, Mayotte, New Caledonia and Vanuatu) bringing concrete experiences of field data collection and communication to decision-makers. As usual the “one size fits all” model does not exist and it is important to determine the pros and cons of the proposed model and identify the best method of utilization. It is also important to note that this method includes only those factors that can be monetized, while policy makers rely on a range of information types and analyses in

$$C = (\sum_{ic} uw_i * f_i * rc_{ic}) * p \quad uw_i < \max. h$$

With :

C: costs of avoided damages

uw_i : coastal area under wave originated flood risk per zone (i)

Max. h : maximum height of waves (historical average)

f_i : Contributing factor of ecosystems in the coastal protection index (CPI) per zone (i)

rc_{ic} : building, infrastructure, crops repair costs estimated per zone (i) (damage function).

P: probability of hazard event

Fig. 1. Formula to assess the value of coral reefs in terms of coastal protection, method of avoided damages costs.

Table 3

Summary of economic valuation from the selected studies. Original results are converted into international Dollar (2011) based on the Power Parity Purchase (PPP) data (Heston et al., 2011) for all countries except Marianas Islands based on The World Fact book (Central Intelligence Agency, 2014). Results are presented in value per km² of coral reef and value per capita based on total population of the study site.

Study Site, sources	Annual value of coastal protection provided by coral reefs or associated ecosystems			Area (Hectares) CR: Coral reefs M: Mangrove	Inhabitants (year of census)
	in Int. Dollar (PPP factor used)	in Int. Dollar Per Capita	in Int. Dollar Per hectare		
Saipan, Beukering et al. (2006)	8 442,000 (1.05)	173	2 723	3 100 (CR)	48,430 (2012)
Marine Bohol Triangle, Samonte-Tan (2007)	405,800 (23.06)	14	1610	252 (M)	28,280 (2010)
Tobago, Burke et al. (2008)	38,250,000 (3.76)	615	12,750	3 000 (CR)	62,219 (2014)
New Caledonia, Pascal, 2010	41,650,000 (0.71)	159	90	460,700 (CR)	262,000 (2013)
Sainte-Croix, Van Zanten et al. (2014) , Yee et al. (2014)	No economic valuation. Measurement of relative contribution of different habitat types to wave energy attenuation				

order to determine the best course of action. One very important element, not captured here is the social element: the importance of coastal ecosystems to human lives for example, and such information should be incorporated as well.

4.1. Interpretation and communication of economic values

As indicated by the high range of valuation outputs, most of the results are very specific to the site and reflect differences in methodologies, ecosystem geomorphology, climate regimes, storm characteristics, coastal exposure and economic development. Experience has shown us that these characteristics can be very variable at the scale of km and this makes difficult any comparison between sites. In this context, the correct use of transfer benefit approaches as defined by ([Wilson and Hoehn, 2006](#)) is a complex exercise with, to our knowledge, a limited track record with coral reef ecosystems ([Burke, 2004](#); [Pérez-Maqueo et al., 2007](#); [Waite et al., 2014](#)).

Regarding presentation of results, the absolute annual economic values, when compared to per hectare or per capita ratios seem to provide clearer indicators of the importance of the ES. Both ratios can suffer bias with a large coral reef area (e.g. New Caledonia) or large, unprotected coastal populations (e.g. Colombia) that would diminish the importance of the ES.

Additionally, we propose to clearly identify the beneficiaries (e.g. the number of households or area size of tourism infrastructure protected) that will aid in communicating the results to decision-makers. This information is extremely useful at the policy level ([Pascal et al., 2014](#); [Waite et al., 2014](#)).

In terms of communication with policy makers and other stakeholders, the method is designed to produce preliminary information that can be of great value for general advocacy. At the scale of 10 s to 100 km (e.g. national or sub-national scale), the results reflect the importance of coral reefs for the local society and can complete the valuation of other ES (e.g. biomass production for fishery, scenic beauty for tourism). It can also assist with the identification of the beneficiaries and generate additional arguments for reef importance. From a conservation point of view, this is extremely positive, as it will help advocate for better management of coral reefs ecosystems.

However, some caution has to be employed, as the method relies on a number of assumptions, which should always be mentioned when communicating the results.

Our approach is not recommended for small-scale processes (1–10 km) or for communication with small-scale stakeholders (e.g. municipal or provincial level) who have specific information needs ([Hein et al., 2006](#)). Small scale processes such as the development of management plans for many Marine Protected Areas (MPAs) or provincial Marine Spatial Planning (MSP) require more

accurate information on the role of the coral reef, than can be provided with this methodology ([Sale et al., 2014](#)).

This method is also not recommended for the design of instruments such as Payment for Ecosystem Services (PES). To our knowledge, the role of economic valuation of marine ES is not extensively described for this instrument. Recent field experiences have revealed that most marine PES are designed for small scales (1–10 km) either for a bundle of services or for a unique service ([Clua and Pascal, 2014](#); [REECS, 2014](#); [The Katoomba Group and Marketplace, 2010](#)). Take for example the case of a beneficiary who wishes to pay for the ES of shoreline protection. As this requires a precise characterization of the ES, at a small scale and a clear identification of the role of live coral ([Agardy et al., 2014](#)), our method would be inappropriate.

4.2. Biological and physical role of coral reefs

Our approach is based on the wave breaking function of coral reefs in coastal protection. It is clear that friction with coral also plays a role in dissipation of wave energy ([Burke, 2004](#); [Gourlay and Colleter, 2005](#); [Sheppard et al., 2002](#); [Van Zanten et al., 2014](#)). According to [Thornton and Guza \(1982\)](#), the relative amount of wave energy dissipation of coral reefs in St. Croix is around 75–85%. In the absence of corals on the reef crest (i.e. friction) the wave energy dissipation function would be 57–66%. The relatively small contributive value of live coral to wave energy dissipation, while important, would be lost in our broad scale methodology proposed. In addition, in spite of the friction factors developed by [Sheppard et al \(2005\)](#) which denote the relative wave energy dissipation for reef types and health and range between 0.1 (coral pavement) to 0.2 (75–100% live coral) this is still considered relatively data demanding and therefore not appropriate for our model. Finally, the scale required by the biophysical models makes them difficult to establish for large areas, such as the tens of kilometers that would be required for the valuation of an island or sub-national areas.

There is also natural coral mortality that should be taken into consideration. The coastal protection service provided by the reef structure (wave-breaking function) is expected to slowly decrease at the rate of natural erosion (10's to 100's of years). This is an aspect that should be reflected in ES valuation studies, through the use of long term projections and discounted rates ([Van Zanten et al., 2014](#)). Then, the importance of live coral on the reef is highlighted and valued.

4.3. Specific role of the coral reef in coastal protection

One other important component of the approach is the assessment of the contributive role of coral reefs to coastal

protection. It is important that this value avoids an over-estimation of the role and value of the coral reef ecosystem service, especially when using the avoided damages costs method. As presented with the CPI calculations, many inter-related factors contribute to the coastal protection service. Linking avoided damages only to the presence of coral reefs does not reflect the role of these other factors. Our proposed method relies on expert opinion to provide information on the relative importance of each of the different factors. As highlighted by different studies (Curtis, 2004; Wilson and Hoehn, 2006) methods of reducing errors associated with the expert approach are supposed to: (1) define a clear and simple quantitative range of values (2) set up a limited number of factors and (3) segment the valuation as much as possible. In this way, the proposed method is based on a quantitative approach, utilizing a maximum of seven different factors.

4.4. Choice between avoided damages and replacement costs approaches

This choice depends on several context variables such as the presence or absence of reef, which might affect the technical feasibility of setting up coastal engineering works, the fetch (distance over water that the wind blows in the same direction) and the economic activities in the zone. Usually, governments, in their strategy against erosion, prefer to concentrate on high priority areas (e.g. with tourism or residential infrastructure) rather than to establish defenses across large areas, which would lead to unrealistic implementation costs. A comparison of the avoided damages costs approach and the replacement cost method shows that the replacement cost method tends to produce extremely high estimates, up to 10 times greater than the lower-bound estimate, calculated using the damage approach (Barbier et al., 2011; Sathirathai and Barbier., 2001; Tri et al., 1996). This suggests that the replacement cost method should be used with caution, in the context of small scales and specific zones. The avoided damage cost approach appears to be better adapted to scales of km and 10 s of km.

4.5. Repair costs calculations

Although Torterotot (1993) gave us a strong basis for the use of damages functions in the economic valuation method, we chose to use a simplified form of damage functions in our methodology in order to adapt it to available data. We assigned to each kind of building, a standard damage percentage for a given inundation depth. The benefit of such an approach is that it eliminates the requirement of collecting parameters required for the calibration of the damage function (e.g. velocity of inundation, duration of immersion, building materials and age) by using a level of damage, dependent on inundation depth and basic type of buildings (e.g. one-level/two-level houses, hospital, school, hotel, restaurant, shops etc). Only average local construction costs per square meter and building type (data which is easy to obtain) have to be collected. The chosen approach via repair costs, allows one also to avoid fluctuations in the real estate market price. The damage function by structure type (wood, concrete etc) is an important element to include in the future. This information is helpful especially in a tropical context, where there is a higher proportion of houses made with wood or local materials, that are more likely to experience severe damage, compared to concrete houses.

4.6. Damages to be valued

Our valuation has been applied to two types of damages only: direct tangible damages (impacts on buildings, infrastructures and crops), and one kind of indirect tangible damages (emergency

costs). The development of damage functions for other specific damages is also important and would complete the valuation. For impacts on humans, other models (PCRAFI, 2011) done in the South Pacific have estimated the casualty number as a function of the total economic losses, which are used as a proxy for the number of damaged buildings.

Another aspect for investigation is the integration of indirect tangible damages, such as loss of industrial production or tourism loss of earnings. For example, regarding impacts on tourism facilities, loss of benefits due to temporal closures of the hotel complex and lower occupancy rates, in addition to the construction replacement costs should provide a better overview of economic benefits of coastal protection provided by coral reefs. Delays in the reopening of hotels due to repairs for example, and the loss of earnings that ensue, are strong arguments for decision-makers and investors, in light of the important contribution of tourism to the GDP of many coral reef countries.

4.7. Climate change

Finally, in an environment where climate change issues are becoming increasingly prominent, leading to their inclusion in environmental policies, it is critical to incorporate this element. In particular, the expected changes in the frequency of extreme climatic events should be considered in the valuation of the coastal protection service (Bender et al., 2010; Van Zanten et al., 2014). There is substantial evidence from theory and model experiments that the large-scale environment in which tropical cyclones form and evolve is changing as a result of climate change (PWA and SAIC, 2009) and that increases in hurricane frequencies are expected (IPCC, 2013). With our model, the probability of extreme events is easily incorporated in the valuation through adapting the annual frequency of occurrence of the events (referred as P, “probability of the hazard event”, in the Fig. 1, formula to assess the value of coral reefs in terms of coastal protection).

5. Conclusion

The result is the design of a 3-step logical method that responds to different socio-ecological contexts, and based on local, publicly available data sources. In addition, we suggested several elements that should be further investigated in order to improve this method, such as extending the categories of repair costs, reflecting the friction role of reefs and including climate change impacts. Most of the methodological choices made have been driven by limited scientific knowledge and the desire for a pragmatic approach, in a data poor environment.

The proposed method demonstrates some promising elements, albeit with clear limitations. Results reflect the relative importance of coral reefs in coastal protection at national or sub-national scales (i.e. at a scale of 10 s to 100 km) and can therefore complete the valuation of other ES (e.g. biomass production for fishery, scenic beauty for tourism). They can also assist in identifying beneficiaries and so contribute required information to policy and decision makers. This sort of information could prompt decision makers to allocate resources for the conservation of coral reefs. The method however appears to be less effective for small scale approaches (km), such as those required for the design of a PES scheme or a process of marine spatial planning.

It is hoped that this paper will assist researchers interested in improving the valuation of the ES of coastal protection provided by coral reefs and other ecosystems (e.g. mangroves, seagrass beds). It further highlights the importance of improved collaboration with other disciplines such as ecologists, geologists, oceanographers, social scientists and those involved in management and policy.

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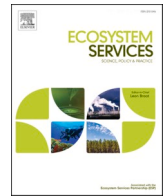
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Chapter 4: When are Payment for Ecosystems Services suitable for Coral Reef derived coastal protection?: a review of scientific requirements



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. Wunder's (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 intervention'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 through 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

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

some cases, they can dissipate greater than 90% wave energy (Kench and Brander, 2006; Ferrario et al., 2014).

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 structures 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 CaCO₃ m⁻²y⁻¹)

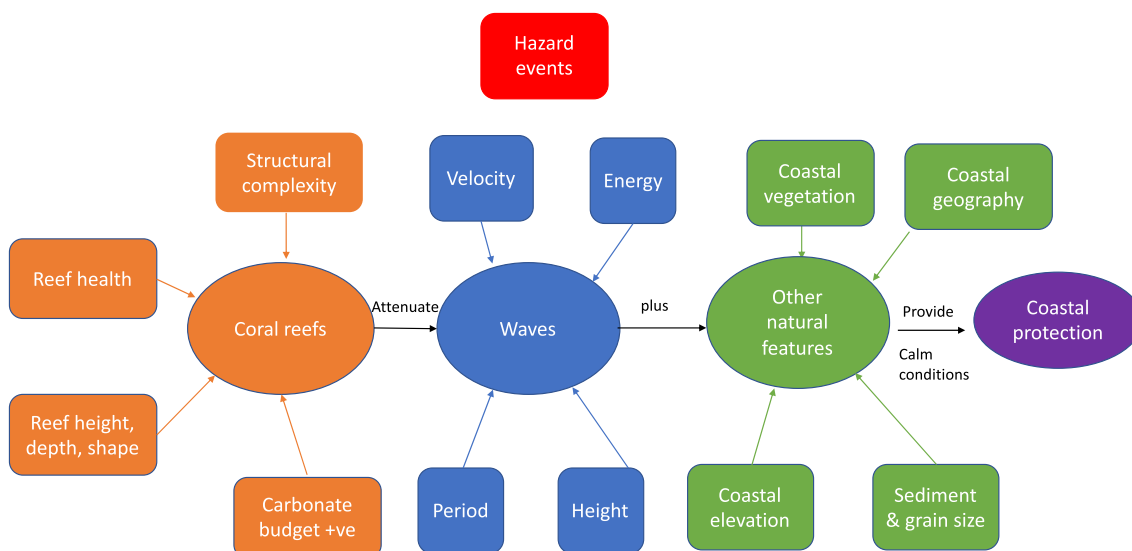


Fig. 1. Natural features that interact to deliver coastal protection. Shapes in orange represent coral reefs and their different ecological parameters. Shapes in blue represent waves and their physical processes. Shapes in green represent other biophysical features that can impact service provision.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

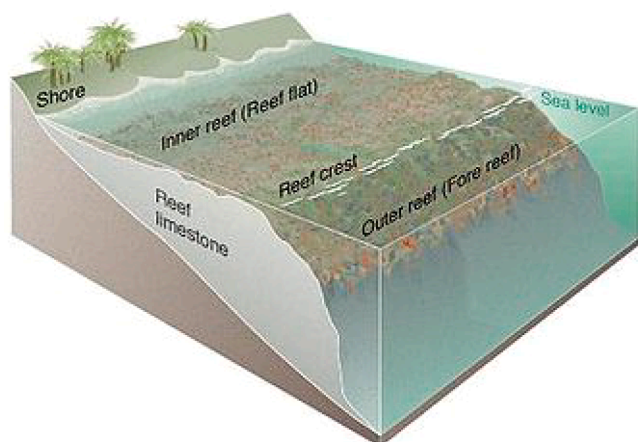


Fig. 2. Cross section of a fringing reef. Image from (U.S. Geological Survey Fact Sheet 025-02). Retrieved September 16th, 2020.

to less so (2.6 kg CaCO₃ m⁻²y⁻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

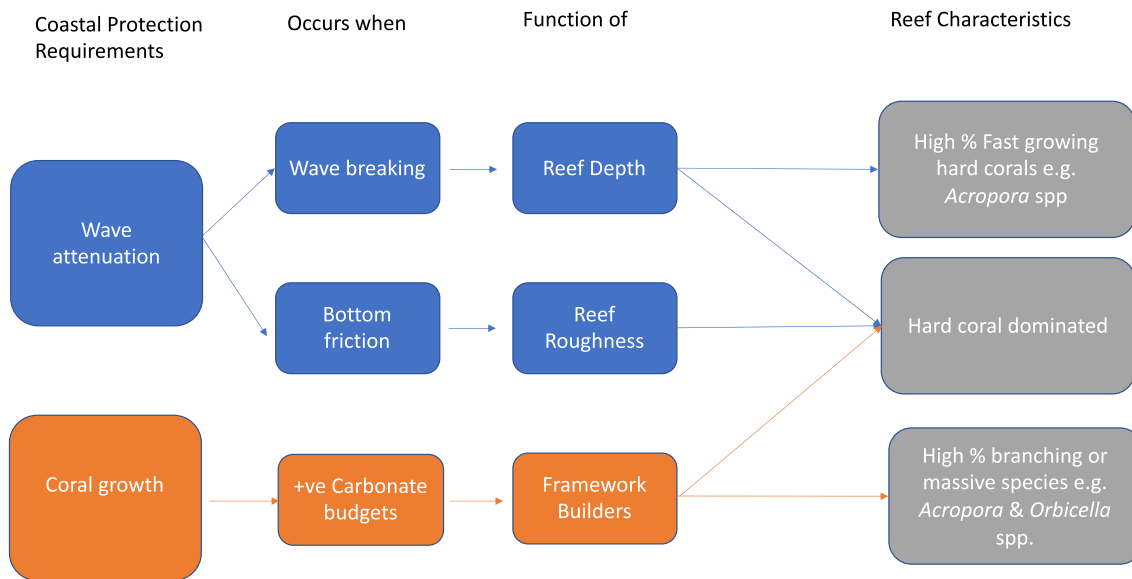


Fig. 3. Summary of key coastal protection requirements and primary coral reef characteristics for their provision.

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

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 of Corals	Macroalgae	Removal of diseased colonies
	Outright Mortality	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).

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 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	
Ecosystem Service	Biological	Monismith et al., 2015, Lowe et al., 2007 Wainger and Boyd, 2009 Reguero et al., 2018, Kushner et al., 2011 Beck et al., 2018, Ferrario et al., 2014
	Reef rugosity	
	Fish abundance	
Socioeconomic	Beach Erosion	Storzazzi et al., 2019, Van Zanten et al., 2014 Burke et al., 2008, Pascal et al., 2016
	Coastal Inundation	
Socioeconomic	Avoided damages	Ferrario et al., 2014, Beck et al., 2018
	Property values	
	Breakwater replacement costs	

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

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
Geographical boundary	Beach erosion	m/yr	C	Sand lost from the beach due to wave energy
	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.

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

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 (Topopova 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; Posingham 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 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). Principe 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.

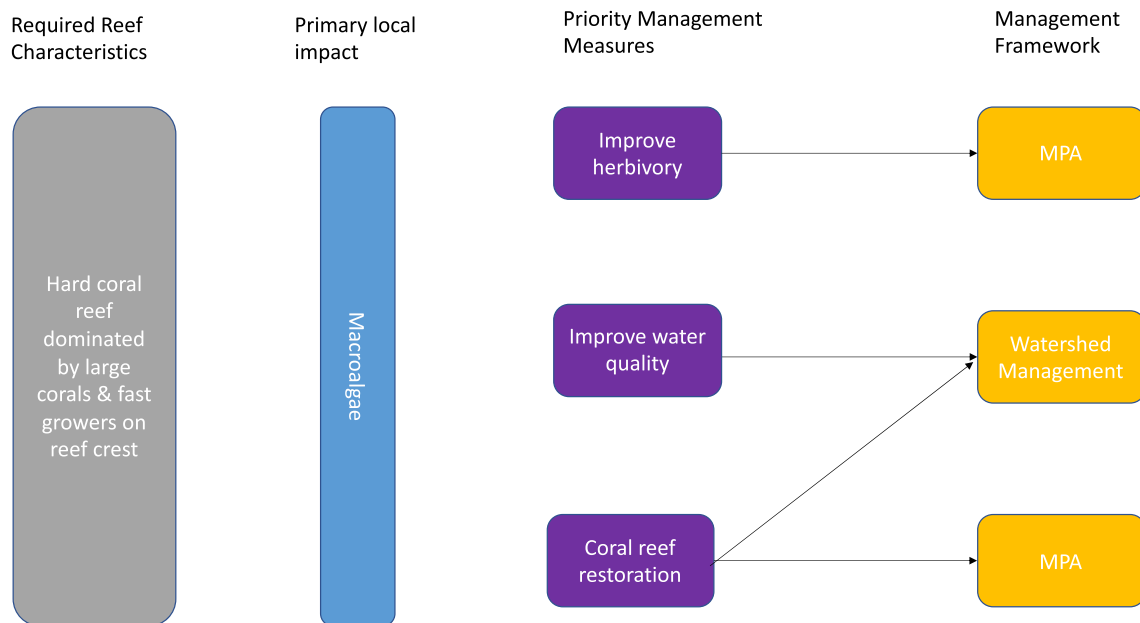


Fig. 4. Summary of primary impact, priority management measures and management frameworks to deliver positive outcomes for coastal protection.

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 <i>Acropora</i> spp and <i>Orbicella</i> 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 ICZM³ 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 <i>Acropora</i> 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 seller's 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

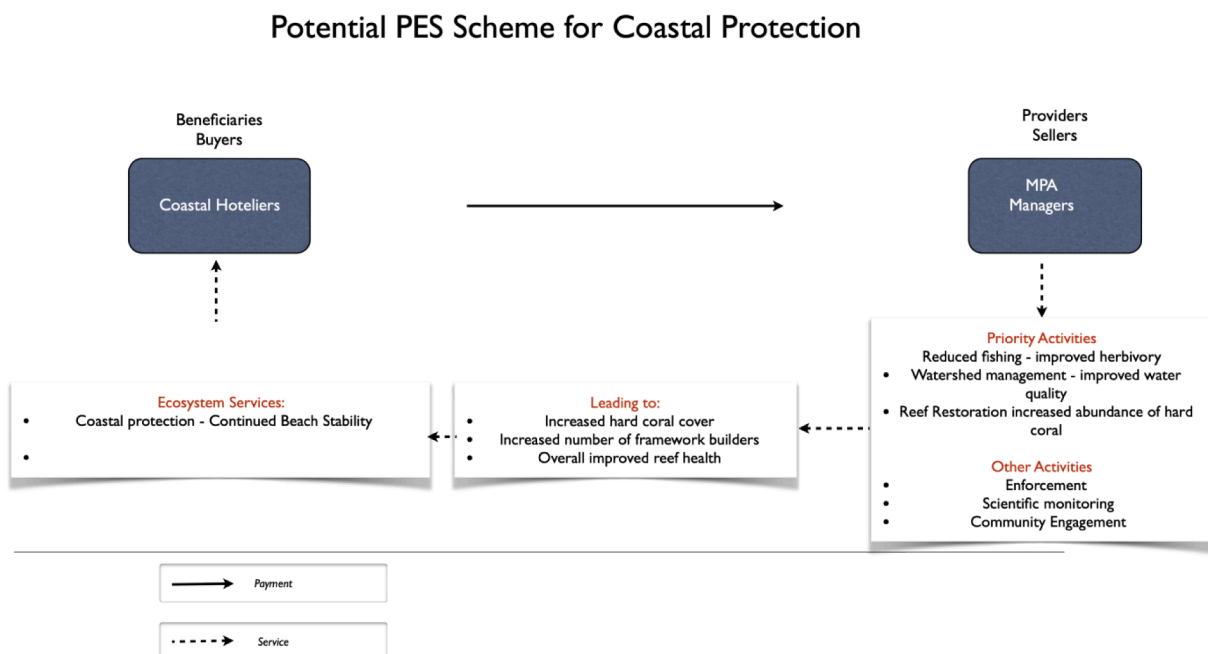


Fig. 5. Potential PES Scheme between buyers (coastal hoteliers) and sellers (MPA managers) of coastal protection.

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 impending 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.

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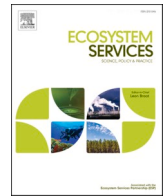
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Chapter 5: Impact Investment in marine conservation



Impact investment in marine conservation

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ABSTRACT

Marine Protected Areas (MPAs) are promising examples of Nature-Based Solutions that can protect diversity while delivering ecosystem services. However insufficient funding for effective management and expansion of MPAs remains a challenge and one that particularly affects developing countries. During the last ten years, a community of investors seeking positive social and environmental impacts alongside financial returns, have stepped in to help fill this marine conservation financing gap. An innovative collaborative management approach has been recently implemented in the Dominican Republic for one of the largest MPAs in the Caribbean. Blended finance solutions mixing catalytic, development and impact finance have been used to cover the up-front capital needs. MPA revenues are being generated for MPA management and investor returns, via a range of sustainable finance tools including fees paid by visitors. The solution offers interesting outcomes that uses catalytic and development finance to mobilise commercial impact finance into MPAs. From a Government point of view, the approach provides empirical evidence of how non-public funding can become part of the financing options for a country's MPA network, reducing the financial burden on Public Budgets. Scalability of the approach seems limited by the number of MPAs with tangible business models.

1. Introduction

The world is reportedly entering the Anthropocene (Steffen et al., 2011) with unprecedented impacts on our ocean, leading to a heightened awareness of marine matters in national and global agendas (Claudet et al., 2019). Initiatives such as national Ridge to Reef programmes and the UN 'Decade of Ocean Science for Sustainable Development' (begins 2021) for example are aimed at mitigating such threats. Such positive intentions must however be coupled with concrete initiatives that both protect the ocean from its many stressors and involve marine stakeholders whose livelihoods depend on it. Marine protected areas (MPAs) are considered the most promising tools in the management toolkit for protecting ecosystems and allowing for their services to be utilized in sustainable ways (Halpern, 2003; Angulo-Valdes and Hatcher, 2010; Mumby and AR, 2010; Pascal et al., 2018a). For MPAs to be successful however, effective management fueled by adequate financing is essential (Edgar et al., 2014; Watson et al., 2014; Gill et al.,

2017). The advantages of MPAs go further than simply protecting ecosystems. Additional benefits include the provision of food and incomes for local communities, enhanced resilience to climate changes impacts and supporting ocean processes (Angulo-Valdes and Hatcher, 2010; Leenhardt et al., 2015). Further they serve as a means of achieving national biodiversity targets and Sustainable Development Goals. The United Nations' Convention on Biological Diversity (CBD) for example established a target of 10% of the ocean to be protected by 2020 ('Aichi Target 11') and the UN's Sustainable Development Goal 14 (SDG 14) adopts and reinforces this commitment. However insufficient funding for effective management and the called for expansion of MPAs (Aichi Target#11) remains a challenge; one that particularly affects developing countries (Marinesque et al., 2012). Globally, 65% of MPAs are estimated to have inadequate management budgets and 91% to have inadequate staff capacity (Gill et al., 2017). Rapid expansion without the necessary investment could see an explosion of 'paper parks', which fail to meet social or ecological goals and cannot financially sustain

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themselves.

Impact investing: strong growth but poor penetration into marine conservation

The concept that economy and ecology can be mutually beneficial was born in the 1970s and evolved in the 1980s to include discussions on sustainable development (Huwylar et al., 2014). Businesses began reducing environmental damages by engaging in “corporate social responsibility.” (Stubbs, 2016). In the 1990s, the term “triple bottom line” to denote economic, ecological, and social performance became a popular catchphrase among businesses that aimed for more than just financial profits. Economists refer to the triple bottom line as “utility maximization,” where utility can include economic, environmental, and social targets (Littlefield, 2011). OECD describes blurring the lines between natural, financial, and social capital and aiming for “blended value.” (OECD, 2010)

The latest iteration in this trend is “impact investing,” which is defined by the Global Impact Investing Network (GIIN) as “investments made into companies, organizations, and funds with the intention to generate social and environmental impact alongside financial return” (GIIN, 2016). This is the definition used in this article, where the term “impact” refers to positive impacts or benefits such as cleaner water, more jobs, or greater protection for species. Unlike corporate social responsibility, which tries to reduce negative impacts of firms’ economic activity, impact investing is characterized by the intent to produce net positive environmental or social outcomes (Forest Trends, 2016).

There is available money/revenue in Impact Investment. The GIIN reported deals worth US\$35 billion in 2017; up 17% from 2016. At the close of 2018, the industry was worth an estimated US\$502 billion (GIIN, 2019) with investments concentrated on energy (15%), micro-finance (13%) and other financial services (11%). A small but growing proportion of impact investments focus on environmental impacts. A recent study estimated that US\$8.2 billion has been invested in conservation efforts since 2004 (Forest Trends, 2016). A 2016 report by Credit Suisse and McKinsey Center for Business and Environment estimated that up to US\$200–400 billion between 2016 and 2020 might be invested in conservation finance (Credit Suisse AG, 2016). However, conservation projects, particularly around marine ecosystems, are not attracting impact capital at the same pace as the rest of the impact investment market, largely due to a shortage of investment-ready projects and the development of future opportunities (Bos et al., 2015). With development finance being a scarce and precious resource, the mobilization of additional funds from commercial investors into marine conservation is indispensable for meeting the financing needs of Agenda 2030 (Dent et al., 2017).

Collaborative management and impact investment

Making MPAs financially sustainable and independent from traditional funding (grants, public money) offers a potential path towards solving the financing gap. This can be achieved by developing a revenue generation model that covers the operational costs of the MPA, which is referred as an Entrepreneurial Marine Protected Area (EMPA) model (Bos et al., 2015).

The few MPAs in the world that have applied this model often use a collaborative management structure between government and non-public entities (Jones et al., 2011). Collaborative management (or public–private partnerships) has been described by numerous authors and is essentially “an arrangement where non-profit organisations partner with state wildlife authorities for the management of a state protected area” (Baghai et al., 2018). A large body of evidence from marine and terrestrial protected areas shows that, when collaborative management is viable and appropriate, it can both redistribute the financial burden on states and attract the long-term economic and technical support needed for effective management (Shurcliff, 2001; Saporiti, 2006; Svensson et al., 2009; Jones et al., 2011; IUCN and UNEP, 2014). Positive experiences with nature conservation in parts of Africa, shows that they can improve service through professional management and marketing, reduce the need for public subsidies, and

mobilize capital for investment in park infrastructure and biodiversity. Pascal et al. (2018a), Pascal et al. (2018b) presented a list of marine protected areas under collaborative management with non-profit entities in the Caribbean, and discussed a select few. Their research showed that many of these protected areas succeeded in becoming financially sustainable, while also demonstrating positive social and ecological impacts.

By using innovative revenue mechanisms, such structures have the potential to attract further investments and diversify MPA funding sources by catalyzing private capital through impact investing for example. However, the field of marine conservation finance is still at an early stage of development and very few examples of MPAs implementing these impact investment tools exist.

In this paper, we present an innovative approach that uses catalytic and development finance to mobilise commercial impact finance for Marine Protected Areas (MPAs) under collaborative management.

The approach was developed in the Dominican Republic’s “Arrecifes del Sureste”. The collaborative agreement was signed in 2019. The impact investment solution was structured in late 2019 with impact investors and is in its concluding phase at the time of this publication.

We first present the Dominican Republic’s case study with a description of the collaborative management agreement, the business model and the impact investment, followed by a discussion on the potential of up-scaling the approach for other MPAs and the precautions to be taken.

2. Context & Method: The Dominican Republic project

2.1. Context

The MPA “Arrecifes del Sureste” in the Dominican Republic (DR) is almost 8000 km², covering approximately 100 km coast and encompassing coral reefs, mangroves and seagrass ecosystems, several major urban centers and 2 of the country’s primary tourism centers (receiving > 3 million visitors annually). The MPA was designated in 2009 and has been mostly inactive since, hence a classic example of a “paper park”.

The growth of the coastal tourism industry in the country was characterized by a lack of planning and oversight from the government causing unregulated coastal development. This, combined with overfishing, pollution, poor tourism practices, lack of marine zoning and surveillance have seriously degraded the health of the marine ecosystem of the area.

In 2018, the government of the Dominican Republic decided to design a collaborative management approach with impact investment, to improve the effective management and sustainable financing of the MPA. This initiative was led by the Ministry of Environment with assistance from United Nations, the NGO Blue finance and local partners.

2.2. Method

The design of the collaborative management agreement for the DR MPA with an impact investment solution was based on a combination of a desk study as well as interviews with local stakeholders. The desk study reviewed the existing literature on:

- a) Collaborative management experiences for Protected Areas worldwide including design recommendations (Carlsson and Berkes, 2005; Saporiti, 2006; Svensson et al., 2009; Jones et al., 2011; Zurba et al., 2012; Hatchwell, 2014; Lindsey et al., 2014; Regan et al., 2016) and in the Dominican Republic,
- b) Documented MPAs with successful business model (Saporiti, 2006; Svensson et al., 2009; Lely et al., 2013; Gill et al., 2017; Pascal et al., 2018a; UN Environment, ISU et al., 2018) and,

- c) Conservation finance guides with a focus on protected areas (Bovarnick et al., 2010; The Katoomba Group, 2010; UNDP and GEF, 2012; The World Bank, 2014; Walsh et al., 2016; UNDP, 2018).
- d) Official statistics, economic reports, market surveys and technical reports to collect data on the socio-economic context of the MPA, the threats to the marine ecosystems, the tourism market, and MPA user fee levels in the country and in the region.

A census of NGOs, tourism associations, tourism service providers and community organisations was conducted through field visit and interviews, to facilitate a clear vision of the different existing stakeholders implicated in the conservation of natural resources.

Semi-directed interviews with local tourism stakeholders (4 diveshops (representing a sample of 30% of all diveshops), 3 day-tour service providers (representing a sample of 30% of all service providers) and 2 tourism associations (100% of the associations) contributed to complete the assessment of the market potential for the revenue mechanisms of the MPA. Data collected comprised of descriptive attributes of their business and clients, market trends, perceptions of environmental issues and recommendations on the price level of the statutory user fees and elements of strategy to implement successfully the fees with their clients.

2.2.1. Collaborative management agreement design

The design of the collaborative management agreement for the MPA in Dominican Republic has incorporated several components which include: (a) Engagement – There are local capacities to drive the social, environmental and business process forward; (b) Achievability - The intended project outcome has a high probability of success for the immediate management activities; (c) Business model - the challenges and opportunities to produce revenues from the different markets are identified and quantified; (d) Investible - Potential leverage from Development Financial Institutions (DFIs), impact investors and donors with an adequate risk-return-impact ratio; (e) Impact - the ecological and socio-economic impacts of the project are measurable through Key Performance Indicators (KPIs).

Globally, collaborative management agreements have been structured in various ways (Baghai et al., 2018; Delmon, 2010; The World Bank, Asian Development Bank et al., 2014): (a) ‘a management lease’ or ‘Public-Private Partnership’: government entrusts another entity with full management responsibilities for a Protected Area; (b) Co-management: management responsibilities might be shared with government and one or more actors; (c) Cooperation or Technical Assistance: a partner might assist government with specific aspects of management, without formal decision-making authority. There is also the possibility of an MPA being privately managed by private actors, but this has rarely been the case to date (Christie and White, 2007). In the DR, the legal framework for collaborative management (Ministerio de Medio Ambiente y Recursos Naturales Viceministerio de Área Protegidas y Biodiversidad, 2015) allows for development of these 3 primary categories as described above (the management lease category is nominated “delegated management”).

The design proposal has to define how the protected area is to be governed and managed. Within it, governance described who has the power to set overall priorities and strategies (including oversight, and setting management priorities), and how such decisions are made. Management described the practical, day-to-day implementation of governance decisions (including hiring and firing of staff, law enforcement operations and revenue generations). The following description of the different collaborative management models is based on several authors (Carlsson and Berkes, 2005; Rutagarama and Martin, 2006; Baghai et al., 2018). The first model falls under a management lease agreement where the management is entirely delegated (incl. hiring and firing of staff) whereas specific aspects of the governance are shared (such as annual work plans). Typically a new structure is specially-created as a non-profit entity by the nonprofit partners. Financing is usually provided by non-profit partner with revenues “ring-fenced” at protected

area level. The second model falls under a co-management agreement, where both the management and governance are shared. The agreement works usually under a bilateral management with public entities and non-profit partners working alongside each other. Financing can be provided by non-profit partner; with or without revenue retention at protected area level. The third model falls under Technical Assistance agreement, where both the management and governance remain under government. The levels of non-profit partner financing is very varying; protected area revenues generally go to central government. Authors highlighted that distinctions between the 3 models are in many cases not so clear.

2.2.2. Tangible business model design

The identification of revenue streams to repay investors, is one of the primary limiting factors to attracting up-front, impact investment financing to marine conservation (OECD, 2018; Cooper and Trémolet, 2019). The instruments and options available for generating “tangible” revenues for the MPAs in the short to medium term (up to 5 years) are mostly limited to revenues linked to tourism (Meyers et al., 2020). Other revenues have a longer incubation term and limited track record and are still considered as conceptual by impact investors (Dent et al., 2017). These revenues include business fees paid by sustainable fisheries and aquaculture, ‘blue’ carbon credits, biodiversity offsets and payment for ecosystem services (The Katoomba Group, 2010; UNDP and GEF, 2012; Agardy and Pascal, 2014; Conservation Finance Alliance, 2014; EKO and TNC, 2014; The World Bank, 2014; Credit Suisse AG, 2016; UNDP, 2018; Meyers et al., 2020).

This dependency on tourism revenues in the short term is different from that of terrestrial protected areas, with a larger portfolio of potential revenue streams, including sales of timber, carbon offsets, other forest products, land rights for permanent conservation (e.g. easements) and land leasing (GIIN, 2019).

Studies show that most visitors would be willing to pay an entrance fee (user fee) to marine sanctuaries if their expectations are fulfilled (Wielgus et al., 2010). Willingness to pay MPA entrance fees will depend on visitors’ income, level of education, environmental awareness, residency and desire to provide a legacy to future generations. From the tourism businesses’ perspective, a previous work (Green and Donnelly, 2003) has conducted surveys with dive operators on their MPA use in 30 countries. The authors found the majority are located within 20 km of at least one MPA and 46% conduct at least 80% of their diving within an MPA (Green and Donnelly, 2003). Several studies (Bacci, 1998; Ahmad, 2009; Wielgus et al., 2010; Imran et al., 2014; Schuhmann et al., 2016) have shown that, for most visitors, MPAs are the trademark of an ideal marine environment.

A good example of an effective user fee comes from the Bonaire National Marine Park where all visitors are required to pay an entrance fee, known as the ‘nature fee’. In 2017 for example, the park earned US \$1.5 million in income from foreign and local visitors, providing 90% of STINAPA Bonaire’s funding for the park (Pascal et al., 2018a).

In addition to user fees, mooring fees (vessel owners pay a fee to use the moorings in the MPAs), license/permit fees (fees paid by companies providing a service to visitors in the MPA) and visitor centre entrance tickets can contribute to the revenue stream (Conservation Finance Alliance 2003).

The potential revenues of these business models linked to tourism have been assessed through (a) estimating the number of existing users of the MPA (i.e. divers, snorkelers, day-tour passengers) and potential users of new attractions such as the visitor centre, and (b) proposing user fee prices for divers and snorkelers based on the interviews completed with the tourism stakeholders (described above). The semi-directed interviews also informed on the best strategy for fee implementation and increased acceptance by their clients.

Carrying capacity concepts have been incorporated in the models and projections. Different tools such as carrying capacity, maximum load capacity and limits of acceptable change (LAC) are the most

commonly used to determine ecological sustainability of tourism (Stoeckl et al., 2010; Lim, 1998). Like any ecosystem, coral reefs are capable of regeneration from minor insults. However, the cumulative effect of a number of impacts can create irreversible damage, especially when threshold levels are exceeded. Controlling tourism numbers is therefore important to overall health of the ecosystem that is attracting visitation.

2.2.3. Business plan and positive impact indicators

The MPA direct annual operating costs and initial capital expenditures have been assessed following categories described by several authors (Beukerlin et al., 2003; Balmford et al., 2004; Pascal et al., 2018a). The direct operating costs include salaries & wages, programme activities (e.g. livelihood programmes with fishers, communication & awareness), operation (fuel & transport), maintenance and overhead costs (office, utilities, accountancy & audit).

The capital expenditures refer to the purchase of the physical assets of the MPA such as vessels, moorings, equipment, underwater facilities, etc. and preliminary studies.

Profit & Loss statements as well as cash-flow projections incorporating the financing options have been produced through consolidated financial modelling techniques (Campbell and Brown, 2003). The MPA capacity to become self-financing (i.e. incomes balancing all the costs, 'break-even point') has been assessed.

A common set of Key Performance Indicators (KPIs) and result metrics has been developed with and for impact investors. Based on International Finance Corporations (IFC) Performance Standards, the KPIs outline the positive Environmental, Social and Governance (ESG) impacts of the investment. The initial list of KPIs includes conservation impacts, social impacts and management impacts. Management

effectiveness audits (which also have social and environmental elements) are proposed to be based on the Management Effectiveness Tracking Tool (METT), a generic system developed to assess protected area management effectiveness (WWF, 2007). Reporting arrangements (frequency, content, audit) as well as responsible agency for the monitoring have been suggested.

2.2.4. Blended facility design for impact investment

Investment barriers for private investors into marine conservation include (i) high perceived and real risk and (ii) poor returns for the risk relative to comparable investments (Huwylar et al., 2014; Dent et al., 2017; OECD, 2018). De-risking strategies are therefore crucial to unlock private capital into new initiatives such as Marine Protected Areas (O'Donohoe et al., 2010). Blended finance is 'the use of catalytic capital from public or philanthropic sources to increase private sector investment in sustainable development' (Convergence, 2018). It is a structuring approach that allows to de-risk transactions and create investable opportunities for more development impact. OECD (2018) has identified several designs of blended finance archetypes to attract impact investment where public or philanthropic investors provide funds to increase the protection or guarantees to private investors or provide technical assistance to strengthen commercial viability and developmental impact. The design of the blended finance facility for the MPA in the DR has built-up on the principles exposed in several guidance documents (Forest Trends, 2016; CFP, 2017; Convergence, 2018; OECD, 2018; Whisnant and Veerle, 2019).

3. Results

The major elements of the design of the collaborative management

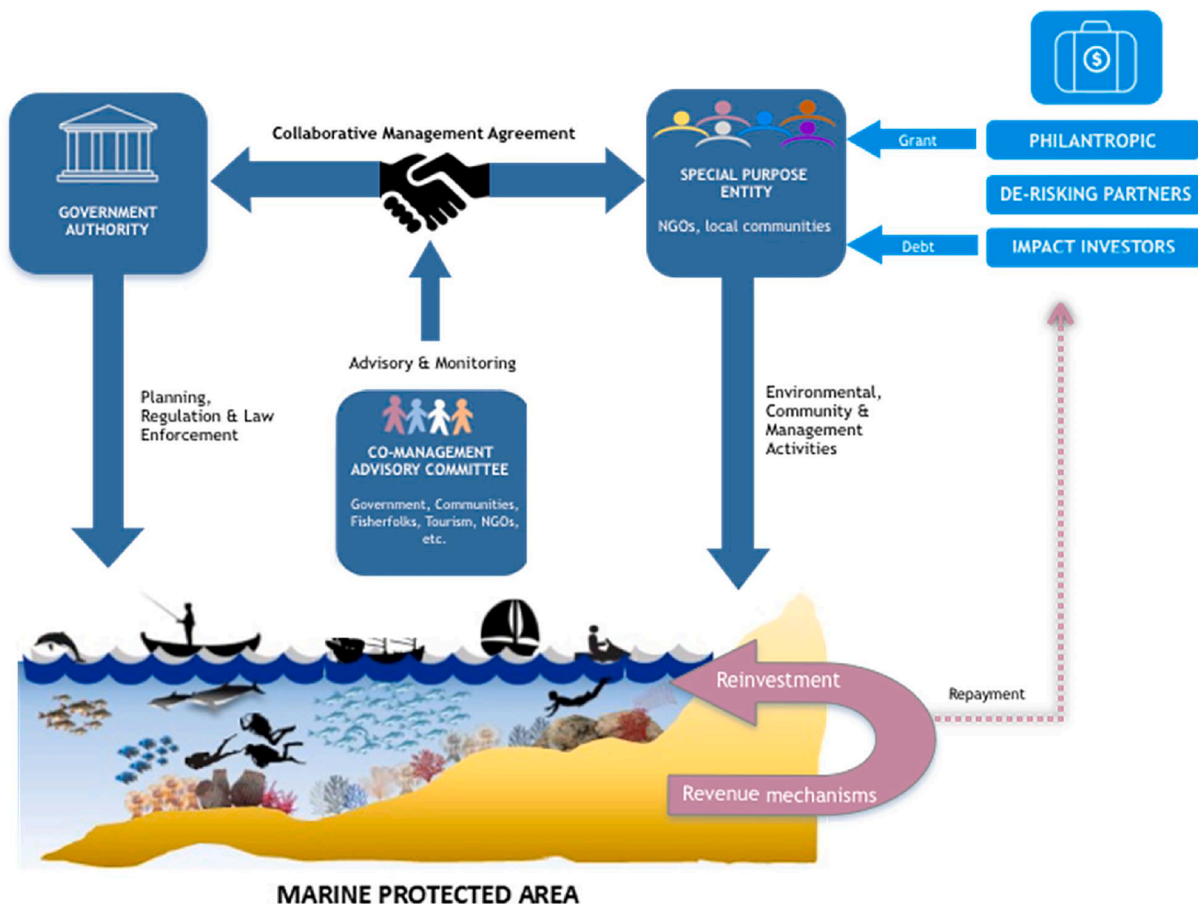


Fig. 1. Summary mechanism for the collaborative management and blended finance solution for the MPA.

approach, the business model and the impact investment are presented in this section (see Fig. 1). The agreement was signed in 2019 and memorialized in a contract.

3.1. Design of the Collaborative management scheme

3.1.1. Governance model

The government maintains its core functions and is responsible for regulations, zonation, enforcement, establishment of user fees and maintenance of specific on-shore facilities. The functions and staff of public agencies (e.g. Coast Guard, Marine Police, Fisheries, Environment and other government agencies) are therefore maintained and their work continued.

A non-profit ‘Special Purpose Entity’ (SPE) was established to manage day-to-day operations. The SPE is comprised of local conservation NGOs, local foundations of the major tourism holdings in the country and other associations.

Participatory management is facilitated via the establishment of a Stakeholders Advisory Committee (SAC). This allows for the provision of regular inputs into the zoning, management, uses and enforcement process. Its membership includes government agencies, tourism associations, local communities, fisherfolk, hoteliers, developers, research institutions, NGOs, civil society and the boating community.

3.1.2. Enforcement

In the DR, legislation allows for the appointment of non-government officers with various powers, including searching and detaining for marine infractions. The enforcement plan for the MPA is being developed collaboratively between the MPA staff and regulatory agencies. In addition to these measures, self-enforcement from Fishers and other stakeholders will be encouraged through education and with financial incentives.

3.1.3. Activities

The SPEs scope of activities are clearly established in the Collaborative management Agreement. With the approval and guidance of Government and in consultation with local key stakeholders (through the SAC), the SPE is responsible for developing and implementing annual work plans with the following 7 primary activities: (i) MPA Environmental Protection and Management (including reducing anthropogenic nutrients; Sewage and storm water management, mooring & demarcation buoys; Erosion/sediment reduction; Reduction of the Invasive Lionfish; Reduction of Solid Waste); (ii) Livelihood Enhancement for Fishers (including supporting income generating activities for fishers that might be displaced; Train local communities members as tourism service providers for eco-tours and fish to table businesses; Train and hire selected fishers in fishery monitoring and coral gardening); (iii) Stakeholder Engagement (including actions to recognise, respect and utilise local knowledge for management; Collaboration with Existing Institutions and Private Sector); (iv) Communication, Education, Awareness and Visitor Experience (including increased awareness, understanding and support for the MPA; Provide products and services for visitors and residents; Educational Programmes and Training; Marketing of the MPA); (v) Monitoring & Science (mainly monitoring coral reef and fish population health, water quality, socio-economic development); (vi) Compliance (including activities ensuring that marine management area users comply with the rules; Design of a Marine Protection System, Training & Engagement, Fish Warden Programme) and; (vii) Administration and Revenue Generation (including maintenance, fee management, visitor centre management). The SPE will also receive long term technical support from external experts in order to improve both environmental management and entrepreneurial skills.

3.2. Monitoring of positive impacts

A common set of Key Performance Indicators (KPIs) and result metrics has been developed with and for impact investors, while specific reporting arrangements have been set-up to guarantee that the SPE is implementing activities as expected. An initial list of KPIs of conservation impacts (e.g. biodiversity metrics on coral reef cover), social impacts (e.g. livelihood improvement for fisher households, gender equity) and management impacts (e.g. MPA performance indicators, METT) have been agreed to with partners. Initial data will be collected to provide baseline indicators and Environmental, Socio-economic and Management audits will be carried out annually by the SPE to assess the KPIs. These will be supplemented by external audits, carried out biennially by independent members.

Semi-annual reporting will be submitted to the Government, the collaborative management advisory Committee and the investors. Reports will describe progress, issues, recommendations, financial status and variance in the KPIs.

3.3. Business model

3.3.1. Revenues

Revenue streams will be generated primarily from visitors via statutory visitor fees and innovative tourism activities (see Fig. 2).

In 2019, 3 million guests have stayed-over in hotels & accommodations in the vicinity of the MPA, of which 260,000 have dived or participated in water activities in the MPA. The SPE has received an irrevocable mandate to charge user fees to divers and non-divers (such as day-tour passengers and water sports). Fees range from US\$3 to US\$10 per day per user depending on the activity. No major concerns were raised by local service providers in charging user fees to their divers and snorkelers. The user fee has been set at the total price paid by the client for the service (e.g. diving cost). The businesses agreed that clients will not have an issue in accepting an increase in the tour price if tangible “value for money” is provided. Vibrant biodiversity, clear communication, adequate facilities and moorings, clean beaches, underwater trails, multimedia apps and eco-activities were recognized as valuable arguments for price increases.

Voluntary contributions from hotel guests will also contribute to the

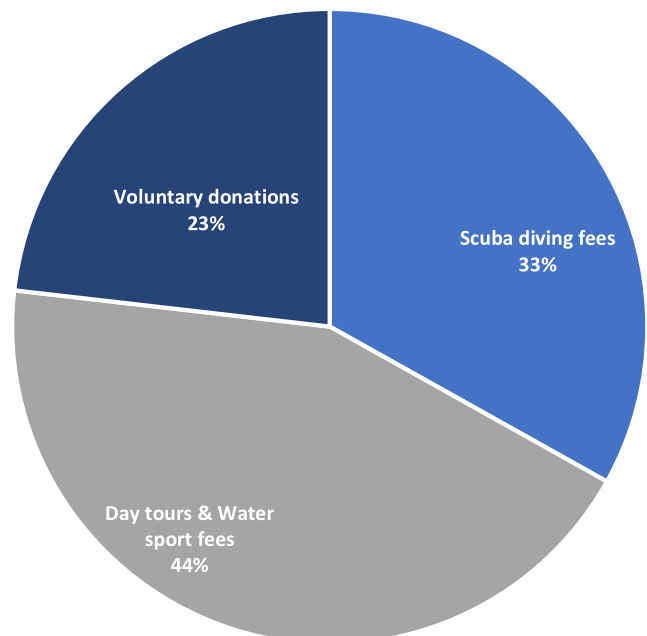


Fig. 2. Distribution of expected incomes for the collaborative management entity.

MPA, while innovative tourism products (e.g. high tech visitor centre) will be assessed in the near future for their ability to effectively contribute to the revenue stream.

Concerns about exceeding carrying capacities at sites were considered in the projections. The number of dive sites and the large choice of day-tour stop-overs allows for the reduction of visitor impacts to below recommended carrying capacity for each site.

3.3.2. Costs

Annual operating expenditures are expected to total US\$1.4 million for MPA activities, which will be guided by annual work plans and budgets. The main costs are personnel (almost 55% of the total Opex), livelihood programmes with fishers, communication and enforcement/compliance (see Fig. 3). The management team will include 18 permanent staff progressively recruited during the 2 years following the start of operations.

The up-front investments of the SPE are estimated at US\$3M which will be spread over a 2-year period and includes: (i) purchase of the physical assets of the MPA such as vessels, moorings, equipment, underwater facilities, etc. and preliminary studies; (ii) working capital (corresponding to the period of preliminary activities of the SPE without incomes) and (iii) a visitor centre (in year 2 and upon confirmation of the business plan)

3.3.3. Financial results

Based on the financial statements of Profit & Loss and cash-flow projections, the SPE is expected to become self-financing (i.e. incomes balancing all the costs, 'break-even point') in Year 3. All surplus will be reinvested in the MPA annual work plans.

3.4. Design of the financing facility

The CAPEX financing has been structured through a blended facility: (i) loans from impact investors; (ii) concessionary loans from Development Financial Institutions (DFIs); (iii) grants from Philanthropic institutions and; (iv) donations from the SPE founder members. In addition, a de-risking partner provides loan guarantees to the lenders to mitigate credit risk (50% shared loss facility on invested capital). Specific conditions have been agreed to with impact investors regarding the

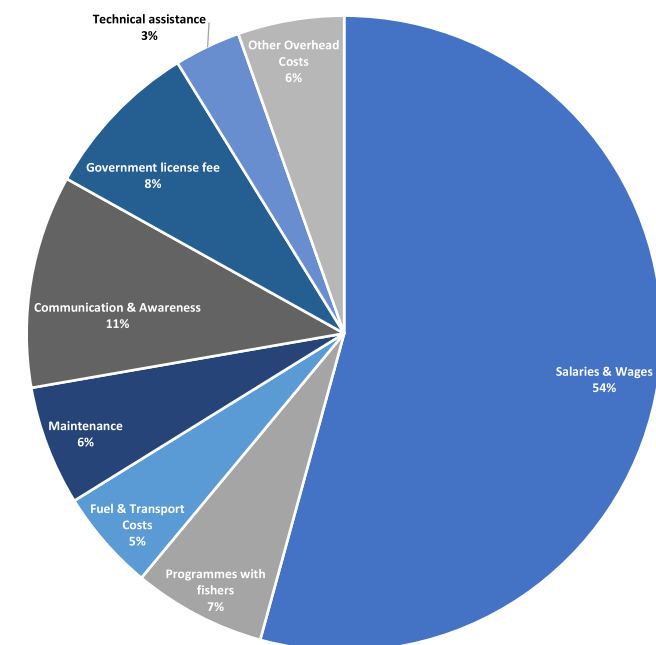


Fig. 3. Distribution of annual operating costs for the collaborative management entity.

tenor and repayment schedule to reduce the financial burden for the MPAs. Currently, capital is being used to hire staff and purchase the required equipment. The development of the Management and Marine Spatial Plans are also in process. Technical assistance for the preparation of the collaborative management agreement and investment solution has been provided by a specialized company since 2018. All financial risks are supported by the collaborative management SPE, with neither increase in public debt, nor on-public budget allocation.

4. Discussion

If proof-of-concept can be demonstrated at scale, this project can serve to address the MPA financing gap as well as to build confidence and catalyze wider growth in the natural capital impact investment market globally.

The solution developed for the MPA in the DR has relied on 2 primary pillars: a collaborative management agreement for Marine Protected Areas (MPAs) and tangible revenue models with positive impacts. In the present discussion we will assess if the enabling conditions behind this approach can be effectively tailored to the varying contexts and needs encountered at many different sites.

4.1. Collaborative management model

The model chosen for the DR is sharing many elements with the management lease model (or delegated management), where the management of the MPA is given entirely over to a non-profit body whereas the governance is shared between this entity and government (Baghai et al., 2018). DR government continues fully its regulatory and oversight role guiding the way in which the MPA is developed and managed. Even if revenues, both user fees and investments are retained at the MPA level, government has been able to attract and develop new resources for the country without any increase in public budget.

The management lease model seems adequate in regards to the following aspects: (a) the lack of present and future public resources and government personnel to be dedicated to the MPA - making difficult any form of co-management or technical assistance approach, (b) impact investor's requirements to establish a clear and transparent vehicle to channel revenues and financing (Pascal et al., 2018a), (c) the needs for adaptive management and entrepreneurial capacities to maximise revenue strategies - usually not compatible with bureaucratic constraints and political interference (Lindsey et al., 2014) and (d) the ability to attract capital (including impact investments) to cover start-up costs without increasing burden on public budgets - funders can sometimes be reluctant to invest where governments are not sharing management activities (Saporiti, 2006; Baghai et al., 2018).

4.2. Collaborative management of MPAs

According to several authors (Saporiti, 2006; Pascal et al., 2018a), collaborative management with non-public partners bring various advantages, including: (i) A business approach: establishing revenue streams around ecosystem services, most commonly nature tourism; greater capacity and expertise to develop, market and manage commercial operations and maximise revenues; (ii) Greater freedom to retain and reinvest profits than government agencies, giving managers incentives for cost saving, accountability and improved management and; (iii) Greater ability to raise capital to cover start-up costs, such as restoring the ecosystem, purchasing equipment and developing visitor facilities.

In a similar way, different studies (Uyarra et al., 2010; Wang and Jia, 2012) have shown that trust in protected area authorities significantly affected Willingness to Pay from visitors. Most tourists interviewed preferred NGOs to the government as the most trustworthy organization type to collect and manage entrance fees (Buckley, 2003).

Precautions have also been highlighted by several authors when

implementing collaborative management approaches for Protected Areas (Ferraro and Kiss, 2002; Delmon, 2010; Jones et al., 2011; UNDP, 2018; Meyers et al., 2020). For IUCN (IUCN and UNEP, 2014), the public sector is and will remain a critical actor/investor in natural capital as dealing with goods that generate public benefits beyond private income streams. Collaborative management agreements necessitate a balance between resource protection, economic development and the financial bottom line (Stolton et al., 2014). Government policy and regulations may require that management provided by non-public entity consider public opinion, provide employment for local people and other elements of benefit to local communities, and include regular monitoring of environmental impacts (Ferraro and Kiss, 2002). Collaborative management requires also a transparent process with the process usually requiring the set up of a competitive bidding process and of a quality control process to oversee external partners' performance (Saporiti, 2006).

4.3. Tangible revenue model

4.3.1. Incomes linked to tourism

Precautions must be taken. Several authors (Font et al., 2004) recognize that MPAs need to be realistic about the potential to raise funds from tourism. Many sites tend to overestimate the benefits and underestimate the costs of tourism: in part, this is because they often fail to establish proper management for tourism, and to understand the business realities of tourism. Past experiences have shown also that the implementation of fee systems has to be carefully designed and communicated to avoid fee evasion as well as lack of stewardship from the marine stakeholders. Common issues include fees deemed too high in regards to services received by visitors, lack of transparency in the use of funds and/or inefficient fee collection system (Lindberg and Aylward, 1999; Brown, 2001; Schuhmann et al., 2016). Initial investments in the MPAs is needed to improve the services offered by the MPAs, as well as increasing compliance to the fee payment.

4.3.1.1. Other revenues. Other sectors are capable of driving sustainable income into MPAs and can be bundled within the structure of site-level projects:

- (i) Blue carbon credits (Pendleton et al., 2012; Forest Trends Ecosystem Marketplace and Bloomberg New Energy Finance, 2014): the credits are generated and sold to organizations seeking to meet voluntary or regulatory climate mitigation targets. MPA Carbon credit buyers would benefit from the emission reductions provided, as well as the co-benefits associated with MPA projects (e.g. community livelihood enhancement);
- (ii) Payment for Ecosystem Services (PES): PES is an emerging tool intended to solve a range of management inefficiencies by linking constructive conservation action to payments (Naeem et al., 2015). Such positive incentives will either compensate those who are providing the service, or stimulate those not providing the service into doing so (Sommerville et al., 2009). Under such a scheme, possibilities exist for beneficiaries of coral reef ecosystem services to make payments to providers, such as MPA managers who will carry out a variety of management actions aimed at improving reef health and ecosystem services (Pascal et al., 2018a). Identified beneficiaries include dive shops benefitting from coral reef aesthetics and/or the presence of emblematic species such as groupers or sharks (Rudd and Tupper, 2002; Clua and Pascal, 2014; Berríos, 2017). Other beneficiaries include coastal real estate owners benefitting from beach protection ecosystem service (Barbier et al., 2008; Van Zanten et al., 2014; Pascal et al., 2016, 2018a). According to Lau (2013) all the elements are present for PES to be tested in the marine environment and the marine community is "keen" for further guidance.

While the potential exists, there are still few marine examples of PES (Bladon et al., 2016).

- (iii) Biodiversity offset credits (habitat banking): this exploratory approach, not implemented yet, corresponds to payments received by the MPA from project developers in compensation for planned damage caused by development projects which impact coral reefs and other marine habitats (Bovarnick et al., 2010; Quétiér and Lavorel, 2011; François and Pascal, 2012; Groot et al., 2013; Pioch, 2015). The MPA would be the guarantor that the habitats are protected and could offer habitat "credits". Permitted impacts are generally associated with beach nourishment, channel dredging, tourism infrastructure (hotels), private docks and piers, private and commercial ports or marinas, laying energy and communication cables, pipelines, and coastal protection projects. Most countries have developed a legal framework based on the mitigation hierarchy to ensure that impacts of a project will be, as far as possible avoided and/or reduced, following Environmental Impacts Assessments (EIAs) (UNEP, 2002). The remaining impacts must be compensated in the form of restoration, rehabilitation, replacement, and/or acquisition to the equivalent of the habitat impacted. The legal systems strongly favor on-site compensation, using the same type of ecosystem as was impacted. Nonetheless, although simple in principle, on-site compensation policies for coral reefs are difficult to implement due in particular to the complexity of the ecosystems, potential large number and heterogeneity of impacts, and costs and delays in assessing damage (Dunford et al., 2004; François and Pascal, 2012; Bayraktarov et al., 2016). Using off-site mitigation such as mitigation banking of the habitats managed by the MPA is proposed as an option, however, its medium-term/long term potential requires exploration.

4.3.1.2. Limited scalability. MPAs can fall into three categories (Font et al., 2004): (i) sites already with high levels of visitation, (ii) sites with high tourism potential but low level of visitations and (iii) sites with low tourism potential, or be subject to restricted access for conservation reasons. Most examples of protected areas that have raised significant funds from tourism (Pascal et al., 2018a) are sites falling in the first category with well-established infrastructure and tourism markets; a context not relevant to many protected areas. The potential to upscale this model where MPA generates most or all revenues through user fees therefore seems limited to these MPA sites. The NGO Blue finance, working on replicating the DR model during the last 2 years, has identified a very limited number of MPA sites in the Caribbean, SE Asia and Western Africa. with potential to become financially sustainable (Pers. Comm.).

Even with this limitation, the solution is attractive for governments allowing them "to focus public resources on the protected areas that cannot be self-financing but are critical to achieving the system's biodiversity objectives" (Saporiti, 2006). In the same way, MPAs can join together to pool their resources to market the tourism they offer, and through this to try to direct tourism to lesser visited sites and away from areas that are already saturated by visitors (Font et al., 2004). Some heavily visited MPAs may also allocate a portion of the revenues generated by tourism to assist conservation projects at less visited sites;

For MPA sites without tangible revenue potential, other solutions with impact investors are being explored such as: (i) Impact bonds as a solution to finance MPAs with non-existent but potential revenues. Impact bonds will allow more patient capital for the MPAs and the obligation to pay-back will be transferred to a third party (Landreau, 2014; Meyers et al., 2020); (ii) Project finance approaches similar to the ones developed in Costa Rica and Brazil (Linden et al., 2012; WWF, 2015). A long-term sinking fund with up-front capital is set-up to bridge the funding gap for protected area operations while government ready themselves for the full responsibility. This has led the governments to

exploring a variety of sources of funding, including new funding from domestic sources and payments for ecosystem services schemes; (iii) Endowment funds: In the Caribbean, the Caribbean Biodiversity Fund initiative has created a group of National Conservation Trust Funds (NCTFs) anchored by a US\$75 million endowment fund to develop reliable, long-term funding for conservation and sustainable development. To access the endowment resources, the NCTFs have to implement with the private sector innovative conservation finance solutions for the Protected Areas; (iv) Debt swap: governments can sell part of their debt to impact capital funds and redirect a portion of the debt payments to fund an independent endowment trust fund for marine conservation initiatives (UNDP, 2018).

4.4. Basing revenues on tourism

Another important limitation of the model is to link MPA revenues with tourism, a potential source of negative impacts on the coral reef ecosystems. Tools to control tourism sustainability involve defining the limits of ecological and sociological changes that could cause some degradation, but that would still be allowed on site (Stankey et al., 1985). Other tools such as Mc Elroy's tourism penetration index (TPI) and Butler's tourist destination product life cycle are useful in analysing the sustainability of current tourism levels (Butler, 1980; McElroy, 2003). These tools take into consideration factors such as the number of days the average visitor stays and the amount of environmental resources used by guests. The TPI is designed for destinations, as an indicator that they may be going over carrying capacity. Different guidelines have been produced for the design of the tourism management plans (Bacci, 1998; Wearing and Archer, 2001) recommending quality over quantity, diversification of the tourism products and community involvement through education and training.

These different tools can inform the conservation goals of the MPA and should be integrated into their management plans (IUCN and UNEP, 2014)

4.5. Entrepreneurial MPAs

Technical assistance is often necessary for entities that are expanding from traditional non-profit programs to innovative for-profit models (O'Donohoe et al., 2010). Providing technical assistance alongside investment, reduces the risk of providing capital to "inexperienced investees" (Dent et al., 2017). Building local capacity for improved management and financial sustainability in social and conservation entrepreneurship is a key aspect to successfully introducing a business approach to MPA management. Mechanisms can be smoothly implemented when capacity building takes place at the operational level. Park managers, agency managers, local administration officers and NGO employees, local communities and individual fishers need to clearly understand the mechanism and its contractual approach (Lapeyre and Laurans, 2016). Without such understanding, resentment and conflicts can emerge.

4.6. Adequate enforcement

Inadequate enforcement can result in poor performance of the MPA's marine conservation objectives and this aspect shall be taken into account when designing the mechanism. In DR, legislation allows the government to appoint officers from the collaborative management entity with various powers of searching and detaining for marine infractions. This situation is however exceptional situation (Orbach, 2010; Edgar et al., 2014). In most cases, "hard enforcement" is the exclusive domain of the regulatory agencies. The staff of the collaborative management entity will therefore limit its activities to "soft enforcement" (e.g. patrolling without detention) or improving self-enforcement from fishers and other stakeholders via regular education and attractive financial incentives. In some countries where capacities of regulatory

agencies are limited, this situation can create a risk to the environmental performance of the MPA.

4.7. Financial under performance

Financial under performance in the DR project (and potentially payment default) might occur from insufficient incomes to cover an elevated operational expenses. Even if a de-risking partner is providing loan guarantees to mitigate credit risk, main project risks must be examined in advance during a thorough due diligence process, using a risk matrix and third-party consulting firms. In the same way, internal controls and support functions must be implemented in the collaborative management entity during the lifetime of the agreement.

One major downside of this model is the vulnerability of tourism revenue to fluctuations in demand and exchange rates (Saporiti, 2006). The tourism sector is sensitive to natural disasters, economic recession and political turmoil (Thompson et al., 2014). However, several authors (Hall, 2001; Biggs et al., 2012; Dahles and Susilowati, 2015) agree that the sector has a great resiliency to these fluctuations and loss in revenues are limited in most cases. The vulnerability to fluctuations in the MPA revenues linked to the tourism market can be mitigated also by developing a basket of complementary revenue sources, creating financial reserves, and developing specific insurance products.

Financial modelling must use very conservative projections. Models in DR were based on observed and estimated annual number of users in the MPA (e.g. diver, snorkeler, day-tour passenger) less a 25% to reflect potential market risks.

Grouping different projects might also improve risk-return profile and helps isolate risk by separating project-related liabilities, tax and regulations from core business.

4.8. Performance indicators

Unbiased monitoring, reporting, and benchmarking of results must be reported to impact investors and donors (Brest and Born, 2013; Tuan, 2014). Appropriate performance indicators must be identified with stakeholder input to complement standard metrics (such as GIIN's Impact Reporting and Investing Standards) to capture MPAs' key social and environmental impacts. For example, indicators of ecological impacts in a coral reef ecosystem may include enhanced live coral cover and water quality. Indicators of the socioeconomic impacts of an MPA might include local employment in tourism businesses and improved fishery productivity. There are numerous intermediaries including development finance institutions, consultants, and non-profit organizations (Bugg-Levine and Emerson, 2011) developing performance indicators for impact investors.

4.9. Blended finance facility

Following Convergence (2018) blended finance archetypes, the proposed facility has incorporated elements of: (a) public or philanthropic investors providing funds to provide an additional layer of protection to private investors, (b) public or philanthropic investors providing credit enhancement through guarantees, (c) public or philanthropic investors providing grant-funded technical assistance facility to strengthen commercial viability and developmental impact. The blended approach in DR has allowed for improved quality in the design and execution of the collaborative management and investment solution, strengthened environmental and social positive impacts and reduced risks.

Blended finance and impact investment solutions as well as collaborative management approach for MPAs, as illustrated in DR are complex and involve a high number of steps and agreements. This kind of innovative mechanism has been coined a type of art form by several experts of the industry (Bos et al., 2015; Althelia Ecosphere, 2016). However, art is generally very expensive and such tools are no exception

(Lapeyre and Laurans, 2016). In DR, the transaction costs for the investment readiness have been covered by external technical assistance grants. If this solution aims to be scaled-up worldwide, the investment arm would have to be completed with a technical assistance facility providing grants from philanthropic institutions and DFIs. The facility would then provide technical assistance during the pre- and post-investment phases to meet the investor's criteria. Following a classic learning curve evolution, the technical assistance cost per project would then decrease progressively with the replication of the approach in multiple sites.

Nonetheless, even with reduced transaction costs, it is important to keep in mind that the reduced size of this kind of investments in MPAs is fine for "boutique" impact funds but large asset owners such as pension funds needs much larger investment opportunities.

5. Conclusion

An innovative solution for the sustainable use of 800,000 ha of marine resources in the Dominican Republic through the effective management and sustainable financing of the "Arrecifes del Sureste" MPA has been presented. The approach relies on a collaborative management agreement with tangible revenue model, leveraged by impact investment and empowering local communities. The solution offers interesting outcomes. It introduces entrepreneurial approaches for the financial sustainability of Protected Areas. It presents an innovative approach that uses catalytic and development finance to mobilise commercial impact finance into Marine Protected Areas (MPAs). From a government point of view, the approach provides empirical evidence of how non-public funding and collaborative management can become part of the financing options for a country's MPA network reducing the financial burden on public budgets.

Such a solution is not suitable for all MPAs and precautions must be taken in the early stages to balance conservation and revenue needs. The scheme is limited to MPAs with tangible business models and a

specific range of "boutique" impact investors who are able to deal with the size of projects and transaction costs. Due to this, at least in the short term, scalability of the approach is limited. In spite of this, the increasing number of collaborative management agreements for MPAs, the future development of entrepreneurial capacities in the world of marine conservation ('the social entrepreneurs') as well as the diversification of revenue sources will create promising prospects for the upscaling of the approach.

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.

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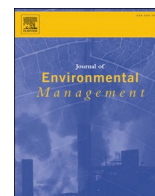
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Chapter 6: Coral reef restoration for coastal protection: crafting technical and financial solutions



Coral reef restoration for coastal protection: Crafting technical and financial solutions

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ABSTRACT

Coastal erosion, aggravated by coral reef mortality is a major issue for Small Island Developing States. Traditionally gray infrastructure, financed by public budgets has been used to combat beach loss. We examined if three Nature-based Solutions (Nbs): (i) coral restoration (green) (ii) restoration + limestone (hybrid) and (iii) restoration + 3D printed concrete (hybrid) could deliver positive outcomes for coastal protection and further incentivize cost sharing for reef conservation, with private beneficiaries. We modelled the impact of restoration on wave attenuation at two reefs off Barbados and simulated up-front and maintenance costs over a 25-year period. All solutions provide additionality when compared to gray infrastructure, especially in mitigating against Sea Level Rise. Restoration was the least costly with the highest risk of failure. The hybrid solutions, were less risky than the green as they provided immediate wave attenuation, alongside complementary services such as increased attractiveness due to the presence of reef fish. Their costs were however between +80% and +450% higher than gray solutions. While this might initially deter the use of NBS, blended finance and in some cases, Payments for Ecosystem Services, could provide options for governments and private beneficiaries to share costs, with ultimately greater benefits for themselves and coral reefs.

1. Introduction

Coral reefs are among the most productive and biodiverse ecosystems on the planet, providing a range of Ecosystem Services (ES) on which humanity depends. Unfortunately, reef health is declining on a global scale and increasingly, these ES are also decreasing (Eddy et al., 2021).

Coral reefs play an important role in protecting beaches from erosion by attenuating wave energy (van Zanten et al., 2014; Eliff et al., 2019; Zhao et al., 2019) with estimates of up to 97% wave dissipation being recorded (Ferrario et al., 2014). Additionally, these complex, three dimensional structures, and their calcareous algae, generate sediments that contribute to and maintain beaches (Harney et al., 2000).

Coastal erosion from high energy waves is a particularly onerous issue for Small Island Developing States (SIDS) with their small land-masses and reliance on coastal tourism (Cashman and Nagdee, 2017). Beach loss, flooding and reduced appeal of landscape aesthetics, are

some impacts, which can prompt major investments aimed at mitigation (World Bank, 2016; Rangel-Buitrago et al., 2018).

The two primary, reef related requirements for coastal protection are wave attenuation and growth, therefore actions aimed at improving the coastal protection ecosystem service should be effective at improving both (Brathwaite et al., 2021). Wave attenuation occurs primarily due to structure, as waves break on the reef framework (Monismith et al., 2015; Harris et al., 2018). Reef growth however, requires living corals to accrete, and produce positive carbonate budgets (Guannel et al., 2016; Perry et al., 2018; Ryan et al., 2019). Naturally derived coastal protection, therefore relies on healthy reefs (Sheppard et al., 2005; Quataert et al., 2015; Zhao et al., 2019) and stable populations of reef building corals, particularly the branching *Acropora* spp. and massive *Orbicella* spp. (Bruno et al., 2019; Estrada-Saldívar et al., 2019).

With the deterioration of coral reefs that provide natural defenses (De'ath et al., 2012; Jackson et al., 2014) and the concomitant increase in beach erosion, traditionally, gray infrastructure (e.g. breakwaters, sea

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walls, bulkheads) has been widely used for coastal protection (Liquete et al., 2013; Silver et al., 2019). These structures can be designed to bespoke parameters, aimed at delivering specific results, such as measurable impacts on wave attenuation and risk reduction, immediately. Typically, gray infrastructure is designed to withstand low frequency-high energy situations (e.g. storm surge) while improving wave attenuation for low energy, daily erosion events.

Gray infrastructure is effective when well designed, however many are: (i) expensive to build; (ii) rigid and unable to adapt to changing conditions such as increasing sea levels; (iii) unable to self-maintain and require costly maintenance; (iv) alter the natural character of a site in terms of both character and ecological processes and (v) might not be as conducive to the provision of other ecosystem services of coral reefs such as fisheries habitat and tourism (Fabian et al., 2013; Sutton-Grier et al., 2015). In addition, such infrastructure does little to improve the health of existing coral reefs, which originally provided the service.

There may however be a way of enhancing the ability of natural reefs to attenuate wave energy, using Nature based Solutions (NbS) such as reef restoration strategies. This type of green infrastructure is aimed at adaptively addressing challenges via protection and/or restoration of ecosystems thereby allowing their benefits to flow (Cohen-Shacham et al., 2016). With an overall goal of re-establishing sexually reproducing, genetically diverse populations, that can sustain themselves (Baums et al., 2019) reef restoration can assist in helping to maintain or recover key coral reef ecosystem services (Hein et al., 2020a, 2020b). With predominantly passive management (e.g. Integrated Coastal Zone Management) failing to significantly improve reef health, the addition of active measures, such as restoration has been suggested as a means of assisting the flow of ecosystem services (Ferrario et al., 2014; Roelvink, 2019).

The ability of green infrastructure to grow is especially important with the impending threat of Sea Level Rise (SLR). Sea levels exert major controls on coral growth, as the organisms attempt to maintain their position in the photic zone, by producing calcium carbonate (Dullo, 2005). Rising seas and major reductions of carbonate producers can result in corals “giving up” and drowning (Schlager, 2003). The IPCC Special Report on Global Warming predicts with high certainty, that a temperature increase of 1.5 °C could result in a global mean sea level rise of 0.43 m by 2100 (relative to 1986–2005), causing up to 90% loss of coral (Nauels et al., 2019). While the natural reef framework can be substituted by other rigid structures such as stone, which is often used in the construction of breakwaters (CIRIA, 2007), their ability to continue to provide the coastal protection service will depend on vertical reef accretion (Beetham et al., 2017; Perry et al., 2018).

Reef restoration is a fairly recent intervention, however there is a wide variety of methods described and a large body of peer reviewed literature that details best practices including (Fabian et al., 2013; Bayraktarov et al., 2016, 2019; Rinkevich, 2019). In this study, Boström-Einarsson et al. (2020) was used as our seminal document, as it is the most recent and extensive review reef restoration to date.

In addition to technical considerations, the financing of any intervention is a major consideration. Investment in NbS ought to at least triple in real terms by 2030 if the world is to meet its climate change, biodiversity and land degradation targets (United Nations Environment Programme, 2021). While public funding for NbS is essential, finding ways to unlock private sector financing for marine conservation would reduce the burden on governments (Salamon, 2014) and help close the conservation gap (Bos et al., 2015). Further, such mechanisms incorporate the private sector – that benefits from the service – in improving ecosystem health.

The west coast of the Caribbean Island of Barbados was chosen as the Pilot Site for investigation, due to its history of coastal erosion coupled with strong government interest in implementing interventions (Downstream Strategies, 2015; IHCantabria, 2019). Coastal erosion has been an enduring issue for Barbados, since at least the 1980's, leading to the formation of the Coastal Zone Management Unit (CZMU) in 1982.

Under this Unit and via an International Development Bank (IDB) loan, the Government of Barbados (GOB) spent over USD\$ 20 million to provide shoreline stability via breakwaters and beach nourishment in the Coastal Infrastructure Project between 2002 and 2010. This investment provided a reduction in coastal erosion and additional benefits from aesthetics, recreation and tourism have also been achieved (Banerjee et al., 2018). The work however has not, nor was it designed to improve the health of the coral reefs that originally provided this service. At the same time, coral health in Barbados severely declined due to a combination of local and global factors that include increasing levels of anthropogenic nutrients, unsustainable harvesting of reef fish and warming seas (Irvine et al., 2021).

Barbados is currently in an IMF restructuring programme with goals of reducing government spending, especially in cases where public services provide benefits for private entities. The recommendation has further been made, that the GOB consider “innovative” means of providing incentives for the private sector that would encourage them to take part in environmental activities (Schuhmann, 2019).

This paper therefore seeks to provide such incentive by strengthening the case for NbS as alternatives to fully built (gray) infrastructure for coastal protection and suggesting mechanisms with the private sector to finance the interventions.

In order to meet our objective, we will:

- Identify fringing reefs on the west coast of Barbados that are providing the coastal protection service
- Demonstrate the ability of restored reefs to increase wave attenuation under storm surge and Sea Level Rise (SLR) at these sites
- Discuss possible NbS restoration interventions for coastal protection
- Discuss cost implications and potential financing mechanisms for NbS interventions

2. Descriptions

2.1. Site characterisation – Barbados

Barbados, with a coastline of almost 97 km, is almost entirely surrounded by coral reefs (Fig. 1). The coastal area represents one of the largest economic assets, with approximately 95% of tourism activities (Corral et al., 2016) and more than 70% of hotels (Cashman and Nagdee, 2017). Barbados is a tourism “hub” with this industry contributing 13% directly and 40% indirectly to the GDP (Kemp-Benedict et al., 2020). Tourism relies heavily on the allure of wide beaches (Dharmaratne and Brathwaite, 1998), so the quality of beaches, and by extension coral reefs are of primary significance to this industry. Investments which target increasing the size of beaches are increasingly important (Banerjee et al., 2018).

Technical agencies in Barbados house extensive data sets on coastal processes, as a means to better manage coastal erosion, which has affected the tourism product. The west and south coasts are already heavily engineered, primarily at the expense of the GOB. The west coast has however seen the onset of a potential change in financing, with select west coast hoteliers contributing to funding for coastal engineering preliminary work in 2010. There is therefore a precedent that can build upon to provide a different funding mechanism for shoreline protection projects.

The west coast of Barbados is home to some of the island's best-developed fringing reefs, which typically extend from headlands, slope gently to 10 m depths and extend approximately 300 m from the beach (Stearn et al., 1977). Reef health is measured by a variety of indicators, with hard coral cover being primarily key to the provision of coastal protection (Graham and Nash, 2013; Harris et al., 2018) and so will be used in this paper as the indication of health. Fossil records indicate that *Acropora* spp. were once dominant on fringing reefs, regionally (Macintyre et al., 2007) and in Barbados, primarily colonized reef crests and inner swash zones (Lewis, 1984). A net decrease in hard coral cover on



Fig. 1. Coral reefs around Barbados. Red dot in inset is the island of Barbados (Source CZMU). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

west coast fringing reefs between 2007 and 2017 has been recorded (CERMES, 2018).

Beaches along the west coast of the island are typically coralline or mollusc in origin, narrow (35m max width and steeper in gradient than south east and east coast beaches (55m max width) (Baird, 2016a, 2016b). Average annual erosion rate is between 0.3 and 0.4 m/yr (Baird, 2015a, 2015b) and 20–50% loss of sand supply rates is predicted, if the least optimistic IPCC scenario (RPC8.5) is reached in the absence of mitigation (Baird, 2016a, 2016b).

2.2. Restoration design for coastal protection

From the range of restoration methods presented in the Bostrum et al. (2020) review we pre-selected (i) a green solution - coral gardening and (ii) two hybrid gray-green solutions - artificial reefs with cultivated corals from gardening (hereafter called artificial reefs). These are reported as the more extensively utilized and tested methods, with the greatest records of success at 66%. Coral gardening and artificial reefs comprise 48% and 21% respectively of the restoration studies found globally. The sexual propagation method of larval enhancement for example, while promising, has been used in only 6% of the studies. Direct transplantation, in addition, was discarded due to its requirement of harvesting corals from existing natural colonies (Boström-Einarsson et al., 2020).

Project goals for restoration are (i) reef growth, indicated by positive carbonate budgets (Perry et al., 2013) and (ii) increased wave attenuation (in either short or long terms) indicated by shallow depths. Shallow reef depths do not necessarily result in increased wave attenuation and the actual depth for wave breaking, depends on different factors,

including wave height (Scott et al., 2021). However as much of the critical wave breaking on Barbados west coast reefs in the past occurred in 1m or less (A. Rowe, pers comms, August 8th 2021), this is used as our restoration goal (Fig. 2).

2.2.1. Coral gardening (green solution – Nbs1)

Coral gardening is at the foundation of any intervention due to our requirement for growth (Bowden-Kerby, 2001; Shaish et al., 2008; Rinkevich, 2019). It is two-step process with an initial nursery phase where coral recruits reach acceptable sizes, in preparation for the second step in which they are out-planted onto a reef. The core methods for coral gardening are the same, but a variety of techniques exist for growing and out-planting. In the absence of peer-reviewed literature assessing and comparing coral propagation methods in the eastern Caribbean, the method used by Seascape Caribbean (Jamaica) was investigated. This, to our knowledge, is the only eastern Caribbean operation in which corals are commercially grown and out-planted to re-populate reefs for coastal protection (Iberostar Group, 2020). It involves an in-situ nursery framework supporting a proprietary coral-to-nursery and coral-to-substrate combined device. Via this device, the coral nubbin remains in the nursery only for the period required to heal and to maximize its metabolic resources (head-starting), of usually 100days or less (Ross, 2016). This allows rapid throughput and minimal in-nursery investment per planted coral (A. Ross April 10th 2021 pers. comm.).

We chose to focus on *Acropora palmata* as our species of interest due to its: (i) importance as a structural builder on reefs (Graham and Nash, 2013); existing extensive use in restoration (Boström-Einarsson et al., 2020; Hein et al., 2020a, 2020b); fast growth and reproductive strategies adapted to the shallow, high energy environment in which they will be out-planted (Boulon et al., 2005; Roelfsema et al., 2018) and history as wave dampeners on Barbados reefs (Oxenford et al., 2021).

2.2.2. *y*-green hybrid – artificial reef – limestone (Nbs2), Natrx™ (Nbs3) + coral gardening

The hybrid consists of artificial reefs populated with coral outplants from gardening. As underwater structures that mimic natural reefs physically, and ecologically (Baine, 2001; Goreau and Trench, 2012), they can be used to structurally and functionally restore degraded reefs (Reguero et al., 2018). Further, while decreased wave energy and reef growth are primary goals, the enhancement of the substratum also allows for complementary secondary goals to be achieved, such as improved biodiversity habitat and aesthetics for tourism (Hein et al., 2020a, 2020b).

Wave breaking and friction occur as a result of the artificial reef structure itself, while its ability to maintain distance with the surface and keep pace with expected changes in SLR is dependent on coral growth (Perry et al., 2018); which itself relies on a variety of ecological parameters. Incorporating corals from gardening is therefore a key component of our hybrid solution, and the ability to attract natural coral recruits, highly desirable (Yanovski and Abelson, 2019). Concrete and stone have consistently been identified as appropriate for both wave attenuation and natural coral recruitment (Lukens and Selberg, 2004; Creed and De Paula, 2007; Burt et al., 2009; Foley et al., 2014) and are already in use in Barbados as breakwater material.

We propose for the hybrids (i) limestone as it is a natural material for coral recruitment and is found locally and (ii) concrete in the form of 3D printed concrete modules, with a patented design by Natrx™. They possess large surface areas (for natural recruitment) and internal void spaces to facilitate growth of organisms (Natrx, 2021).

2.3. Dimensions/design

In a real-life scenario, final designs will be made only after extensive modelling and examination of oceanographic conditions such as waves and water levels. The proposals suggested are at a conceptual level only,

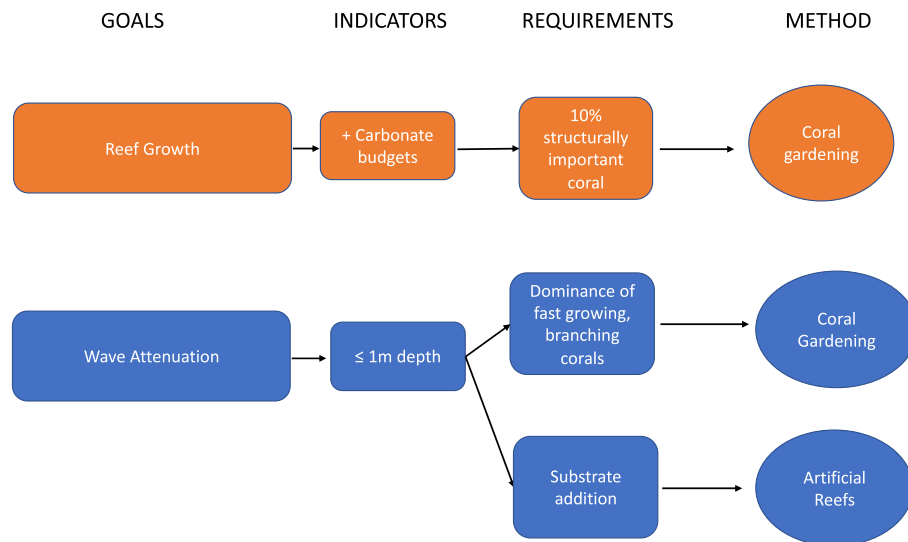


Fig. 2. Restoration project goals, indicators of success, contributing factors and methods for implementation.

and consist of one layer of stone on bare reef substrate.

Dimensions of 30m long and 12m width, equivalent to a 360 sq.m area were based on past breakwater construction work for similar projects in Barbados. These dimensions also fall within those modelled by Roelvink et al. (2021) (Section 3.2). Structures proposed were 1m in height, to be placed in 1.5m depth.

Acropora palmata is reported to grow on average 9–10 cm per year in the Caribbean (Gladfelter, 1984; Shinn et al., 1989; Dullo, 2005). We therefore used a ratio of 1 fragment transplanted per sq.m. for coral gardening, in order to reduce the chances of crowding.

The same dimensions were used for the green solution, as the authors were unable to locate data or guidelines on the reef area dimensions required for replanting in order to achieve similar wave attenuation as artificial breakwaters.

3. Methods

The protective functions of specified reefs were first demonstrated, then areas where restoration would be effective, identified, and the 3 NbS interventions then costed.

3.1. Identification of fringing reefs providing coastal protection

The following steps were taken to identify beaches where reefs could be credited with providing significant protective functions:

1. Google Earth™ maps were used to identify sandy beaches, fronted by natural fringing coral reefs (slope/crest, spur and groove).
2. The Shoreline Change Study (Baird, 2015a, 2015b), a coastal classification system based on 30 years of erosional trends was used to identify sections of these sandy coastlines categorized as Stable Coast (Natural), Accreting Coast (Natural) or Dynamically Stable Coast (Natural).
3. Aerial photos from 2015 in the Shoreline Change Study (Baird, 2015a, 2015b) were used to identify stable/accreting beaches, with convex morphology. This shape is indicative of beaches that are not sheltered from incoming waves, due to being within or part of a bay
4. Beaches that fit the criteria were identified as being protected by reefs that front them.

3.2. Ability of restored reefs to increase wave attenuation under storm surge and sea level rise (SLR) scenarios

3.2.1. Restoration location

The ability of coral restoration to attenuate wave energy depends strongly on the reef profile and location of the work. Roelvink (2019, 2021) determined where restoration would be most effective at reducing wave run up by modelling 4 different wave profiles, which were derived from over 30,000 reef shapes. The reef profile of Fitts Village corresponds to Roelvink's Convex Reef Profile which is described as a gentle transition between the reef flat and forereef. Sandy Lane corresponds to the Typical Fringing Reef Profile, where there is a defined reef flat with a relatively steep forereef slope (Roelvink, 2019).

We pre-selected 3 potential restoration locations and on each of our reef profiles, on reef flats and crests (Fig. 3). Restoration at such shallow areas has been identified as effective at attenuating wave energy, in the literature (Ferrario et al., 2014; van Zanten et al., 2014; Quataert et al., 2015; Elliff and Silva, 2017) and via past modelling at 2 other west coast reefs in Barbados (Baird, 2017). We then visually compared these locations to those of Roelvink, in order to determine their expected effectiveness at reducing wave energy. The green boxes in Fig. 3 show the locations chosen for further investigation, based on effectiveness and cost.

3.2.2. Impact of restoration

In order to confirm the effectiveness of our chosen restoration location, XBeach non-hydrostatic (XBNH), was used to assess the impacts of coral reef restoration on wave attenuation and the effect of the water level at the shoreline. XBNH is an open-source, process-based, morphodynamic model (Roelvink, 2019), which was applied in one-dimensional mode for both cross-shore transects. The model simulates 2 hours of waves, and the results are used to calculate wave run up and wave heights.

Simulations were carried out for storm surge under existing conditions (Runs 1–3) and in a future with predicted SLR (Runs 4–6). The impact of living coral was determined by friction coefficients. A friction factor of 0.1 represents reefs with essentially no live coral and 75–100% algal turf; 0.16 represents reefs with 25–50% live coral; and 0.2 represents reefs with 75–100% live coral (van Zanten et al., 2014). Wave heights, periods and directions remained the same. The three friction values combined with the two water levels, formed the six different model runs (Table 1). The offshore wave condition used for all model runs was set to replicate a common storm for this area (Baird, 2016a,

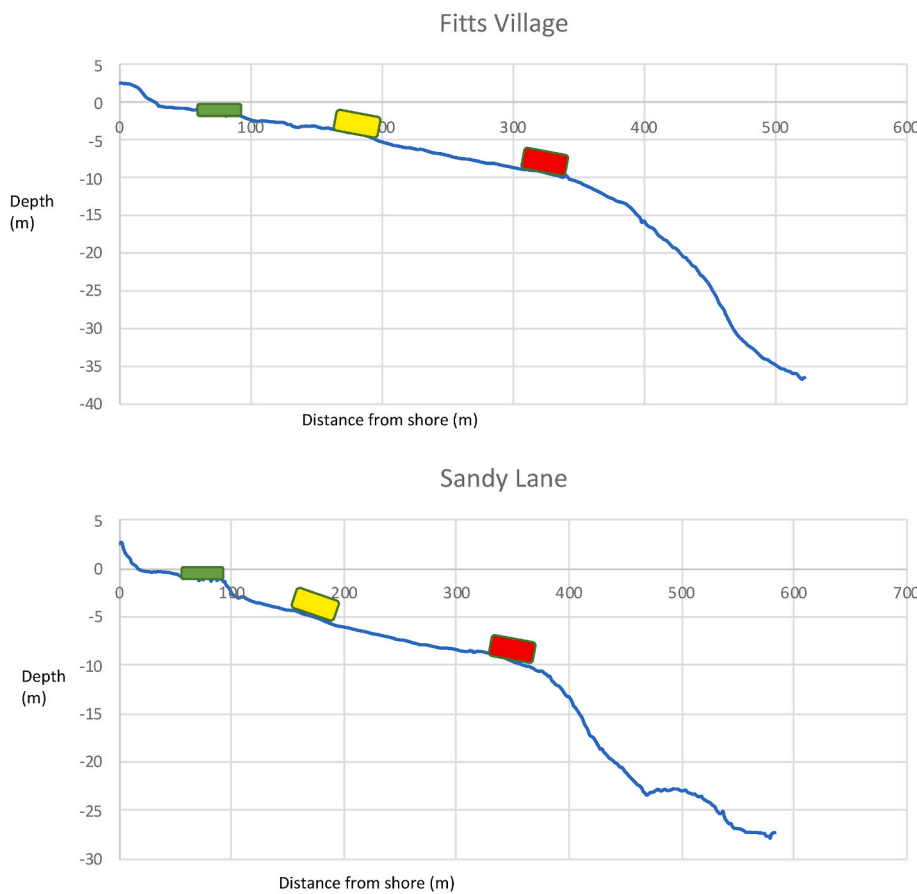


Fig. 3. Cross sectional reef profiles of Fitts Village and Sandy Lane, with a selection of potential restoration locations, represented as boxes. Green boxes represent the proposed options due to their effectiveness and lower costs; yellow boxes represent effective but expensive options and red boxes represent areas where restoration should not be carried out as it could result in greater wave run up at the shoreline. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1

Summary of XBeach input parameters. Runs 1–3 are of existing water levels with increasing friction and runs 4–6 are of increased water levels due to SLR, with increasing friction.

Input Parameters	Present Scenario			SLR Scenario		
	Run 1	Run 2	Run 3	Run 4	Run 5	Run 6
Water level (m + MSL)	0.65	0.65	0.65	0.95	0.95	0.95
Friction (–)	0.1	0.16	0.2	0.1	0.16	0.2
Wave Height (m)	0.7	0.7	0.7	0.7	0.7	0.7
Wave Period (s)	10	10	10	10	10	10
Wave Direction (°N)	345	345	345	345	345	345

2016b). To obtain a direct comparison between the model runs, the exact offshore waves were replicated for all runs.

The model results were then processed to compare the hydrodynamic response of the storm waves over the initial profile to the restored profile (with a reef augmented structure).

The parameters below were assessed:

Wave heights

- Significant wave height at 100 m offshore (Hs 100) - the average height of the highest 1/3 of waves at 100 m offshore.
- Significant wave height at 50 m offshore (Hs 50) - the average height of the highest 1/3 of waves at 50 m offshore.

The reduction in wave heights between 100m and 50m indicates wave transformation moving from offshore to nearshore. The transformation shows the effects of reef structure (including restoration), friction and water level (among others).

3.3. Setup at the shoreline (Setup)

- the increase in mean water level due to breaking waves. This indicates the effect of reef structure on increasing water level set up, which would typically result in greater wave run up and beach erosion. Our aim is to find structures that will reduce set up.

3.4. Two percent exceedance wave runup (R2%)

- wave run up is the maximum water-level elevation, with respect to mean sea level, measured on the foreshore. If wave run up is greater than beach crest elevation, flooding poses a threat to land. A common measure of wave runup in engineering is the 2% exceedance value (R2%)

3.5. Swash (IG & HF)

- generally defined as the time-varying location where the ocean meets the beach. It can be broken down into the infragravity (IG) and incident frequency bands (HF), which provide additional information about the processes responsible for the water level at the shoreline. (Stockdon et al., 2006).

The percent difference between restored and unrestored reef was calculated via the following equation:

$$PercentDifference = \frac{X_{Restoration} - X_{Initial}}{X_{Initial}} \times 100\% \tag{1}$$

4. Cost analysis

We considered the full stream of costs for coral gardening and

artificial reef solutions, as well as the time-value of money for solutions covering an area of 360 sq m.

(i) for coral gardening, average costs were calculated for growth and outplanting of 360 fragments of *Acropora palmata* (1 fragment per sq m). A survival rate of 66% of the fragments after 2 years of transplant as reported by Bostrom et al. (2020) was used in the calculations. Cost categories include: a. Design & Permits - basic design, ecological surveys and permit applications; b. Material costs (e.g. frames, anchors, drills) c. Construction costs - staff costs (marine biologist, divers), vessel and scuba equipment rental and insurance; d. Annual maintenance and monitoring costs expected during the following 25 years, including staff, transport, replacement materials. An additional 10% of total costs was added as unexpected costs.

(ii) for artificial reefs & gardening, average costs included: a. Coral gardening as calculated in (i); b. Design & Permits for artificial reefs of same dimensions (12m width x 30 m long) - engineering design, ecological surveys and permit applications. Engineering designs have fixed minimum costs usually sized for larger projects. Following industry standards, we have limited design costs to 10% of the total initial cost of the project to avoid over-weighting this cost category; c. Material costs - quarried limestone and production of the Natrx™ modules (based on direct quotation from the manufacturer); d. Construction costs - including placement, environmental monitoring and engineering observations; e. Annual maintenance and monitoring costs expected during the ensuing 25 years - staff, transport and replacement of select stones after major climate events. Maintenance costs were estimated at 10% of the initial material costs

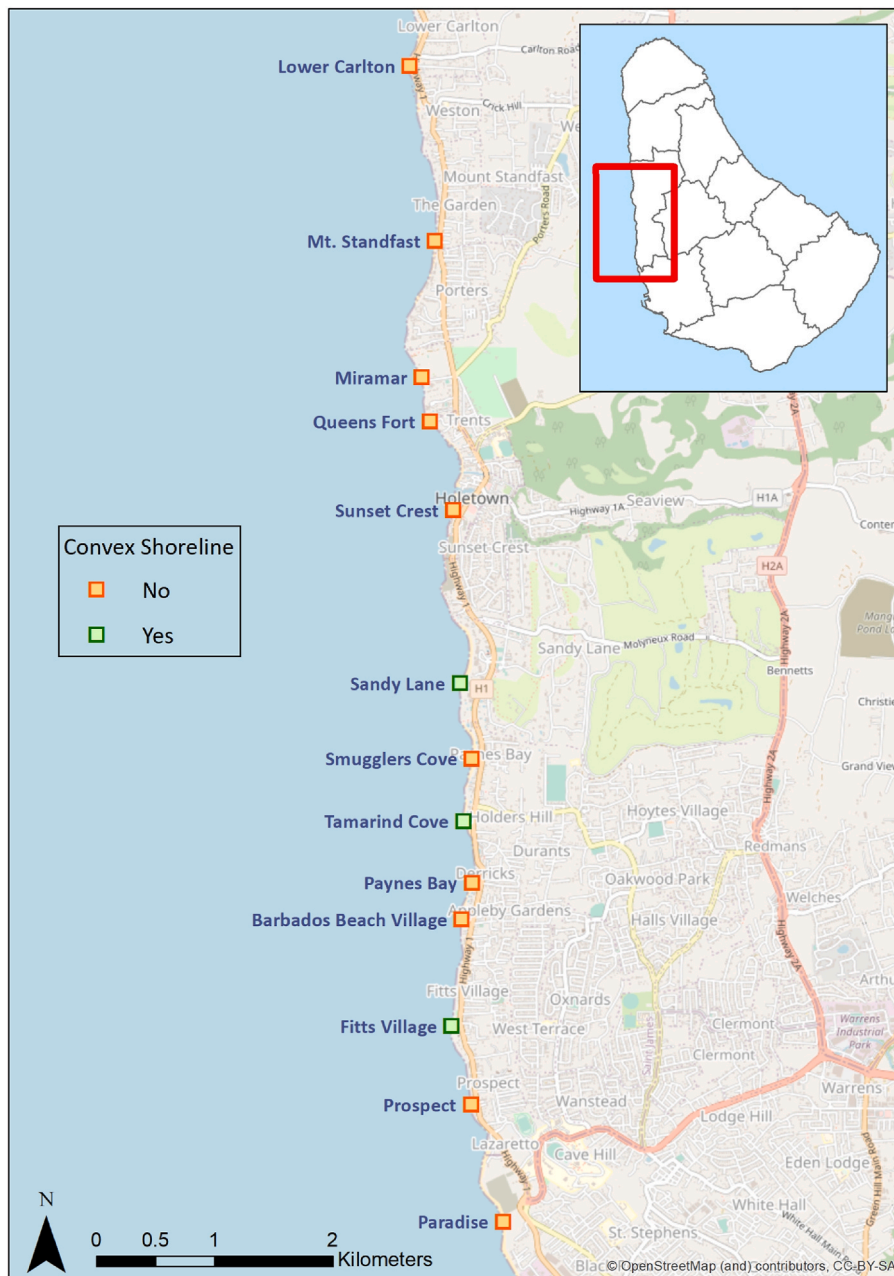


Fig. 4. Fringing reefs fronting beaches going from north to south on the west coast of Barbados, with their status and shoreline shape. Sites with green boxes met the criteria for reefs with protective functions. Sites with orange boxes did not meet the criteria for protective functions. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

and were annualized evenly over 25 years. Average costs were assessed through expert opinion and review of gray literature.

Costs were calculated on discounted values for a 25y period from the start of the project. This range of 25 years was proposed in similar studies to reflect the ecological responses of ecosystems to the tested scenario (Balmford et al., 2008; Wielgus et al., 2008). A discount rate of 5% was applied in computing present values (Ehrlich, 2008).

4.1. Evaluation for investment potential

In order to discuss the investment potential of the 3 different NbS, we examined: costs, maintenance requirements (over 25 years), timescales (for delivery of coastal protection service), potential negative environmental impacts to the coral reef, and risk of project failure. A gray solution (limestone breakwater) was also presented for comparison.

There are a variety of private sector applicants that would benefit from coastal protection, however, for many SIDS, with the importance of tourism and its strong relationship to coastal protection, one of the most obvious beneficiaries is coastal hoteliers, who desire beach presence and width as a selling point for their businesses. Visitors to Barbados for example, were reported as less inclined to return (for the same price) if the beaches were narrower (Uyarra et al., 2005).

5. Results

5.1. Identification of reefs with protective functions

Thirteen beaches were classified as stable on the west coast of Barbados (Fig. 4). Of these, Sandy Lane, Tamarind Cove and Fitts Village also had convex shorelines and therefore met our criteria for reefs with protective functions. Tamarind Cove and Sandy Lane reefs are of similar size. The decision was taken to focus on Sandy Lane, as this hotel has already explored alternative methods for protecting their shoreline, and therefore in a real life scenario, would be expected to be more amenable to cost sharing.

Both project sites, are characterised by sandy beaches fronted by

fringing reefs, with similar profiles (Fig. 5).

Site beach and reef specifications are summarized in Table 2. The beaches of Fitts Village and Sandy Lane are characteristic of the west coast, varying between 10 and 44m in width. Fringing reefs of 250m (Sandy Lane) and 400m (Fitts Village) width are located between 80 and 100 m from shore, in approximately 2 m of water. Between 2007 and 2017, a statistically significant decline in hard coral abundance was reported at Sandy Lane. At Fitts Village, hard coral abundance fluctuated within the decade, however these differences were not significant (CERMES, 2018). Hard coral cover at Fitts Village (10.1%) in 2017, just meets our reef growth requirement of 10% hard coral cover, with Sandy Lane (6.6%) being slightly below.

5.2. Ability of restored reefs to increase wave attenuation under storm surge and sea level rise (SLR) scenarios

5.2.1. Restoration location

Outputs (Fig. 6) indicate that the locations chosen for restoration at both sites are favourable for improving coastal protection for both current, and expected future increased water levels. Increasing friction, by increasing coral cover for example, (runs 1–3, 4–6), results in decreasing wave runup and wave heights for all the parameters measured. At Sandy Lane for example, Hs wave heights 50m offshore (bottom right graph) decline from 0.57m at friction value 0.1–0.49m at a friction value of 2. The outputs also demonstrate that increasing water levels, (runs 1 vs 4, 2 vs 5 and 3 vs 6) result in wave run up and nearshore wave heights also increasing, thus increasing the probability of coastal erosion. Swash is not included in the outputs, as the other parameters indicate the situation adequately.

5.2.2. Impact of restoration

The percentage difference between restored and unrestored reefs for the 6 runs is shown in Figs. 7 and 8. The bars indicate the percent difference with and without restoration for the parameters shown on the Y axis. The colours indicate positive (green bars) or negative (red bars) impact of restoration.

Almost all of the bars are green, indicating an overall positive benefit

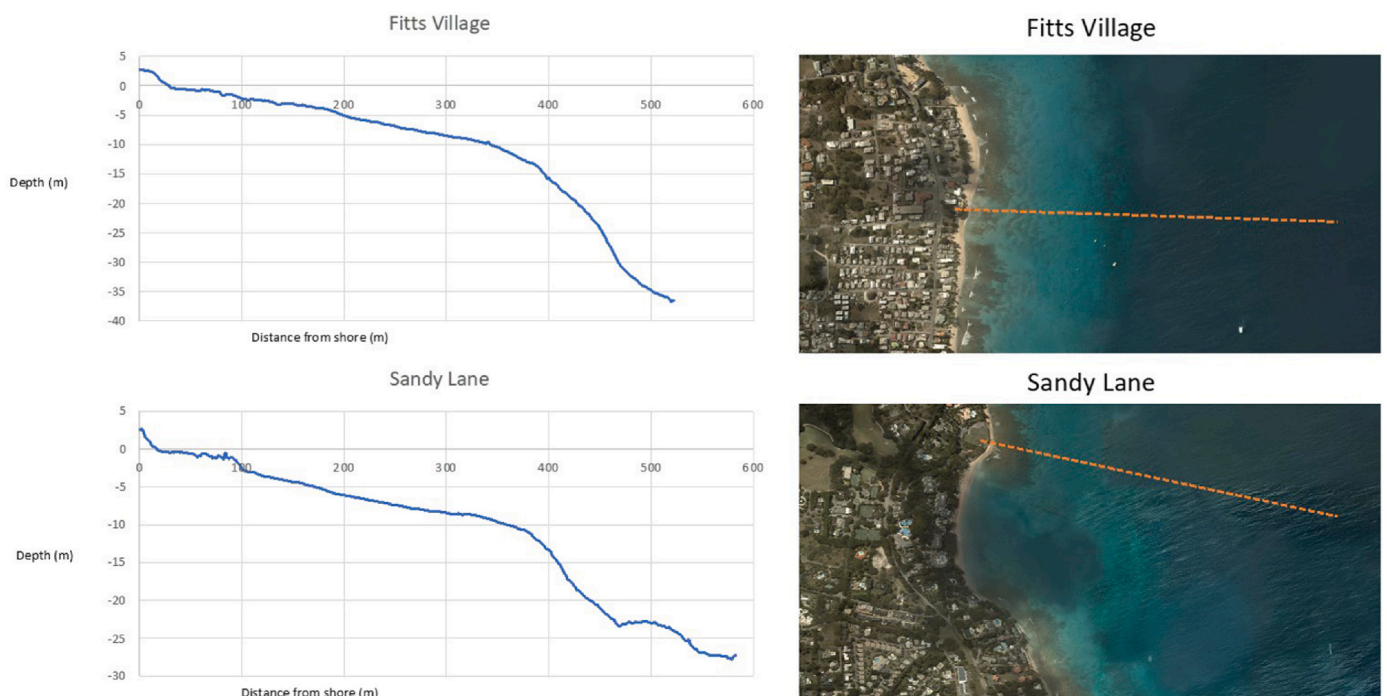


Fig. 5. Aerial photographs of Sandy Lane Bay and Fitts Village fringing reefs and their corresponding cross section profiles. Source CZMU, 2015 - Coastal Zone Management Unit, Coastal Risk Assessment and Management Program, LiDAR Topographic Surveys, 2015

Table 2

Beach characteristics and reef health at Fitts Village and Sandy Lane, Barbados (Sources: CRMP Sediment Transport Study Final Report. June 2016; CRMP – Shoreline Change Study Phase 3 Report. October 2015; CERMES, 2018).

		Fitts Village	Sandy Lane
Reef Health	Year	Mean % hard coral cover	
	2007	8.5	15.2
	2012	6.3	7.5
	2017	10.1	6.6
Beach	Characteristic	Description	
	Length (m)	780	505
	Slope	Flat back beach, steep slope (0.16)	Continuous downward slope from back beach to water (0.13)
	Width (m)	33–44	10 to 20
	Erosion trend	Stable Natural	Stable Natural
	Erosion rate	Net Stable	Net Stable

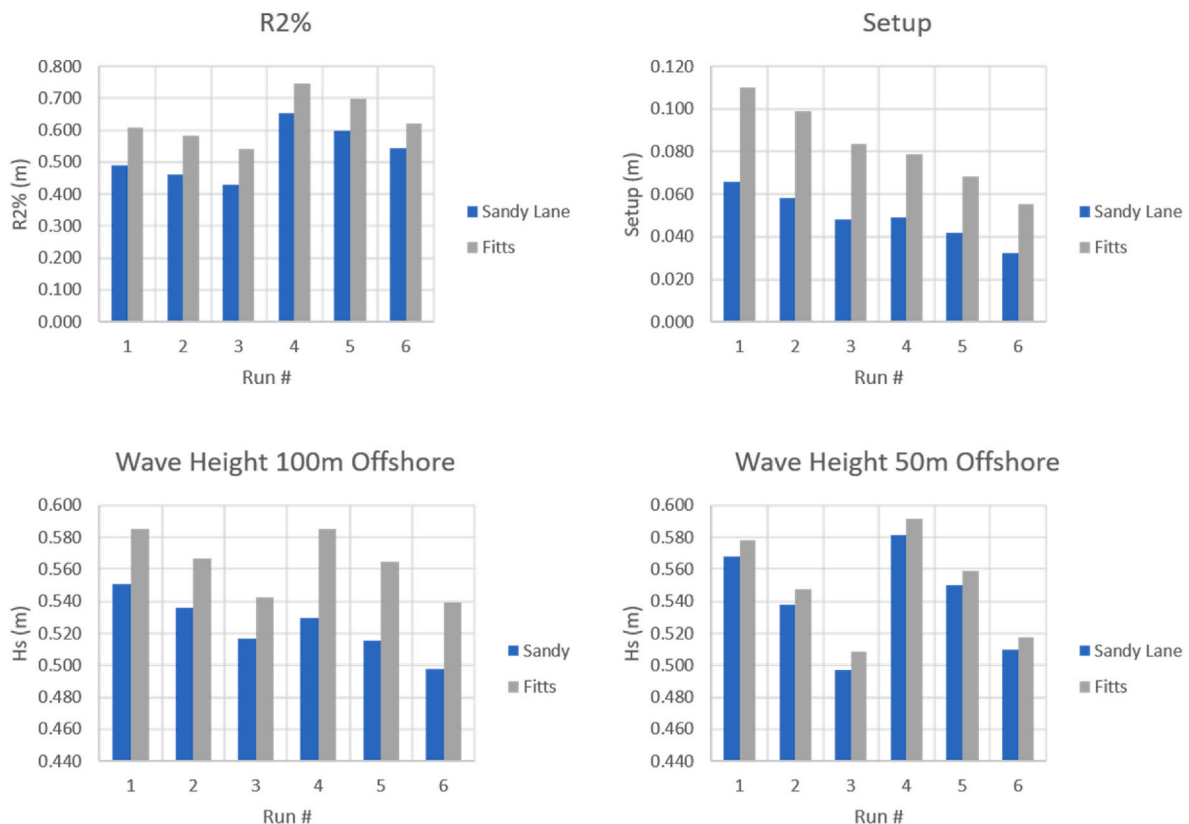


Fig. 6. Outputs of XBeach modelling at Fitts Village and Sandy Lane reefs from Runs 1–6 on: R2% - Two percent exceedance wave runup– indicates flooding potential; Setup – increase in mean sea levels at the shoreline; Wave height 100m offshore – average height of highest 1/3 of waves at 100 m offshore; Wave height 50m offshore – average height of highest 1/3 of waves at 50 m offshore. Runs 1–3 represent current water levels with increasing friction. Runs 4–6 indicate increased water levels due to future SLR.

of restoration. Red bars are seen for Setup at Sandy Lane and Hs 50 for Fitts Village. This might be due to location of the restoration and reduced water depth, causing wave asymmetry and the wave height increasing before it breaks. The main point for our analysis however, is that the R2% value is always reduced, which means less wave energy reaches the shore.

5.3. Costing of NbS

The standardized costs expected for each solution are summarized in Table 3.

There is a 6-fold difference in cost between the hybrid Natrx™ (highest) and the green coral gardening (lowest), with the limestone structures (hybrid and gray) falling in between these costs. Incorporating the coral gardening component increases total costs of artificial reef solutions by approximately 80% (limestone) and 17% (Natrx™).

When compared to fully gray solutions, the “gray-green” hybrids solutions have an additional cost between +80% and +450%.

5.4. NbS assessment

The NbS and gray infrastructure all have the ability to improve coastal protection with varying technical, ecological and cost-related strengths. NbS, however allow for growth and therefore confer additionality in terms of maintenance and the ability to keep up with SLR over gray solutions (Table 4).

When compared to gray infrastructure, the coral gardening solution has the lowest cost, highest risk and longest time to effectiveness but with positive ecological impacts. The limestone hybrid represents medium cost and risk, providing long term resilience benefits with an increase (+80%) in costs when compared to standalone gray limestone solutions. The Natrx™ hybrid represents the most costly solution, with

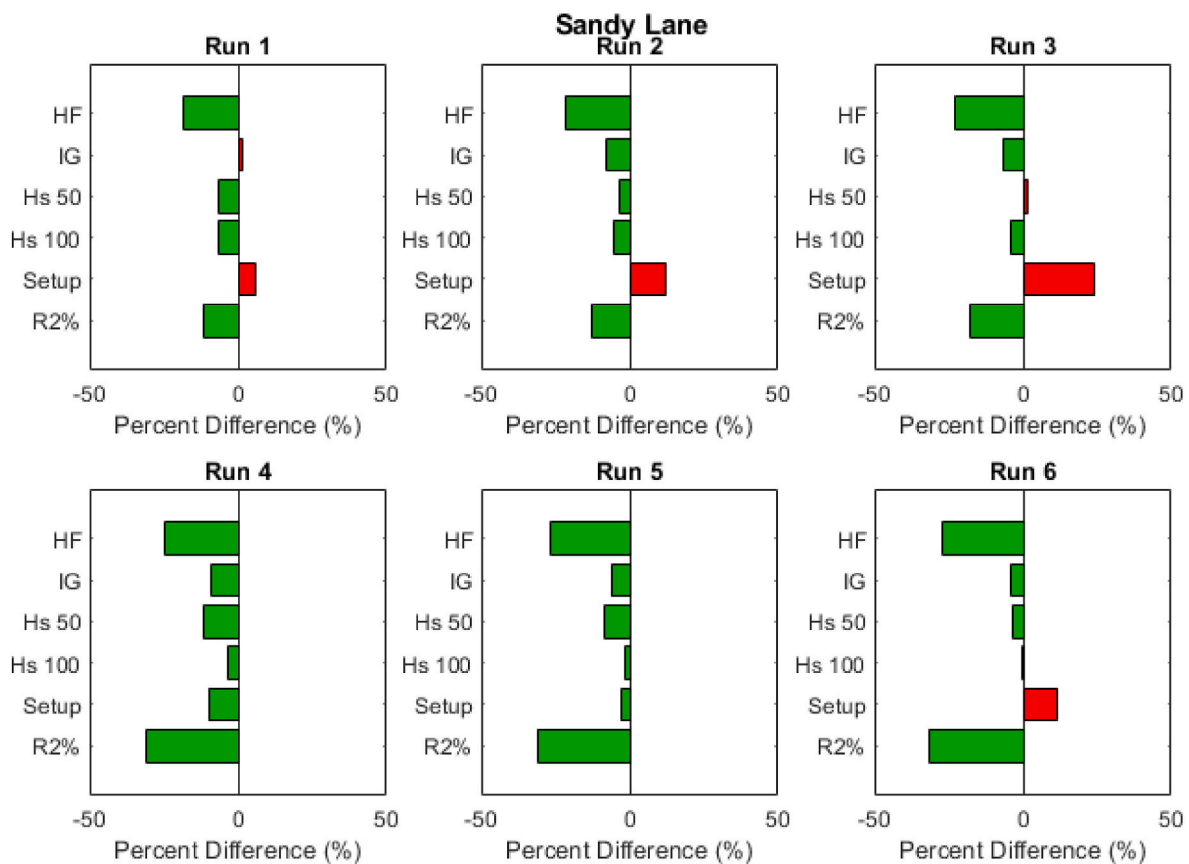


Fig. 7. Graphs demonstrating the percentage difference in the different parameters, between reefs with and without restoration at Sandy Lane. HF – Incident frequency bands; IG – Infragravity bands; Hs 50 – average height of highest 1/3 of waves at 50 m offshore; Hs 100 – average height of highest 1/3 of waves at 100 m offshore; Setup – increase in mean sea levels at the shoreline; R2% – Two percent exceedance wave runup – indicates flooding potential. Green represents that restoration has a positive impact (less wave energy reaching the shore) and red represents a negative impact (greater wave energy reaching the shore). Runs 1–3 represent current water levels with increasing friction. Runs 4–6 indicate increased water levels due to future SLR. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the lowest risk and high ecological benefits. There is a +200% increase in cost when compared to the limestone hybrid and a +450% cost increase when compared to a limestone breakwater.

6. Discussion

Coral reef health is deteriorating and their ability to protect coasts is subsequently declining. We confirmed that some reefs, if restored, can regain their ability to attenuate waves and examined three NbS, as alternatives to gray infrastructure to do so. We determined that the NbS solutions can contribute to coastal protection, provide additional resilience benefits and attract financing that is not accessible for solely gray infrastructure. Depending on the solution chosen, however, and when compared to the gray solution, the NbS can be an initial source of cost reduction, or increase. These findings provide important information for both ecologists, interested in restoration as a means of aiding coastal protection and financiers who are interested in investing in technologies to improve coastal protection.

6.1. Reefs with protective functions

We identified reefs on the west coast of Barbados that provide a coastal protection service. All reefs do not protect beaches and it is important to know which ones have this capacity, especially in cases (such as this) where investment is sought for the specific output of increasing or maintaining coastal protection services and not simply general coral reef conservation. We showed that the fringing reefs at

Fitts Village and Sandy Lane act as wave attenuators and further that restoring them in specific areas, could improve on this service even under storm surge and projected SLR. Currently the Barbados government does not allow construction on the reefs (Baird, 2017), which is sensible if there is live coral to destroy. In cases where reefs are already primarily dead, placement of gray-green infrastructure (hybrids) could increase health of the existing reef, as well as improve its wave attenuating ability. Increasing hard coral cover on the reef, especially our target species *Acropora palmata*, has been shown to greatly improve the recovery potential of reefs (Ortiz et al., 2021) and is therefore beneficial to the reef as a whole, in addition to coastal protection.

6.2. Restoration interventions

While wave dissipation will occur almost immediately with physical contact (on the reef or artificial structure), positive ecological outcomes will take place in the medium to long term timeframes (5–10 years) (Hein et al., 2020a, 2020b) due to the period of time required for coral growth and recovery of ecosystems. *Acropora* spp. for example, one of our key species and one of the fastest corals in the Caribbean, grows at around 10 cm for year, while by comparison, other important reef builders, such *Orbicella* spp. have growth rates of 5–10 mm per year (Dullo, 2005). Coral mortality of outplants is one risk of a fully green solution. Bostrum et al. (2020) for example report that while survival rates of transplanted corals were relatively high at 66%, more than half (60%) of the projects monitored outplants for less than eighteen months. Given the aforementioned length of time it takes corals to grow, and the

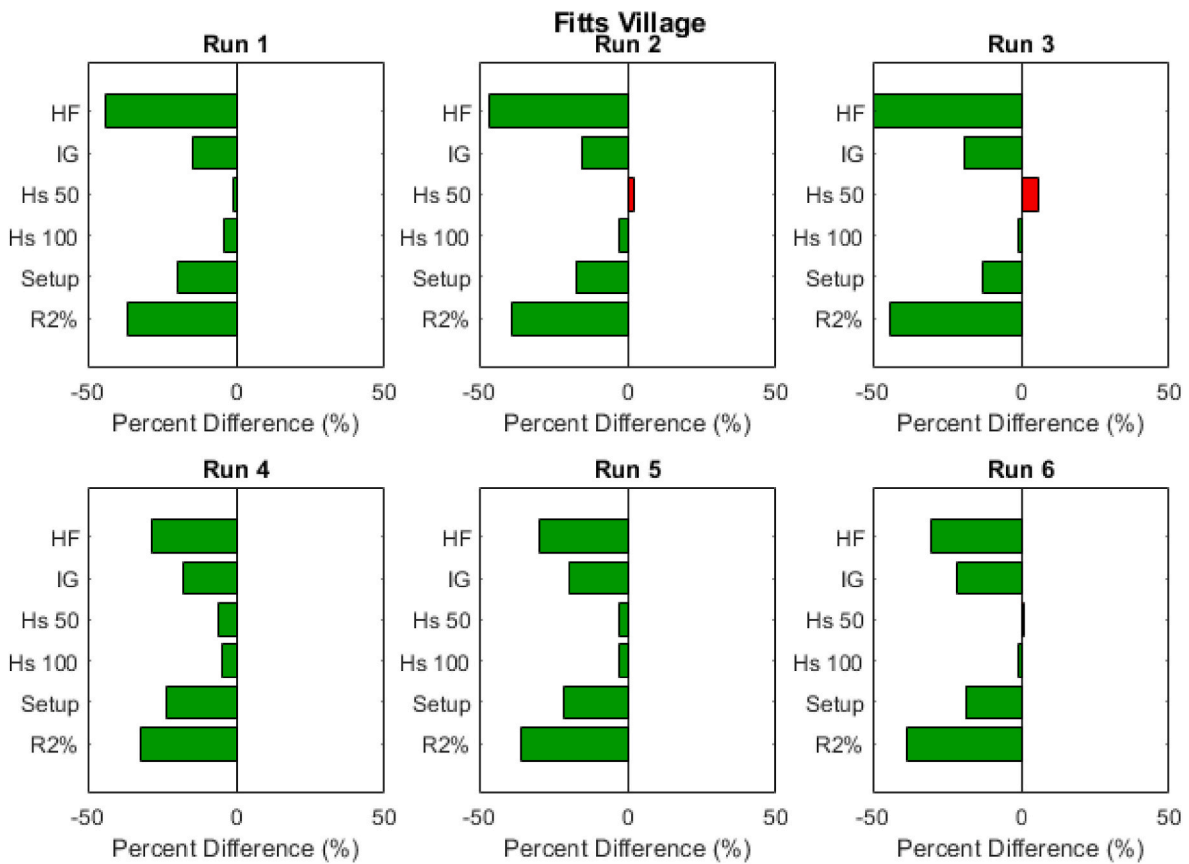


Fig. 8. Graphs demonstrating the percentage difference in the different parameters, between reefs with and without restoration at Fitts Village. HF – Incident frequency bands; IG – Infragravity bands; Hs 50 – average height of highest 1/3 of waves at 50 m offshore; Hs 100 - average height of highest 1/3 of waves at 100 m offshore; Setup – increase in mean sea levels at the shoreline; R2% - Two percent exceedance wave runup– indicates flooding potential. Green represents that restoration has a positive impact (less wave energy reaching the shore) and red represents a negative impact (greater wave energy reaching the shore). Runs 1–3 represent current water levels with increasing friction. Runs 4–6 indicate increased water levels due to future SLR. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 3
Costs of green (NbS1), hybrid (NbS2&3) and gray infrastructure.

	US\$ per sq.m						US\$	
	Design & permits	Material	Construction	Total initial project cost	Monitoring cost 25y Present Value	Maintenance cost 25y Present Value	Total project cost 25y Present Value	Total project cost 25y Present Value
NBS 1 Coral Gardening	2	18	26	46	34	57	138	49,500
NBS 2- Limestone + coral gardening	18	133	38	188	54	68	310	111,600
NBS 3 - Natrx + coral gardening	18	748	52	818	54	72	944	339,700
Gray solution - Limestone breakwater	16	115	11	142	19	11	173	62,100

range of stressors already impacting corals, higher levels of mortality are likely over time. Relying solely on coral growth for coastal protection is very risky. However, in cases where the situation is not urgent, in that erosion is not imminent (as at Fitts Village and Sandy Lane) and the hotelier is prepared to assume the risks involved, coral gardening as a stand-alone solution would be suitable.

The hybrid solution is a way of “bridging the gap” between a fully green solution with its long timeframe and risk of failure and a fully gray solution, which is unable to adapt to changing conditions and grow without further intervention. Such eco-engineering strategies have been reported as maximizing storm protection benefits (Gedan et al., 2011); with the greatest chances of success in terms of wave attenuation

(Fabian et al., 2013) as well as providing an opportunity for improvements to reef health (Foley et al., 2014). They also operate at different temporal scales which might also improve outcomes (Rinkevich, 2019). There are many types of artificial reefs that can be utilized for coastal protection (Zepeda-Centeno et al., 2018) and each will require assessment and modelling at each location to ensure its efficacy. Artificial structures used in the hybrids, can also benefit tourism, as divers find their structural complexity attractive (Polak and Shashar, 2013) and are willing to pay for such experiences (Kirkbride-Smith et al., 2016). These solutions, however, in the Barbados case, are substantially higher in cost than gray infrastructure and attractive financing mechanisms will be required for the private sector to consider NbS as alternatives.

Table 4
Evaluation of NbS and Gray Infrastructure in terms of their ability to attenuate waves, ecological impacts, time, risks and costs.

Technical Solution	Green Coral Gardening	Hybrid Artificial Reefs + Coral Gardening		Gray
		Limestone + Coral Gardening	Natrx + Coral Gardening	Limestone breakwater
Functional performance (Ability to attenuate wave energy)	Medium	High	High	High
Reef Growth (Ability to maintain + ve budgets)	High (but conditional on a variety of factors)			Low (relies solely on natural coral recruitment)
Ecological Impact during construction	Low - minimal coastal and marine disruption	High - marine and coastal disruption for placement of material	Medium-low - minimal coastal & marine disruption	High impacts - marine and coastal disruption for placement of material
Time to effectiveness	10 years minimum	Immediate - 5 y minimum for full effectiveness	Immediate - 5 y minimum for full effectiveness	Immediate
Risks of project failure	High risk of complete failure - hurricanes, bleaching, disease events, pollution from land, etc can result in 100% coral mortality, post transplant stress (mortality rate >30%)	Medium risk of complete failure - even with 100% coral mortality, the limestone will still (i) effectively act as a breakwater and reduce wave energy (ii) provide substrate for coral recruitment thus conferring the growth/accretion benefits over time.	Medium risk of complete failure - even with 100% coral mortality, the concrete will still (i) effectively act as a breakwater and reduce wave energy (ii) provide substrate for coral recruitment thus conferring the growth/accretion benefits over time.	Low risk to functional performance (once adequately designed)
Natural Habitat Restoration	Medium - addition of hard coral to the existing reef	High - addition of hard coral and additional substrate to the existing reef	High - addition of hard coral and additional substrate to the existing reef; improved colonization of marine fauna due to void spaces	None
Costs	\$49,500	\$111,600	\$339,700	\$62,100
Costs per sq.m	\$138	\$310	\$944	\$173

6.3. Financing mechanism

There is a role for public, philanthropic and private monies in coral reef conservation finance. The coral reef is a “public good” and ultimately its health is the responsibility of governments, whose funding will always be required but is inadequate (Bos et al., 2015). Grants and philanthropy partially fill the conservation gap and can also be used to leverage public and private investment (Pascal et al., 2021). Delegating some financial responsibility to the private sector is now deemed essential in the quest for achieving sound coral reef conservation (Iyer et al., 2018).

At the level of a single hotel, the costs of coral gardening are below those of the hybrids or gray infrastructure. Given the scant (but encouraging) evidence to support the long-term efficacy of coral gardening techniques and the high risk involved, we suggest catalytic capital for the fully green, coral gardening solution. Grants for example are well suited to this action, as they can be used to implement coral restoration and with effective monitoring, can assist in providing the evidence required for scaling up such a solution. Until there is robust data to support the impact of coral restoration in coastal protection, such catalytic capital is recommended.

Payment for Ecosystem Services (PES) could also be used for coral gardening, once carried out within the confines of a Marine Protected Area (MPA). In this case, hoteliers (Beneficiaries) can make payments to MPA management entities (Providers), for specific conditions such as increasing and/or maintaining the abundance of *Acropora palmata* at 10%, or for a toolbox of conservation action such as fishery effort control and water quality improvement. Broad conservation action reduces the risk of failure to reach conditions (Brathwaite et al., 2021). Further, bundling payments for coastal protection with other ecosystems services such as fisheries and aesthetics, could allow for payments to be made for a range of services, without the complexity of trying to tease out one service from another (Lau, 2013).

Both Limestone and Natrx™ hybrids are more robust than either coral gardening or the breakwater option on its own. The costs are higher, but the risk of project failure is greatly reduced. Blended finance (which combines impact, catalytic and concessional) is recommended.

Impact Investment could allow hotels access to new sources of financing, with more generous requirements than traditional commercial banking. Catalytic funders could provide technical and scientific assistance to the projects to: increase positive impacts, reduce engineering and environmental risk and, provide funding for equipment dedicated to natural resource management activities. Blending impact and catalytic with grant capital is a way of making the investments even more feasible, by reducing the overall amount to be borrowed. Risk is lowered and profitability enhanced thereby being more attractive for private sector participation (Thiele et al., 2020).

There is available funding from the Impact Investment world. At the close of 2018, the industry was worth an estimated US\$502 billion (Core Characteristics of Impact Investing) with investments concentrated on energy (15%), micro-finance (13%) and other financial services (11%). A small but growing proportion of impact investments focus on environmental impacts. However, conservation projects, particularly around marine ecosystems, are not attracting impact capital at the same pace as the rest of the impact investment market, largely due to a shortage of investment-ready projects and the development of future opportunities (Pascal et al., 2021).

Investment in NbS such as coastal restoration projects however respond to several criteria of impact investment: ecological benefits are clearly identified with critical habitat restoration and likely fish biomass recovery; social benefits can be linked to outsourcing of maintenance to local community workers, as well as awareness campaigns with schools and adults in the restoration areas and financially, the operation is bankable with the hotel being the investee and paying-back the financial obligations from its business operations. While the size of individual NbS might be too small for impact investment funds, aggregating a pipeline of projects allows for reduced expense for each hotel and the minimum target to be reached.

Time and uncertainty are other important factors to consider. The time required for interventions to become effective is especially pertinent when involving the private sector (whose incomes depend on outputs) in cost sharing. Added to this, is that significant costs are paid up front, while benefits take a much longer time to manifest (Thiele et al., 2020). At this time, especially with the impact of Covid, the large

increase in cost for the hybrids compared to gray infrastructure could be a great dis-incentive, for hoteliers, even with accessible capital. One way to increase the attractiveness of the hybrid solutions in particular, is via decreasing insurance costs when NbS are used. While each case will differ, insurance premium reductions over a period of 5 years are reported as being able to pay 44% of restoration costs, specifically for hybrids, where risk reduction is assured (Reguero et al., 2020). Bearing in mind however that any improvements to reef health will benefit all, not only hotels, and is ultimately the responsibility of governments, we suggest that the government shares cost with the hoteliers for NbS.

6.4. Study limitations

Our outputs should be considered as illustrative, as we have shown the major steps required and justification for investing in NbS for coastal protection. We have not carried out a full feasibility exercise, which would encompass more intensive modelling and engineering work, nor have we carried out a cost-benefit analysis due to the differing time frames involved in providing the service.

We have proposed very preliminary designs, for the gray-green hybrid solutions, and the “devil is in the details”. Poorly designed and constructed artificial reefs will have similar negative impacts as poorly designed gray infrastructure. There is no attempt here to provide full engineering designs for artificial reefs, but simply identifying appropriate materials and preliminary dimensions such as height and crest width, which will require modelling and testing (with their inherent costs) to determine their efficacy. Our assumption here is that such a structure will be effectively designed with the engineers to provide the required wave attenuation outputs.

The complexity of historical fringing coral reefs cannot be fully replicated by artificial means, due to the diverse interactions between coral morphology and reef hydrodynamics. As a result, the exact mechanisms by which coral colonies attenuate wave energy remain unclear. At present, there are no models that can be referred to in order to determine the size of the area to be replanted, number, positioning, density etc. of each species of coral required, even though the impact of waves on reefs has been extensively measured and numerically modelled (Monismith et al., 2015; Eliff et al., 2019) (Hardy and Young, 1996; Scott et al., 2021). Information from physical models, such as, water flows through adjacent coral canopies, could inform restoration with regards to the quantity and placement of corals. However, while progress is being made, this has proven to be extremely complex (M. Tissier pers comms, May 4th 2021).

We have based our concept of health on live hard coral cover of structural builders. There are however a plethora of interactions, not considered here, that work synergistically to promote reef health. Corals exist in a balance with a variety of organisms, from crustose coralline algae gluing dead coral to form a stable framework and attracting coral larvae, to bioeroders who remove dead coral tissue providing space for new colonization and grazers who remove macroalgae facilitating growth (Dikou, 2010). Our solutions however, provide the building blocks for these processes to occur.

6.5. Future direction

Coral restoration and green engineering (the addition of artificial substrates, designed to mimic and be incorporated within natural systems) are still in their infancy compared to terrestrial ecosystems (Rinkevich, 2021). However exciting innovations such as new seeding approaches for sexual propagation (Chamberland et al., 2017) and accelerated evolution, enhanced tolerance and assisted gene flow (Bay et al., 2019) and eco designs (Mohammed, 2016; Pioch and Léocadie, 2017; Zepeda-Centeno et al., 2018; Hein et al., 2020a, 2020b) are promising signs for increasing success in the future.

The worlds of finance and conservation have not yet successfully merged due to challenges from both sides. These include: a

disconnection between finance and conservation planning, lack of expertise in marine conservation finance, limited baseline data on positive social and environmental impacts, operational risk, hesitancy of NGOs to enter into loan agreements and limited information on expected returns (Pascal et al., 2018). This gap can be bridged by awareness and engagements (on both sides) as well as interdisciplinary collaboration (Bos et al., 2015). The outputs of NbS projects, both successes and failures must be carefully documented and widely shared to increase the knowledge base.

7. Conclusion

In Barbados, the beach asset on the west coast is estimated to be losing BBD\$ 6–8 million annually from erosion linked to SLR (CZMU, 2020). Solutions, such as those suggested in this paper, are actively being sought to improve delivery of the coastal protections service and cost sharing with the private sector. The findings from this study can assist the private sector, in playing a greater role in financing NbS for coastal protection, by (i) providing a framework for determining the protective features of reefs (ii) demonstrating the effectiveness of green and gray-green solutions as alternatives to gray infrastructure and (iii) providing suggestions for mechanisms by which these efforts can be financed. Such information is especially important in SIDS such as Barbados, where flooding and erosion events are already increasing due to coastal storms (Wong et al., 2014). In this age of the Anthropocene, with unprecedented impacts on the coral reefs that buttress their livelihoods, hoteliers in particular, cannot afford to “sit on the sidelines” and must play an active role in conserving/protecting coral reefs. There are many variables that have to be controlled for coral reefs to be able provide this ecosystem service and so it will be important for governments and the private sector to work together on passive and active management interventions in a holistic fashion, in order to give coral reefs their best chance at delivering the coastal protection service. NbS provide significant benefits over time, compared to gray infrastructure, and these must be taken into consideration, even if the upfront costs prove to be higher for some solutions.

Credit author statement

Angelique Brathwaite: Conceptualisation, Writing – original draft & Review; **Ramon Roach:** Software, Visualisation, Writing – review & editing; **Eric Clua:** Project administration, Writing Review; **Nicolas Pascal:** Supervision, Writing Review.

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.

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Chapter 7: Private capital to improve tropical coastal protection: time for a boost

Commentary

Private Capital to Improve Nature-Based Solutions for Coastal Protection: Time for a Boost

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Abstract: Coastal protection, an important coral reef ecosystem service, is threatened by increasing coral mortality, exacerbated by global climate change. Nature-based solutions in the form of coral restoration, while not perfect, can assist in rebuilding reef structure and improving the flow of the service for some sites. With a financing gap existing between what is required for conservation and what is being accessed, private investors should be playing a larger role in such restoration activities. Especially so as coastal hoteliers in particular, benefit from stable beaches and also have additional income generating potential with healthy reefs. Blended finance solutions in particular, are especially suited to restoration that incorporates substrate addition, while payments for ecosystem services are more suited to coral gardening. Conservation and finance practitioners must engage further and understand each other's worlds, in order for these private sources to be effectively sourced and utilized.

Keywords: coastal protection; reef restoration; financing mechanisms; nature-based solutions



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1. Introduction

Coastal protection against beach erosion, is a complex and highly important service, especially for small island developing states (SIDS) with their small land masses, high coastal populations and dependence on coastal tourism [1–3]. This ability of coral reefs to attenuate wave energy, as well as to supply and trap sand, allows for reduced flooding and erosion of beaches, protecting coasts and saving lives [4–6]. Coral reefs, while not solely responsible for provision of this ecosystem service [7], can play major roles in providing it [8–10]. Healthy reefs with high coral cover are reported to result in greater wave attenuation and protection for beaches [7,11,12] and their efficacy is reduced with coral deterioration [13,14].

Coral health is declining on a global scale and impacting the ability of reefs to provide a range of ecosystem services [15–17], including coastal protection. There is an increasingly important need to protect and rebuild coral reefs, especially in a time of a changing climate, where this ecosystem is among the most vulnerable [18]. Protecting and rebuilding coral reefs simultaneously reduces risk from global climate change (GCC) impacts, while buttressing economies of coastal nations [19–21].

Solutions do exist for minimizing coastal erosion in the form of built or gray infrastructure. When well designed, breakwaters, for example, can work very efficiently at reducing beach erosion [22,23] however there is no added benefit to the reef that originally provided the service. In addition, such hard infrastructure cannot grow to maintain levels with expected sea level rise (SLR) [24,25]. It seems prudent, therefore, to take steps to improve the condition of coral reefs, as one means of aiding beach protection, and these management measures must be supported by strong investment [26].

2. The Need for Alternative Financing Sources for Conservation

Globally, while public expenditure on biodiversity averages \$USD6.1 billion per annum, only 4% of this is directed towards marine biodiversity [27]. Traditional sources, such as governments and philanthropy, are required for conservation but are inadequate [28]. New financing means must be found quickly given the small window of opportunity, reported to be less than 50 years, for both mitigation against stressors and adaptation to a changing climate [26]. With this timeframe, it is imperative that effective action is taken now. Involving the private sector, whose businesses rely on healthy reef systems for a variety of ecosystem services, makes sense. Coastal hoteliers in particular, should play a major role in financing interventions for coastal protection, as they are among the primary beneficiaries of coastal protection, with much to lose from eroding beaches.

For there to be an uptake in private investment for the ecosystem service of coastal protection provided by coral reefs, the service must first be quantified and economically valued, which will vary by site. Methods, both high [21,29] and low tech [30] exist for calculating value, often in terms of avoided damages and replacement costs [8]. Valuation can be used for a variety of reasons, and in this case, advocacy to convince stakeholders of the monetary significance of coastal protection and the need to improve reef health is valid [31]. Protecting beaches is expensive, with the United States of America spending USD\$1.8 billion on the effort [21], followed by Cuba's USD\$400 million spend [29] and over USD\$24 million spent by the Government of Barbados in tackling eroding beaches during 2002–2010 [32]. It is further projected to cost coastal cities (annually) around USD\$1 trillion, to avoid “unacceptable losses” by 2050 [33] and coastal insurers have been reported to have paid out (globally) more than USD\$300 billion in one decade [8].

3. Provision of Coastal Protection and Factors That Impact the Service

Coral reefs provide the coastal protection service by dissipating wave energy, which occurs first as waves break on the shallowest section of the reef [4,34,35] and is further enhanced by friction as the bottom of the wave moves along the sea floor. Then, in order for the service to be maintained, growth via the deposition of calcium carbonate is required to keep the appropriate distance with the sea surface.

SIDS are highly vulnerable to GCC, with SLR of particular importance to the provision of coastal protection [36]. Reef building corals must remain within the photic zone for vertical reef growth to occur and sea levels exert great control on this process, as corals will maintain their growth rates to keep up with rising seas [37]. However, rapidly increasing sea levels—especially when combined with large mortality events, for example from coral bleaching and disease—can result in reefs giving up and drowning [38]. The impact of SLR on coral reefs is site specific and heavily influenced by reef bathymetry and coral species [39]. However, the probability of sea levels rising by 1 m with an increase of 1.5 °C by 2100 is high. Without interventions, this could result in a 90% loss of coral reefs [40]. Beneath this threshold, reefs are not expected to keep up with rising seas, and many regional reefs already exhibit coral cover below this [41].

Coral reef ecosystems, under normal conditions, have the ability to recover naturally [42], however they are reported to require at least 10 years free from major disturbances for the recovery of short lived species and much longer for others [43–45]. Recovery is stymied by an almost continuous onslaught of impacts, exacerbated by lessening natural recruitment and reduced times between catastrophic events [46]. Under such conditions, natural recovery to pre-impact conditions is unlikely [47,48] and human intervention is required.

4. Active Management for Coastal Protection

Management should be aimed at achieving no net loss of biodiversity along the mitigation hierarchy: avoid, mitigate, restore, offset, compensate [49]. In many cases, avoidance of stressors is impossible. Mitigation, which is aimed at reducing human pressure, also known as passive management, has dominated marine conservation [50–52]. However,

these interventions (e.g., actions carried out by marine protected areas—enforcement etc.) have been largely unsuccessful [53,54], and while necessary, have failed to significantly improve reef health [55,56]. Many questions remain about the management of human pressures [57], however it seems evident that simply eliminating stressors is inadequate [58]. Restoration, the next tier in the hierarchy, is a form of active management, where humans intervene to directly assist in improving reef condition [59]. Incorporating active alongside passive management interventions has been recommended for increased chances of reef recovery [12,60,61].

Coral restoration is a nature-based solution (NbS) [62] that seeks to rebuild reef structure, which is key to restoring coastal defence [63,64]. Restoration can take a variety of forms as described in [65], and in order to protect beaches, they must allow for wave breaking, friction and coral growth (as described in Section 3). Here, we consider coral gardening as the core of our solution.

Coral gardening—which refers to the out planting of corals after an intermediate nursery phase—is a vital component of any restoration response to coastal protection, as it seeks to directly increase hard coral cover. Gardening can occur either on its own as a fully green solution, or in combination with artificial substrate, as a gray-green or hybrid solution. The green solution refers to out-planting reared corals directly onto a coral reef, while the hybrid entails out-planting corals onto artificial reefs [47].

In order for these interventions to be effective with respect to coastal protection, survival and growth rates of reared coral are particularly important. Corals must withstand the rigors of transplantation and remain alive until at least sexual maturity in order to ensure long-term resilience. In one of the most extensive reviews on coral restoration to date [47], an average survival rate of 66% for restored corals, from an assessment of 363 case studies in 56 countries, was found. Many of these studies were short-term, i.e., 12 months, with only five extending to over a decade. The few long-term monitoring efforts recorded, indicate either high survival—80% on average—or no difference in mortality between out-planted corals and those on control sites [66]. Restored corals must also grow high enough to break waves, and faster growing corals, such as *Acropora* spp.—with growth rates averaging 100–150 mm per year [37]—are already among the most common species used in restoration. Their suitability for the process is well documented [64]. They are also important as structural builders [67] and employ reproductive strategies adapted to the shallow, high-energy conditions in which they are often out-planted [68,69].

A case study carried out in Barbados [70] provides details of coral gardening and hybrid interventions as a means of promoting coastal protection (Table 1). In this study: i. reefs were assessed to determine if they provided coastal protection; ii. numerical models were used to demonstrate that restoring reef structure could improve the service; and iii. suitable restoration techniques were suggested and costed. Total costs were determined for an area of 360 sq m. The size of the restoration was based on breakwaters constructed under similar conditions in Barbados and fell within the parameters suggested as suitable for restoration for coastal protection with these reef shapes [71].

The solutions can be scaled up if required for Caribbean reefs using *Acropora* spp; however, restoration for coastal protection is not solely a matter of size and a larger area therefore is not necessarily better. The size of the area to be restored will be determined by the solution chosen and site characteristics.

Coral gardening can therefore be summarized as a low cost, high-risk solution, with a relatively long time frame to medium effectiveness, while the hybrid is a high cost, low-risk solution with a relatively short time frame to high effectiveness.

Restoration is a relatively new field, but we believe there is adequate scientific certainty that when properly carried out and within a holistic system of reef management, it can be effective at increasing hard coral cover [47,72], hence improving coastal protection. Further studies, such as [71], have demonstrated via numerical models, the size, shapes and locations on reefs, where restoration can be most effective at protecting beaches.

Table 1. Summary of performance, timeframes, risk and costs (\$USD) for coral gardening and hybrid solutions, over 360 sq m. from a case study carried out in Barbados.

	Green Coral Gardening	Gray-Green Hybrid Artificial Reef + Coral Gardening
Functional performance—ability to attenuate wave energy	Medium—the ability of restored corals to reach the required height for wave breaking is not assured. Even if they succeed in wave-breaking, they cannot replace the ecological complexities of a natural reef, that also aid in wave attenuation.	High—artificial structures can immediately provide the required shallowness for wave breaking. They also provide suitable substrates for natural coral recruitment and planting from gardening.
Time to effectiveness	Minimum 10 years—for corals to grow to suitable heights for wave breaking.	Immediate for wave attenuation Minimum 5 years for corals to colonise artificial substrate.
Risk of project failure	High—e.g. from hurricanes, bleaching, disease	Medium—even with full mortality, artificial structures will still (i) effectively act as breakwaters, reducing wave energy (ii) provide substrate for coral recruitment, thus conferring growth and accretion benefits over time.
Costs—growing & outplanting, design & permits, material, construction, maintenance (25 years)	\$50,000	\$112,000

There is still a risk, as methods for coastal protection have not been extensively tested in the field, and reared corals are also susceptible to a plethora of other stressors (e.g., storm damage, coral bleaching, disease and eutrophication) that could result in high levels of mortality, especially in a changing climate [57]. However, with little hope of impact reversal and ecosystem recovery in the short term [73], new methods, even with their element of risk should be employed. This risk factor will however have implications for private investment.

5. Funding Restoration of Coral Reefs for Coastal Protection

NbS are not yet being heavily financed. Out of a cumulative investment of USD\$94 billion, from the Global Environment Facility, Green Climate Fund, Adaptation Fund and the International Climate Initiative, for example, only \$12 billion has been spent on NbS [74]. Additionally, NbS for coastal erosion seem almost perfectly positioned to take advantage of private investment and to unlock additional funding [32]. Coastal hoteliers rely heavily on beaches for their business as tourists rate beach fronts among their most important attributes [75]. The building blocks therefore exist for viable payment for ecosystem services models, where beneficiaries (hotels) would pay for improved beach protection from NbS. Such mechanisms could also attract private investors seeking environmental and financial returns and providing the up-front capital investment to the hotels. Additional incentives for the hotels include the possibility of reducing premiums for business interruptions insurances due to beach loss. Preliminary reports indicate that the cost of NbS could be covered by these gains on insurance premiums [76].

However, there are significant challenges related to establishing such funding mechanisms, that include knowledge gaps, technical constraints and lack of track records of success. These issues affect all stakeholders: the private investors who provide funds; the coastal hoteliers who can access the funds; and the marine conservation practitioners who carry out the work and may or may not be responsible for accessing funds. Conservation practitioners are often unaware that private investment is an option, and even when they are introduced to the concept are often hesitant to access monies that require repayment [77]. Traditional private investors typically select low-risk projects with business models designed to provide the greatest returns on investments. Even those who fund

environment projects tend to concentrate on sectors, such as energy, rather than on the marine environment [78]. Technical constraints include risks associated with uncertainties of success; minimum tickets for investors, which exceed the monies required; and a shortage of investment-ready projects [79].

The issues are not insurmountable. In terms of knowledge gaps, increasing the attractiveness of marine conservation projects to private investors and of private investment to marine conservation practitioners entails improved communication and understanding of needs between two parties with historically divergent views towards money. For the technical constraints, financing strategies exist to minimize risk, such as de-risking instruments provided by development banks. At the same time, innovations in ecological engineering, for example, are increasing to reduce functional risks. With such challenges addressed, the foundation is created for projects that are ready for investment.

In Section 6, we briefly consider two private financing mechanisms for coastal hoteliers, that respond to some of the technical challenges identified.

6. Private Financing Mechanisms for Coral Restoration

Coastal hoteliers can access traditional debt finance from commercial banks for the more commonly utilized gray infrastructure to protect their beaches. This type of loan instrument, aimed at generating the highest returns with the lowest risk, is not suitable for restoration-like projects, with fledgling track records of success and ecological engineering uncertainties [80]. Payments for ecosystem service (PES) and impact investment, however, are both financing mechanisms with goals of generating positive impact, that can accommodate various levels of risk, but with very different requirements.

PES is a tool for managing ecosystems by providing incentives for behavioral changes. The system relies on beneficiaries of an ecosystem service making payments to providers of the service, which are conditional on specific targets or environmental action [81]. A review on the suitability of PES as a means of providing finance for coastal protection [82] concluded that PES would be most useful for a toolbox of coral reef conservation action within a marine protected area (MPA) framework. Such action is expected to improve the flow of a bundle of ecosystem services, including coastal protection, by improving coral reef health. Coastal hoteliers, as beneficiaries of coastal protection, could for example, make payments to MPA managers for providing the coral reef ecosystem service of protecting their beachfront. Specific targets—an intrinsic part of a PES mechanism—could include general improvements to coral health and/or restoration aimed at increasing cover of structural builders (Figure 1).

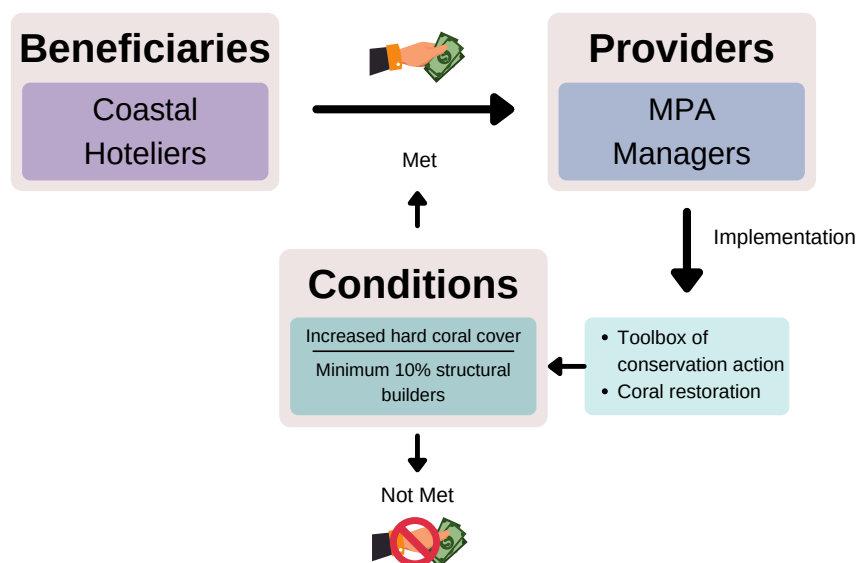


Figure 1. Potential PES mechanism to improve coastal protection.

Impact investments are made in order to generate positive and measurable environmental, social and financial impacts [83]. This is a relatively new type of endeavor where investors are willing to accept lower financial expectations in exchange for having greater environmental or social impact. However, returns are required and, in the case of impact investment in debt, the borrower must repay the capital and interest over time [84]. These returns are dependent on investor goals and can be greater or less than the market price.

There is available money, as evidenced by the market size of impact investments, which averaged \$USD715 billion in 2020 [78]. Even with good intentions and available funding, however, impact investment has not succeeded in addressing many critical environmental needs, primarily as a result of “inflexible expectations for financial returns” [85].

Catalytic capital is defined as capital that allows for inordinate risk and/or sacrifices some financial gain for social benefits [86]. It can be in the form of grants and/or a de-risking instrument, such as a loan guarantee for the impact investor from a development bank. Catalytic monies are therefore more patient and flexible than those from solely impact investment.

Combining impact investments with catalytic capital allows for investments more suited towards emerging technologies, such as coral restoration. This merging of capital with different risk levels, in order to meet the objectives of all, is called blended finance [87] and has been recommended as a means of financing NbS [80]. Blended finance solutions are complicated to develop and expensive [79]. Transaction costs, for example to ensure investment readiness can be high and there is a minimum investment requirement for impact investors, below which they will not invest.

For coastal protection, the blended finance funds can be accessed by either the hotelier or the marine conservation practitioner that is responsible for implementing the project. A potential blended finance scheme with a hotelier as the entity requesting funds is outlined in Figure 2.

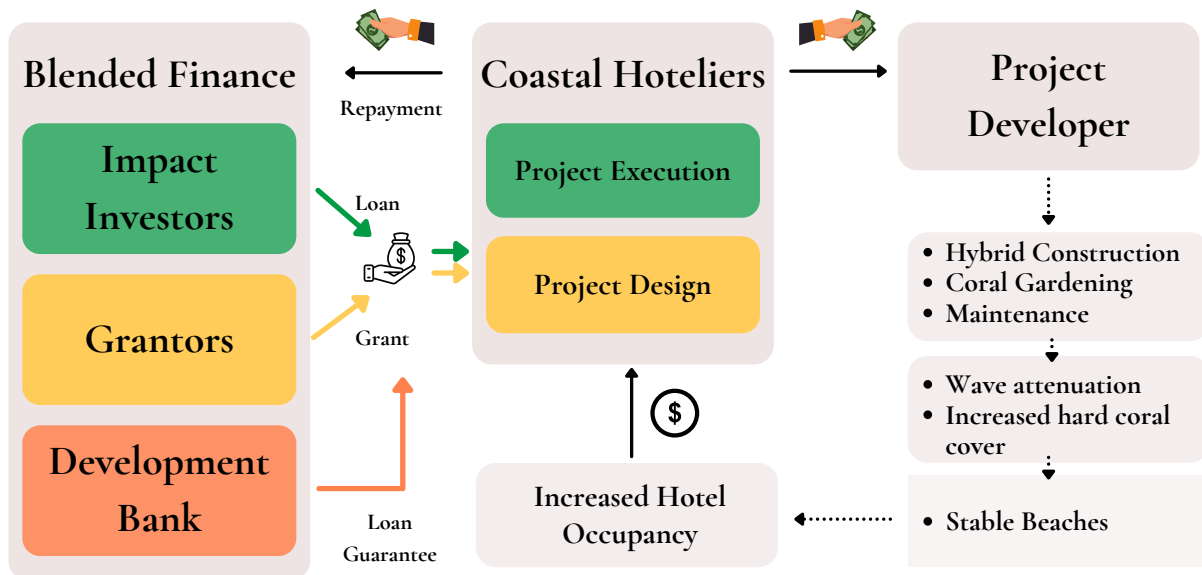


Figure 2. Potential blended finance investment model, with coastal hoteliers as recipients of funds.

Coral gardening with its low up-front cost and long timeframe to effectiveness is especially well suited to PES, once coastal erosion is not an immediate issue. As a direct ongoing, engagement between the hotelier and MPA practitioner, coral health can potentially be gradually improved over time, from a hotel’s budget, without a need for external investment, and the stress of having to make financial returns. Environmental (and social) returns can be achieved from complementary coral reef ecosystem services, such as recreational and aesthetic. Thus, benefits can be achieved during the time that corals take to become effective at attenuating waves for coastal protection. This type of arrangement

would only be feasible in cases with stable beaches, where hoteliers have time to make calm decisions for the future.

The hybrid NbS solution with its high up-front investment cost and relatively short time frame to high effectiveness is more suited to blended finance. In this case, the hotelier or the marine conservation practitioner would interact with external investors, who require financial and environmental returns on investment, within specific timeframes. The higher upfront costs will be more attractive to impact investors, coupled with a reduction in both engineering risk and timeframe, due to the presence of the artificial reef, that immediately allows for wave breakage and beach protection. At the same time, the financial risk linked to the NbS performance could be reduced by a loan guarantee from a development bank to the impact investors.

It should be noted that for both financing mechanisms, the details have been greatly simplified to demonstrate what is possible. There are many other steps and agreements required prior to implementation [79].

7. Conclusions and Recommendations

With a financing gap of \$598 to \$824 billion identified for biodiversity [88], it is imperative that monetary solutions are found. The situation is especially dire for coral reefs, which are on the brink of collapse [26]. There is little time remaining to seek greater certainty for restoration; therefore, while there are still many questions to be answered, and hence risk in terms of investment, inertia is not the most judicious course of action.

The private sector's inclusion in the world of coral reef conservation is a natural fit, as these ecosystems not only protect the investments of coastal hoteliers, but have additional income earning potential related to their aesthetics. Encouraging coastal hoteliers to invest in the protection of coral reefs that contribute to stable beaches should not be a difficult endeavor. Both high and low-risk NbS interventions and high and low costs can be accommodated, with different private sector financing mechanisms.

One of the serious challenges for coral restoration, is scaling up [47]. However, for private investment in coastal protection, one can work at the scale of one reef to protect a specific beach for coastal hoteliers who are willing to invest in their own solution. Such smaller scale solutions are also important for *inter alia* trialing new technologies and promoting tourism economies [72].

We recommend the following as means of encouraging private investment in coral restoration.

Recommendation 1: Improve the visibility of private investors to conservation practitioners, as well as communication between both parties.

With one primary hindrance being a lack of knowledge of opportunities available with private investment, solutions revolve around information transfer. The idea of economy and ecology being mutually beneficial is not a new concept, but it is taking time to gain traction. Initial ideas emerged in the 1970s and have continued to evolve to the current concept of a triple bottom line of improving economic, environmental and social targets [79,89,90]. On this trajectory, the inclusion of private sector investment in conservation is expected to increase. Sharing information on private investment opportunities for conservation can be accomplished in physical spaces, such as the Finance Pavilion of the 2021 IUCN World Conservation Congress, and online events, e.g., the Next Normal Now series organized by the Global Impact Investing Network, aimed at conservation practitioners.

Recommendation 2: Provide evidence of the ability of coral restoration to positively impact coastal protection.

Research Institutions and NGOs must start providing robust evidence of the ability of coral restoration to aid in coastal defence, and the costs to do so. Such outputs will contribute to a greater increase in private investor confidence. At present, numerical models have demonstrated that enhancing reefs can improve coastal protection [71,91], while restoration practitioners are increasingly improving their methods, success rates and reporting [47]. What is now required is implementation of coral restoration for coastal

defence, in a variety of scenarios and documentation of the outputs—both successes and failures.

Recommendation 3: Improve the attractiveness of NbS to private investors.

Governments can play key roles in incentivizing private investment in restoration. While direct funding is often limited, government subventions will remain important, with seas being common resources and for the benefit of all [28]. Further, public entities can play enormously important roles in catalyzing the flow of funds from the private sector to conservation organizations via incentives, appropriate regulatory environments and market structures. In addition, the provision of pre-hazard mitigation funding, such as the grants provided by the Federal Emergency Management Agency (FEMA) in the United States of America, can also assist in reducing the overall cost of restoration efforts.

While conservation projects, especially marine ecosystems, are not strongly attracting impact capital from investors, this is slowly changing with NGOs, such as Blue finance (www.blue-finance.org, accessed on 5 January 2022) for example structuring blended finance solutions for the MPAs under its purview.

It is unlikely that the benefits of assisting coral reef recovery and enhanced protection are larger than the cost of doing nothing [92]. The dearth of information on successes and failures of NbS has impeded investment opportunities [79]. However, with scientific evidence and numerical models strongly suggesting the likelihood of restoration succeeding at improving coastal protection, coupled with more flexible financing aimed at positive impact, “the stage is set” for private investment.

The years 2021–2030 have been designated the UN Decade on Ecosystem Restoration. It is a good opportunity, with a global focus on restoration, to highlight private investment as means of financing this type of intervention. Practitioners in the conservation and investment fields must learn and understand each other’s worlds so that this emerging source of funding can be effectively engaged and deployed.

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Chapter 8: Discussion & Conclusions

Coral reefs provide major ecosystem services but are declining in health, Effective management is required, however, this has been stymied by inadequate finance. Coastal protection is particularly important for island nations and is also a very complex service, with little direct scientific investigation on the role of the reef in service delivery and specific management measures to aid in service delivery. Without this knowledge, incentivizing the private sector to participate in conservation action is hampered. The thesis therefore investigated both financial mechanisms and ecological interventions that can assist in conserving coral reefs, with the intent of improving the flow of the coastal protection ecosystem service. Improving our understanding of the mechanisms involved in service production, identifying NbS and providing potential financing mechanisms with the private sector, facilitates the development of new and innovative economic and ecological means of improving the health of coral reefs. Each paper either raised related questions or responded directly to the aim.

In Chapter 2 we used CBAs of CRES to demonstrate the economic sense of investing in MPAs. Our most relevant result re the overall thesis was the identification of coastal protection (and tourism) as priority services, which are often interlinked in small islands. Further, coastal land owners (primarily hoteliers) were identified as priority beneficiaries. Knowledge gaps, especially with respect to linkages between ES and the processes that support them as well as the need of linking investment to ES were highlighted. Valuations of the added value of management are still rare (Wielgus et

al., 2010) and outputs provide not only incentive to governments to establish MPAs, but also to the private sector, to invest in conservation.

In Chapter 3, we focused solely on coastal protection, providing a novel methodology for valuation, in countries with low data availability. In calculating value, the role of the reef and live coral cover were determined. The latter (live coral) is especially important, as while the literature is clear that the non-living portion of the reef (the skeleton) breaks waves, the contribution of the thin veneer of live coral was not made as definitively. With conservation action only possible on living reef, an understanding of its importance was crucial in determining management measures. Our valuation method was deemed too imprecise for PES and Impact Investments, and more suited to advocacy with policy makers and private investors. Such conclusions echo those of (Waite et al., 2014), who utilized even more rigorous valuation methods. No further economic valuations were deemed necessary throughout the thesis.

In Chapter 4 an intensive review of PES and ecological requirements revealed the suitability of PES for coastal protection under specific conditions. However, while the literature is clear that coral reefs can have important roles in delivering coastal protection, quantification of that role has not yet been fully determined. Reefs primarily via wave breaking and friction, attenuate waves, which is a key element in protecting beaches. Causality has been strongly demonstrated between coral reefs and wave attenuation, with the link between coral reefs and beach protection being made less strongly. Even more tenuous, is the link between living coral cover and beach protection. In terms of coastal protection, live corals are required for wave breaking (short term) and growth and maintaining positive carbonate budgets (long

term). It is not possible at this time to state for example the abundance and diversity of coral required to ensure beach stability. However the link between healthy reefs and beach protection can be posited, with the knowledge gained from numerical models run on wave attenuation (van Zanten et al., 2014), demonstrations of the reef's role in protecting beaches (Kushner et al., 2011) and studies linking declining reef health to decreased beach protection (Sheppard et al., 2005).

With the previous caveat of our valuation method (Chapter 3) not being suitable, as well as the issues highlighted in this chapter of quantifying the service; PES was determined to be most useful when conditional on a toolbox of management interventions, such as within an MPA rather than based on the value of a single service. PES could also be conditional on maintaining 10% hard coral cover, which was identified as the minimum required for coral growth. Finally, active management in the form of reef restoration was highlighted as an important intervention for coastal protection. This paper therefore provided insight for our three queries, which were actioned in Chapter 6. With no previous papers identified in the literature, that sought to utilize PES as a financing mechanism for coastal protection, outputs have filled in knowledge gaps on PES requirements and provided the rational and framework for implementing a new option for financing coral reef conservation.

In Chapter 5, we used a case study in the Dominican Republic to explore the second financing mechanism identified in our Introduction, of Impact Investment for conservation. The blending of impact investment with public and or philanthropic monies was outlined, as well as important and enabling factors, including collaborative management with an existing MPA, tangible revenue streams and

effective enforcement. This is the only example of impact investment for MPAs found in the literature, and outputs align with the benefits of co-management and entrepreneurial MPAs (Riedmiller, 2000) which play key roles in this mechanism. With this paper, the feasibility of using Impact Investment for marine conservation has been outlined.

In Chapter 6 outputs from the previous chapters were used in “crafting” suitable solutions for coastal protection. A method was devised for selecting reefs that provide the service, then the positive impact of restoring reefs demonstrated, and options provided for doing so. The additionality obtained from using such Nature-based Solutions over traditional gray infrastructure was highlighted for mitigating coastal erosion and financing mechanisms matched to restoration type. We further showed how the ecological engineering risk of project failure could be reduced via the use of hybrid structures. Within this paper, we were able to answer our three queries by demonstrating the role of specific reefs in service delivery, identifying which management interventions could aid in promoting coastal protection and suggesting financing mechanisms suitable to each intervention. The findings from this study provide the incentive and provide conceptually a framework for private investors interested in playing a greater role in financing NbS for coastal protection.

Our NbS findings are similar to those found in the literature, with the exception of costs of NbS. Many authors (Fabian et al., 2013, Ferrario et al., 2014, Sutton-Grier et al., 2015, Rinkevich, 2021) have suggested that NbS are less costly than gray infrastructure, however in this paper, we found that cost was dependent on the type of NbS. Coral gardening was, as expected less expensive than gray infrastructure,

however the hybrid solutions, which provide the least risk, were both more expensive than gray infrastructure. While this might initially serve as a deterrent to investment, the additionality offered by such hybrids, especially over long timeframes must be highlighted and communicated to potential investors.

8.1 Study Limitations and Next Steps

In spite of the extensive research carried out in these papers, the efficiency of our Nature-based Solutions to improve service flow for coastal protection has not been tested. We have based the majority of our conclusions on literature reviews, basic numerical modelling and expert interviews. Related to this is that in spite of best efforts on site and within the boundaries of MPAs, actions made outside of this designated area could still have major impacts on project success (Bruno et al., 2019). A neighboring hotel for example could construct a breakwater which results in beach starvation. A pollution event outside the MPA boundaries could result in full mortality of corals. This is one of the key reasons that the MPAs themselves should fall within areas that are managed in a holistic fashion (e.g. Integrated Coastal Zone Management) which would reduce the risk of negative impacts. Both active and passive management interventions are therefore required.

Numerous complexities exist within coral reefs, many of which are unknown (Moberg and Folke, 1999, Brandl et al., 2019). Attempting to replace what was lost for coral reefs is impossible and our restoration actions can only hope to give corals the time and boost needed to recover in specific areas. Reef restoration cannot ensure that the full ecosystem service provision is replaced.

Much of the research was carried out via case studies, which cannot be extrapolated to other sites, due to site specific complexities. Coral reefs are extremely important for service flow, however there are a many other factors that influence this service related to: the reef (e.g. shape, structural complexity); waves (e.g. velocity, energy, height); and coastal features (e.g. coastal geography, elevation, presence of vegetation). Intensive investigations will be required per site.

The most important next steps identified are:

Ecology/Engineering – Quantification of the coastal protection service and strengthening of the relationship between coral reef health and beach protection

Reef Restoration – i. Exploration of eco-engineering technologies and sexual reproduction strategies ii. Improved understanding and quantification of the impact of reef restoration on beach protection.

Financing Mechanism – i. Communication between conservation practitioners and private investors ii. Practical implementation of private investment in marine conservation.

8.2 Concluding Statements

The work carried out in this thesis highlights the importance of coastal protection as well as the role of coral reefs in providing it, and lays the foundation for implementation of NbS that can improve service provision. The framework has been provided for interventions both ecological and financial, that can be built upon. This

and much of the work found in the literature is theoretical. We must move to the next step and put the ideas into practice, which would entail the construction of interventions with the requisite intensive modelling to define size, slope and elevation for the NbS. At the same time, outputs introduce the concept of private investment to coastal managers and provide information on mechanisms that can be utilized. This “paves the way” for new and exciting financing mechanisms to complement the traditional means, with an aim of closing the financing gap, a critical step in improving coral reef health .

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Chapter 6: Conceptualisation; Writing, Review and Editing

Chapter 7: Conceptualisation; Writing, Review and Editing

RÉSUMÉ

Les récifs coralliens déclinent à l'échelle mondiale en raison d'impacts locaux et mondiaux, exacerbés par des financements inadéquats pour leur gestion.

La protection côtière est un service écosystémique (SE) important rendu par les récifs coralliens, en particulier avec des impacts négatifs croissants du changement climatique mondial. Ce SE n'a cependant pas retenu la même attention que d'autres SE, peut-être en raison de difficultés à déterminer clairement le rôle du récif corallien dans la fourniture du service.

Cette thèse étudie des solutions pour mieux comprendre, quantifier et potentiellement optimiser le SE de protection du littoral rendu par les récifs coralliens. Le travail explore aussi le rôle du secteur privé, comme bénéficiaire du SE, dans le financement des interventions.

La thèse est divisée en 7 chapitres suivant l'ordre suivant :

Chapitre 1 - Fournit une introduction générale et le contexte de la thèse, en présentant les éléments essentiels de la protection, de la gestion et du financement de la conservation du littoral, y compris la justification des sélections effectuées.

Chapitre 2 – Article: « Bénéfices économiques des investissements publics dans les aires marines protégées (AMPs) ». L'article propose un cadre pour l'analyse écologique et économique des services écosystémiques des récifs coralliens. Dans cet article, nous avons mené des analyses coûts-avantages sur la gestion des récifs coralliens dans les aires marines protégées de 2 nations insulaires. La protection du littoral a été identifiée comme l'un des SE les plus importants pour les propriétaires immobiliers et l'industrie du tourisme.

Chapitre 3 - Article: « Évaluation économique du service écosystémique des récifs coralliens de protection côtière : une approche pragmatique ». L'article fournit une méthodologie d'évaluation plus précise de la protection côtière fournie par les récifs coralliens, en particulier pour les pays où les données sont peu disponibles.

Chapitre 4 - Article: « Sous quelles conditions le paiement pour les services écosystémiques convient-il à la protection côtière dérivée des récifs coralliens?: un examen des exigences scientifiques ». L'article examine la fourniture par les récifs coralliens de la protection côtière et détermine sa faisabilité pour un paiement pour les services écosystémiques (PSE).

Chapitre 5 - Article: « Investissement avec impact dans la conservation marine » . L'article, explore plus en détail les moyens innovants de financer la conservation marine via des investissements d'impact et des financements mixtes.

Chapitre 6 - Article: « Restauration des développement de solutions écologiques et économiques pour la protection côtière » . Dans cet article, nous démontrons l'impact positif de la restauration

des récifs, proposons des options et montrons l'additionnalité obtenue en comparant ces solutions (basées sur la nature) avec des infrastructures grises traditionnelles.

Chapitre 7 – Article « Des capitaux privés pour améliorer la protection des côtes tropicales : le moment d'un coup de pouce? » , synthétise les résultats des chapitres précédents et conclut que la restauration des récifs financée par des capitaux privés est peut-être une opportunité d'améliorer la santé des récifs pour la protection des côtes.

Les résultats de la recherche montrent qu'en dépit des complexités écologiques associées au service écosystémique de la protection côtière rendus par les récifs coralliens, l'investissement privé dans la restauration des coraux est prometteur en tant que moyen de contribuer au flux de services. Les progrès de l'écologie et de l'ingénierie permettent régulièrement d'identifier et de quantifier le rôle des récifs coralliens dans la protection côtière. De même, les solutions basées sur la nature sont de plus en plus affinées et consolidées, permettant d'améliorer la santé des récifs et augmenter l'atténuation des vagues. En parallèle, certains acteurs de l'investissement privé semblent maintenant plus enclins à accepter des rendements financiers plus faibles, si les retours environnementaux et sociaux sont présents.

MOTS-CLÉS

récifs coralliens, protection côtière, financement de la conservation, solutions fondées sur la nature, restauration des coraux, payment for ecosystem services

ABSTRACT

Coral reefs are declining on a global scale from local and global impacts, exacerbated by inadequate and unsustainable funding for their management. It is imperative that we determine the most suitable methods for improving reef health, in a world where the situation is both dire and time sensitive.

Coastal Protection is an extremely important coral reef Ecosystem Service (ES) especially with the impending and increasing negative impacts of Global Climate Change (GCC) on coastal and marine ecosystems. This ES has however not received the attention of other ES, perhaps due to challenges in clearly determining the role of the coral reef and living coral, in providing the service.

This thesis therefore investigates solutions for improving coastal protection and exploring the role of the private sector in financing interventions.

Chapter 1- Provides a general introduction and background for the thesis, by presenting the core elements of coastal protection, management and conservation finance, including the rationale for selections made.

Chapter 2 - **Evidence of economic benefits for public investment in MPAs**, sets the “scene” for the following chapters, by proposing a framework for the ecological and economic analysis of coral reef ecosystem services. In this paper we conducted Cost Benefit Analyses on coral reef management within Marine Protected Areas of 2 island nations. Coastal protection was identified as one of the most important ES for real estate owners and the tourism industry.

Chapter 3 - **Economic valuation of coral reef ecosystem service of coastal protection: A pragmatic approach**, provides a more precise valuation methodology for coastal protection provided by coral reefs, specifically for countries with low data availability.

Chapter 4 - **When are Payment for Ecosystems Services suitable for Coral Reef derived coastal protection?: a review of scientific requirements** reviews coral reef provision of coastal protection and determines its feasibility for a Payment for Ecosystem Services (PES). In the process, the role of living coral was analyzed and management actions identified that could improve reef health for service provision. This paper identifies coral restoration as a key intervention for coastal protection, identifies primary beneficiaries and provides the economic rationale for Chapters 5 and 6.

Chapter 5 - **Impact Investment in marine conservation**, further explores innovative means of financing marine conservation via impact investments and blended finance.

Chapter 6 - **Coral reef restoration for coastal protection: Crafting technical and financial solutions**, further develops ecological and economic solutions for coastal protection. We demonstrate the positive impact of restoring reefs, provide options for doing so, and show the additionality obtained from using such Nature-based Solutions over traditional gray infrastructure for mitigating coastal erosion.

Chapter 7 - **Private capital to improve tropical coastal protection: time for a boost**, synthesizes the outputs of the preceding chapters and concludes that while restoration financed by private capital is perhaps our best chance at improving reef health for coastal protection.

Outputs from the research, show that in spite of ecological complexities associated with coastal protection, private investment in coral restoration is promising as a means of contributing to service flow. Advances in ecology and engineering are steadily allowing for the role of coral reefs in coastal protection to be identified and quantified, while NbS interventions are being refined and discovered, to improve reef health and increase wave attenuation. At the same time, the private investment sector’s priorities are shifting from a strong reliance on financial returns, to an acceptance of lower financial returns, once environmental and social returns are also achieved.

KEYWORDS

coral reefs, coastal protection, conservation finance, nature-based solutions, coral restoration, payment for ecosystem services