Harnessing the climate mitigation, conservation and poverty alleviation potential of seagrasses: prospects for developing blue carbon initiatives and payment for ecosystem service programmes

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Prospects for developing seagrass blue carbon initiatives and payment for ecosystem service programmes

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Abstract (1905 characters; 2000 limit)

Seagrass ecosystems provide numerous ecosystem services that support coastal communities around the world. They sustain abundant marine life as well as commercial and artisanal fisheries, and help protect shorelines from coastal erosion. Additionally, seagrass meadows are a globally significant sink for carbon and represent a key ecosystem for combating climate change. However, seagrass habitats are suffering rapid global decline. Despite recognition of the importance of 'Blue Carbon', no functioning seagrass restoration or conservation projects supported by carbon finance currently operate, and the policies and frameworks to achieve this have not been developed. Yet, seagrass ecosystems could play a central role in addressing important international research questions regarding the natural mechanisms through which the ocean and the seabed can mitigate climate change, and how ecosystem structure links to service provision. The relative inattention that seagrass ecosystems have received represents both a serious oversight and a major missed opportunity. In this paper we review the prospects of further inclusion of seagrass ecosystems in climate policy frameworks, with a particular focus on carbon storage and sequestration, as well as the potential for developing payment for ecosystem service (PES) schemes that are complementary to carbon management. Prospects for the inclusion of seagrass Blue Carbon in regulatory compliance markets are currently limited; yet despite the risks the voluntary carbon sector offers the most immediately attractive avenue for the development of carbon credits. Given the array of ecosystem services seagrass ecosystems provide the most viable route to combat climate change, ensure seagrass conservation and improve livelihoods may be to complement any carbon payments with seagrass PES schemes based on the provision of additional ecosystem services.

Keywords: Blue Carbon, carbon sequestration, coastal management, marine conservation, payments for ecosystem services, poverty alleviation

1 Introduction

Seagrasses represent a diverse and globally distributed group of aquatic flowering plants (angiosperms) with up to 76 species occurring in boreal, temperate, and tropical waters (Green and Short 2003). Seagrass meadows are commonly dominated by a single species, although in tropical regions meadows comprising 12 distinct species have been recorded. They are often significant or dominant primary producers, supporting local food-webs and driving local nutrient cycles (Hoard et al., 1989; Gullstrom et al. 2008; Hemming and Duarte 2000). Seagrass meadows are primarily adapted to coastal environments where their spatial distribution is heavily influenced by environmental factors such as light, temperature, salinity, nutrient availability and wave action (Hemming and Duarte 2000). However, the shallow coastal habitat colonized by most seagrass meadows means they are especially prone to significant human-related disturbance (Waycott et al. 2009).

Human actions provide a triumvirate of environmental, biological and climatological stressors that act across spatial and temporal scales delivering locally-specific impacts (Orth et al. 2006). Drivers of seagrass ecosystem decline include: eutrophication and solid waste disposal (nutrient pollution); aquaculture; thermal pollution; physical alteration or habitat damage (via dredging, coastal infrastructural developments, land reclamation and mechanical destruction); disease spread and invasive species introductions; climate change; over-fishing; overexploitation; and land-runoff from deforestation, mining and agriculture (Duarte 2002; Erftemeijer and Robin Lewis III 2006; Orth et al. 2006; Waycott et al. 2009; Short et al. 2011; Zuidema et al. 2011; Hicks and McClanahan 2012; Cullen-Unsworth and Unsworth 2013; Cullen-Unsworth et al. 2013; Baker et al. 2014).

Over several decades the global integrity of seagrass ecosystems has been seriously undermined by business-as-usual approaches to coastal development (Duarte 2002). Occurrences fuelled by increasing population densities in coastal regions, which are about three times higher than the global average and increasing (Small and Nicholls 2003). In some cases rapid population growth and urban expansion has shifted farming practices towards increased agricultural output leading to the persistent eutrophication of coastal lagoons and reduced seagrass biomass (Rivera-Guzmań et al. 2013). Similarly, nutrient loading and sedimentation have markedly reduced the extent of several seagrass meadow sites in the Western Pacific (Short et al. 2014).

Globally, twenty four percent of seagrass species are now classified as threatened or near threatened on the IUCN's Red List (Short et al. 2011). Estimates of the rate of seagrass decline have increased over the last 70 years, from 0.9% yr^{-1} prior to 1940 rising to 7% yr^{-1} since 1980 (Waycott et al. 2009; Duarte et al. 2013a; Fourqurean et al. 2012). The global decline of seagrass ecosystems threatens to exacerbate climate change (Duarte et al. 2010; Kennedy et al. 2010; Fourqurean et al. 2012; Lavery et al. 2013), undermine the supply of a range of other ecosystem services (Bujang et al. 2006; Orth et al. 2006; Waycott et al. 2009; Short et al. 2011; Cullen-Unsworth and Unsworth 2013) and consequently detrimentally affect subsistence livelihoods (Cullen and Unsworth 2010; Nordlund et al. 2010).

This reality reflects the complexity of seagrass ecosystems, particularly the connections seagrass meadows have with marine and terrestrial systems, and therefore the difficulties and challenges associated with their management, which are embedded within broader coastal and ocean management issues (Rudd and Lawton 2013). For example, in a recent global ocean research priorities exercise (Rudd 2014) several top-ranked priorities had implications for seagrass ecosystems, including: 'greenhouse gas flux' (7th); 'climate change mitigation and manipulation' (8th); 'ecosystem structure to service linkages' (16th); 'upland hydrology effects on oceans' (24th); 'coastal hazard management' (35th); 'ecosystem management alternatives' (40th) and 'integrated upland coastal management' (43rd). Our view is that research is needed on multiple fronts to create enabling conditions and the evidence base needed to craft innovative new policy tools for conservation and mitigating the potential adverse effects of climate change.

Our purpose here is to summarize the prospects for using new approaches to aid seagrass conservation. This will help address key coastal and ocean research questions, and provide substantive direction on future seagrass research needs. We address these issues in the context of incorporating seagrass habitats into climate change mitigation strategies jointly focused on ecosystem service provision, carbon management and livelihood support. In particular, we analyse prospective financing options in relation to carbon management, alongside other investment opportunities for including seagrass meadows into incentive-based mechanisms (e.g., PES) through a co-benefit and bundled ecosystem service approach. In so doing we consider science, policy and governance perspectives acknowledging the important barriers and challenges existing across those domains.

We examine five key issues. In Section 2, we summarize ecosystem services (ES) provided by seagrass ecosystems and the salient information needed concerning these ES to develop incentive schemes. In Section 3, we ask how ecosystem service valuation information could be applied to design and implement new policy innovations. In Section 4, we examine the prospects for seagrass carbon finance based on current climate policy frameworks. In Section 5, we broaden the scope to financing instruments that could be developed based on the multiple ES that seagrass ecosystems provide. Lastly, in Section 6, we summarize the key design, implementation and governance issues that must be addressed to bring functioning seagrass PES schemes to fruition. In addition, we highlight the relevant ocean priority research questions that relate to each stage (Rudd 2014), setting our seagrass-oriented research in the broader context of ocean research prioritization.

2 Seagrass ecosystems and ecosystem services

Seagrass ecosystems provide supporting, regulating, provisioning, and cultural ecosystem services (Barbier et al. 2011; Raheem et al. 2012; Cullen-Unsworth and Unsworth 2013). Here we briefly summarize the literature regarding the most widely cited ES supplied by seagrass ecosystems and the information required about each in order to include them in an incentive mechanism (Table 1).

Table 1 Seagrass ecosystem services and the corresponding information needed to contribute towards incentive scheme development

Ecosystem Service	What we need to know[a]
Climate regulation (carbon storage and sequestration)	(a) The spatial distribution, density and species assemblage of seagrass meadows. Two relatively accurate and reliable means of achieving this are: • Acoustic side scan sonar which is useful up to 25m depths and has been used to map seagrass communities in the Mediterranean (e.g., Montelfalcone et al. 2013; Sanchez-Carnero et al. 2012). • Remote sensing, for example, Landsat 5 TM and 7 Enhanced Thematic Mapper, which is more appropriate for shallow waters of between 2 to 5m and has been used in Australia (e.g., Dekker et al. 2005; Phinn et al. 2008), Zanzibar (e.g., Gullström et al. 2006) and the Coral Triangle (Torres-Pulliza et al. 2014) (b) Assessment of carbon stocks, rate of accumulation (e.g., Duarte et al. 2013a; Fourqurean et al. 2012; Macreadie et al. 2013), in particular: • Belowground biomass and soil: soil depth (thickness of deposit), dry bulk density and organic carbon content (Duarte et al. 1998; Fourqurean et al. 2012) • Aboveground biomass: represents only ~0.3% of total organic carbon stock (Duarte and Chiscano 1999) • Accumulation rate: direct measurement, radiocarbon, ^{210}Pb, soil elevation (Duarte et al. 2013a)
Erosion and natural hazard regulation (coastal and shoreline protection)	(a) Local vegetative characteristics such as canopy height, shoot density and below-ground biomass (e.g., Bouma and Amos, 2012; Christiansen et al. 2013; Ondiviela et al. 2013) (b) Bulk density, organic content of sediment and porosity (e.g., de Boer 2007)
Biodiversity	(a) Species inventory, richness, diversity and community structure (e.g., Bell and

	Pollard, 1989; Barnes 2013) (b) Habitat usage of fish species and correlations with life-cycle stages (e.g., Heck, 2003; Jaxion-Harm et al. 2012; Seitz et al. 2014) (c) Presence of charismatic and Red List species (e.g., Williams and Heck Jr, 2001)
Fisheries	(a) Fish species caught, landed and sold (e.g., average catch sizes, market value etc.) (b) Frequency, location(s) and time spent fishing, for example, by using participatory GIS (e.g., Baldwin, Mahon & McConney 2013; Baldwin & Oxenford 2014) (c) Degree of overlap between commercial and artisanal fish species (i.e. commercial fishing impacts on artisanal fishing activities) (d) Types of fishing methods and gear employed and their capacity to damage seagrass beds (e.g., Tudela, 2004) (e) Invertebrate gleaning activities (e.g., species gleaned, frequency of gleaning etc. Unsworth and Cullen, 2010)
Nutrient cycling and water quality Regulation	(a) Seagrass biomass and production (e.g., de Boer, 2007) (b) Levels of leaf litter (e.g., Chiu et al. 2013) (c) Water turbidity (e.g., Petus et al. 2014) (d) Dissolved nutrient concentration (e.g., Cabaco et al. 2013)
Cultural services (tourism and recreation)	(a) Hotels (coastal distribution and ownership of land) (b) Tourist numbers , demographics and usage of inshore areas (reasons for use) (c) Local employment of staff in tourism (community-based tourism e.g., Salazar 2012 (Tanzania); Kibicho 2008; Steinicke and Neuburger 2012 (Kenya)) (d) Local food supply to hotels (e.g., Pillay and Rogerson 2012) (e) Associated infrastructure developments and impacts on seagrass meadow stability (e.g., Daby 2003 in Mauritias; Zuidema et al. 2011 Turks and Cacos Islands)
Cultural services (social-ecological)	(a) Composition of household income and reliance on seagrass-derived ecosystem services (b) Gender differences in use and benefits derived from seagrass meadows e.g., gleaning vs. fishing (e.g., Cullen-Unsworth et al. 2013) (c) Cultural significance of seagrass meadows to 'traditional way of life' (e.g., Unsworth and Cullen, 2010
Ecosystem Service Threats	(a) Agricultural land run-off : nutrient loading (e.g., Waycott et al. 2009) (b) Coastal developments and population and urban impacts: infrastructure, conversion of seagrass meadow beds to alternative uses, sewage discharge (e.g., Short et al. 2011; 2014)

[a] In relation to the information outlined three points need to be emphasised: First, it is not necessary to obtain detailed information on all ES provided by seagrasses to develop a payment scheme. Second, their needs to be agreement between the operating scale of the payment scheme and the scale at which ES information is acquired. Third, the information we list is not meant to be exhaustive.

2.1 Regulating services: climate regulation

Historically, seagrass meadows had been virtually ignored in global carbon budgets (Duarte et al., 2005). More recently their role in combating climate change through carbon storage and sequestration has become more clearly recognised, while simultaneously the spatial extent of seagrass meadows has continued to decline (Duarte et al. 2010; Kennedy et al. 2010; Fourqurean et al. 2012; Duarte et al. 2013a; Lavery et al. 2013). Although a small fraction (18 to 60 x 10^6 ha) of the world's ocean area seagrass meadows sequester 20% of global marine carbon and store 10% of annual buried organic carbon (C_{org}) (Fourqurean et al. 2012; Pendleton et al. 2012). Consequently, seagrass ecosystems play potentially central roles in how oceanic ecosystems can mitigate climate change, a question ranked 8th in global importance by marine scientists (Rudd 2014).

Seagrass meadows are highly productive systems, especially in Indo-Pacific regions, and provide habitat for diverse communities (Short et al. 2011). However, worldwide, seagrass standing biomass is small (76-151 Tg C) relative to the biomass of the vegetation in other coastal ecosystems (Fourqurean et al. 2012). Nonetheless, the high productivity of seagrass meadows, with potential net community production (NCP) of 6.7 t C ha^{-1} yr^{-1} (several times higher than NCP rates associated with Amazonian forests and North American wetlands), contributes significantly to their carbon sink capacity (Duarte et al. 2010). Approximately 20% to 60% of this aboveground productivity derives from the autotrophic epiphytes that seagrass meadows support (Duarte et al. 2013a). Moreover, seagrass meadows trap allochthonous material, including large amounts of particulate carbon, which combined with their ability to bury carbon enables seagrass meadows to store large amounts of carbon (Duarte et al. 2013a).

Carbon stored belowground, as dead roots and rhizomes and as C_{org}, may be stable for millennia (Duarte et al. 2010; 2013a). However, the amount of C_{org} locked beneath seagrass beds varies considerably according to the interplay of different abiotic and biotic drivers, with the result that in some cases deposits of organic-rich sediments beneath seagrass meadows can be up to 11m thick (Duarte et al. 2013a). In addition, most seagrass production (approximately 80%) is not consumed by herbivores and may therefore be buried, where a combination of low nutrient content and anoxic sediment conditions contributes to low rates of remineralization aiding long-term storage (Duarte et al. 2013a). Burial rates are therefore somewhat difficult to estimate; however, the most robust data suggests mean local C_{org} burial rates of 1.2-1.38 t C ha^{-1} yr^{-1}: equivalent to 30-50% of NCP (Kennedy et al. 2010; Duarte et al. 2013b). Nevertheless, others (Siikamaki et al. 2013) have suggested a much lower burial rate, equivalent to 0.54 t C ha^{-1} yr^{-1}.

Globally, the organic carbon that accumulates in the sediments below seagrass meadows is much greater (4.2 to 8.4 Pg C) than the biomass (Fourqurean et al. 2012). However, the areal extent of seagrass meadows is poorly mapped, meaning these estimates remain highly uncertain (Duarte et al. 2013b; Lavery et al. 2013). Further uncertainties arise from the fact that some 50% of below-ground carbon derives from autochthonous production while almost 50% is contributed from phytoplankton and terrestrial sources (Kennedy et al. 2010; Duarte et al. 2013a). Indeed, significant quantities of carbon are also exported away from seagrass meadows to adjacent areas, although the fate of this carbon is poorly understood (Duarte et al. 2010; 2013a).

Despite the uncertainties, alongside the lack of attention given to the potential implications of extensive conversion of standing carbon pools beneath vegetative coastal ecosystems more generally, it is clear that seagrass meadows constitute an important global carbon sink whose continued loss threatens to exacerbate climate change (Duarte et al. 2010; Pendleton et al. 2012). Indeed, global carbon emissions maybe enhanced by an additional 3% to 19% from the destruction of vegetative coastal ecosystems (Pendleton et al. 2012). Based on current assessments seagrass biomass loss may release between 11 and 23 Tg C yr^{-1} into the ocean-atmosphere system, and a further 63 to 297 Tg C yr^{-1} into the ocean-atmosphere CO_2 reservoir through the oxidization of the underlying sediment (Fourqurean et al. 2012). Additionally, seagrass loss reduces the overall carbon accumulation rate (equivalent to between 6 and 24 Tg C yr^{-1}). Collectively, this represents considerable CO_2 emission potential (131 to 522 Mg CO_2 ha^{-1}), a figure comparable to roughly 10% of that emitted annually from land-use change, with associated economic costs approaching US$1.9 to 13.7 billion yr^{-1} (Fourqurean et al. 2012; Pendleton et al. 2012).

2.2 Regulating services: erosion and natural hazard regulation

Coastal vegetated wetlands such as seagrass meadows can provide effective natural protection from the destructive powers of storms and wave action (Barbier et al. 2008; Bouma and Amos 2012; Duarte et al. 2013b). They are therefore important ecosystems to study in order to understand the spatial extent, frequency, and risk of marine hazards affecting coastal waters and how their effects can be minimized (ranked 35th in Rudd, 2014). Direct coastal protection is achieved through energy dissipation resulting from wave breaking, friction and energy reflection (Ondiviela et al. 2013), processes significantly influenced by seagrass shoot density and canopy structure (Hansen and Reidenbach 2013). Even low biomass and heavily grazed seagrass meadows can significantly reduce wave action by decreasing the hydrodynamic energy associated with current flow (Christiansen et al. 2013). For example, in temperate regions current velocities have been reduced by up to 60% in summer (high biomass) compared to 40% in winter (low biomass) in relation to adjacent non-vegetated sites (Hansen and Reidenbach 2013). By reducing wave action and current velocities seagrass habitats also protect the seafloor against hydrodynamic 'shear stresses' (de Boer 2007).

Seagrass canopies act as efficient filters, stripping particles from the water column and adding to sediment accumulation (Hendriks et al. 2007). Soil accretion (~1.5mm yr^{-1}) is important in helping coastal wetlands, and seagrass meadows in particular, adapt to sea level rise (Kirwan and Megonigal 2013; Lavery et al. 2013), thus contributing to Rudd's (2014) 26th ranked question on sea level rise and vulnerable coasts. Below-ground seagrass biomass is particularly important for sediment accretion as well as stabilization against storm erosion (Bos et al. 2007; Christiansen et al. 2013). By helping to immobilise sediment, seagrass meadows also reduce re-suspension and increase water transparency (Duarte 2002; Ondiviela et al. 2013). In the Arabian Gulf, for example, sediment stabilization and shoreline protection represent important ecosystem service functions of seagrass meadows (Erftemeijer and Shuail 2012). Overall, the effectiveness and efficiency of the coastal protection services provided by seagrass ecosystems varies across spatial and temporal scales due to differences in species type (i.e. vegetative characteristics), coastal distribution, flow-vegetation interactions and water dynamic properties (Ondiviela et al. 2013). In monetary terms, the erosion control services provided by seagrass beds (inclusive of algal beds) have been estimated at US\$25,000 $ha^{-1}yr^{-1}$ (Costanza et al. 2014).

2.3 Provisioning services: biodiversity and fish nurseries

The physical and biological structure of seagrass meadows is central to their significance as a marine biotope (Gullström et al. 2008; Saenger et al. 2013). The high primary productivity of seagrass, their epiphytes and associated benthic algae provide an important energy source to support local, transient and distant food webs (Heck et al., 2008). In addition, the structural complexity of seagrass meadows offers sites for attachment and a place to avoid predation (Farina et al. 2009). These attributes mean seagrass meadows function as foraging areas, refuges and nursery habitats for diverse communities of marine life, many of which are commercially important or endangered (Bujang et al. 2006; Orth et al. 2006; Unsworth and Cullen 2010; Erftemeijer and Shuail 2012; Jaxion-Harm et al. 2012; Browne et al. 2013; Cullen-Unsworth and Unsworth 2013). Organic matter produced in seagrass meadows is also exported to adjacent ecosystems and supports a large range of marine and terrestrial consumers (Heck et al., 2008). Connectivity between mangrove and seagrass ecosystems has also been shown to be important for supporting inshore fisheries, the abundance and assemblage of fish and crustacean communities and fish life-cycle stages (Bosire et al. 2012; Saenger et al. 2013). Seagrass ecosystems are thus important for ocean priority research questions on biodiversity contributions to ecosystem function (ranked 6th) and biological connectivity (ranked 28th) (Rudd 2014).

2.4 Supporting services: nutrient cycling

Seagrass meadows are directly involved in nutrient cycling through their uptake of water column nutrients, storage in biomass, detritus and sediment, and indirectly through the effect of seagrass metabolism on water column and sediment nutrient re-cycling (Saenger et al. 2013). The nutrient cycling capacity of seagrass meadows has been estimated to contribute about US\$26,000 $ha^{-1}yr^{-1}$, or US\$1.9 trillion in aggregate, to the global economy (Waycott et al. 2009; Costanza et al. 2014).

2.5 Cultural services: social relations

Wetland ecosystems play vital cultural, economic and ecological roles, supporting livelihoods and reducing poverty (Verma and Negandhi 2011; Kumar et al. 2011; Senaratna Sellamuttu et al. 2011). Frequently, the fish and marine invertebrate populations supported by intact seagrass ecosystems maintain stocks of commercial and artisanal importance, and their exploitation makes significant economic and food security contributions to many coastal communities (Jackson et al., 2001). In some cases seagrass supported fisheries may provide a harvest value of up to US\$3500 ha^{-1} yr^{-1} (Waycott et al. 2009). In Tarut Bay, (Arabian Gulf), seagrass ecosystems support a US\$22 million yr^{-1} fishery (Erftemeijer and Shuail 2012). Prawns are also the basis for extensive fisheries, particularly along warm-temperate and tropical coastlines, and previous estimates of the potential total annual yield from seagrass ecosystems in Northern Queensland, (Australia), equated to a landed value of between AUS$0.6 million to AUS$2.2 million yr^{-1} (Watson et al., 1993). In the Caribbean and Indo-Pacific region valuable species such as queen conch (*Euatrombus gigas*), spiny lobster (Palinuridae), and smudgespot spinefoot (*Siganus canaliculatus*) also support local fisheries (Cullen-Unsworth and Unsworth 2013; Baker et al. 2014).

Shellfish gleaning frequently supports artisanal fishers' subsistence and generates income for rural households (Unsworth and Cullen 2010). Invertebrate harvesters in Zanzibar, East Africa, can earn between US$8.51 to US$17.01 per catch from gleaning activities, emphasizing the social-ecological connections between coastal community livelihoods and seagrass ecosystem functioning (Nordlund et al. 2010). In some locations, the scale of inshore fisheries supported by seagrass ecosystems have been shown to be more significant (in economic terms) than those supported by mangroves or coral reefs. Recent evidence from Chwaka Bay (Zanzibar) indicated that fishers spend 70% of their time fishing seagrass meadows and preferred fishing there compared to mangrove and coral reef habitats (de la Torre-Castro et al. 2014). As a consequence, over 50% of the fish sold in the central market derived from seagrass meadows. In Wakatobi National Park (Indonesia), 60% of invertebrate collectors and 40% of fishers and gleaners preferred harvesting from seagrass meadows compared to 20% of collectors, fishers and gleaners who preferred to harvest exclusively from coral reefs (Unsworth et al. 2010).

3 The value of ecosystem services provided by seagrass ecosystems

3.1 Ecosystem service valuation

Valuing ecosystem services has become an increasingly important tool for demonstrating the significance of biodiversity and ES to society and informing policy decisions (Gomez-Baggethun et al. 2010; Brondizo et al. 2010; Dendoncker et al. 2014; Liekens and De Nocker 2014). ES valuations have been criticized for focusing disproportionately on utilitarian values, overly commodifying nature and ignoring ecological complexity, potentially leading to erroneous policy decisions (Kosoy and Corbera, 2010; Norgarrd, 2010; Gowdy and Baveve, 2014). In light of these criticisms efforts to value ES have increasingly sought to focus on integrating the ecological, social and economic dimensions of ES into a unified whole (Figure S1 Supplementary Material) (TEEB, 2010; UK NEA 2011; Dendoncker et al. 2014). A plethora of monetary and non-monetary techniques have recently been developed to try and capture the broadest range of 'values' across the breadth of ecosystem services (Table S1 Supplementary Material). Seagrass ecosystems provide a potentially tractable environment within which to conduct multi-faceted valuation research and address an important ocean research question (ranked 53rd, Rudd, 2014) on ecosystem service valuation implications.

3.2 Seagrass and wetland valuation studies

Economic valuations of seagrass ecosystems remain few in number, with most focusing on the market value of commercial fisheries as the primary ecosystem service of importance (Table 2).

Table 2 Valuation studies of seagrass meadows

Study	Location	Description	Value
Watson et al. (1993)	Queensland (Australia)	Multi-species prawn fishery	A$1.2 million yr^{-1}
McArthur & Boland (2006)	South Australia (Australia)	Secondary fisheries production	A$114 million yr^{-1}
Unsworth et al. (2010)	Wakatobi National Park (Indonesia)	Ecological and socio-economic assessments of the importance of seagrass meadow fisheries	US$230 million (value extrapolated to the national level)

Kamimura et al. (2011)	Seto Inland Sea (Japan)	Wild juvenile black rockfish (*Sebastes cheni*) production	US$78600 yr^{-1}
Rudd & Weigand (2011)	Newfoundland, Canada	Choice experiment to estimate citizens' willingness to pay (WTP) for improvements in three ecosystem services associated with a reduction in wastewater pollution in the Humber Arm, with eelgrass (*Zostera marin*) used as an indicator for estuarine biological diversity	$2.63 sq km^{-1} $household^{-1}$ yr^{-1}
Lavery et al. (2013)	Australia	Estimation of the value of stored organic carbon beneath Australia's seagrass ecosystems (17 habitats, 10 seagrass species). Valuation based on the C_{org} content of the top 25cm of sediment	A$3.9-5.4 billion
Vassallo et al. (2013)	Isle of Bergeggi (Italy)	Natural capital assessment of *Posidonia oceanica* seagrass meadows using emergy analysis. Focused on the collective value of four ecosystem services: nursery function, sedimentation and hydrodynamics, primary production and oxygen release	€172 m^{-2} a^{-1}
Tuya et al. (2014)	Gran Canaria Island (Spain)	Primary and secondary fisheries associated with *Cymodocea nodosa* seagrass meadows	€673269 yr^{-1} (whole island value)
Blandon and zu Ermgassen (2014)	South Australia (Australia)	Meta-analysis of juvenile fish abundance to assess the juvenile fish enhancement capacity of seagrass ecosystems. Thirteen commercial fish established to be recruitment enhanced	Species were enhanced by approx. A$230000 ha^{-1} yr^{-1}

In several respects seagrass ecosystems have been marginalized in favour of other coastal and estuarine ecosystems, meaning valuation studies conducted for other wetland biotopes (i.e. mangroves, coral reefs and saltmarshes) are the only suitable avenue to identify comparative estimates for commonly shared ecosystem services that may offer insights into the expected range of values for seagrass meadows (Table 3).

Table 3 Valuation studies of coastal and wetland ecosystem services

Study[a]	**Description**	**Ecosystem Service Values**
Barbier et al. (2011)	Global synthesis of estuarine and coastal ecosystem services	*Coastal protection*: (US$174 ha^{-1} yr^{-1} coral reefs in the Indian Ocean, US$236 ha^{-1} yr^{-1} saltmarshes in the US and US$966-1082 ha^{-1} yr^{-1} mangroves in Thailand). *Maintenance of fisheries*: (US$5-45000 Km-^{2} yr^{-1} coral reefs (local consumption and exports) in the Philippines, US$647-981 $acre^{-1}$ saltmarshes (recreational fishing) in Florida and US$708-987 ha^{-1} mangroves in Thailand).*Carbon sequestration*: (US$30.50 ha^{-1} yr^{-1} for saltmarshes and mangroves based on global sequestration rates). *Tourism, Recreation and Research*: (US$88,000 coral reefs in the Seychelles, £31.60 $person^{-1}$ saltmarsh (otter habitat creation) in the UK)
UNEP (2011)	Total economic value of the ecosystem services delivered by mangroves in Gazi Bay, Kenya	*Total Economic Valuation*: (US$1092 ha^{-1} yr^{-1}) e.g., *Fishery*: (US$44 ha^{-1} yr^{-1}) *Coastal protection*: (US$91.7 ha^{-1} yr^{-1}) *Carbon sequestration*: (US$126 ha^{-1} yr^{-1}) *Biodiversity*: (US$5 ha-1 yr-1) *Existence value*: (US$594.4 ha^{-1} yr^{-1})
Verma and Negandhi (2011)	Livelihood dependency and economic evaluation of the Bhopal wetland, India	*Fisheries production*: (US$33 $month^{-1}$ $fisherman^{-1}$) *Boating activities* (US$2264 yr^{-1} $boatman^{-1}$) *waterchestnut cultivation* (US$222 yr^{-1} $family^{-1}$) *cloth washing activities* (US$66 $month^{-1}$ $household^{-1}$) *secondary activities e.g., sugar cane juice sellers* (US$6000 yr^{-1})
Brander et al. (2012)	Meta-analysis of the value of ES supplied by mangroves mainly in	Overall mean and (median) value: US$4185(239) ha^{-1} yr^{-1}

	Southeast Asia. Valuations based predominantly on fisheries, fuel wood, coastal protection and water quality	
Salem and Mercer (2012)	Global estimates of mangrove ecosystem services	*Fisheries*: (US$23,613 ha^{-1} yr^{-1}). *Forestry*: (US$38,115 ha^{-1} yr^{-1}). *Recreation and Tourism*: (US$37,927 ha^{-1} yr^{-1}). *Non-Use*: (US$17,373 ha^{-1} yr^{-1}). *Water purification*: (US$4,784 ha^{-1} yr^{-1})
Brander et al. (2013)	Global meta-analysis of ES delivered by wetland systems in agricultural landscapes, with a focus on three regulating services: flood control, water supply and nutrient cycling	Mean and (median) values presented *Flood control*: US$6923(427) ha^{-1} yr^{-1} *Water supply*: US$3398(57) ha^{-1} yr^{-1} *Nutrient cycling*: US$5788(243) ha^{-1} yr^{-1}
Camacho-Valdez et al. (2013)	Socio-economic benefit of saltmarshes in Northwest Mexico	US$1 billion yr^{-1}
James et al. (2013)	The social (non-monetary) values attached to mangroves across three villages in the Niger Delta region of Nigeria. Social values assessed were: therapeutic, amenity, heritage, spiritual and existence	Mean values for the village-level importance placed on these aspects of the social value of mangroves *Therapeutic*: (14%-71%) *Amenity*: (65%-73%) *Heritage:* (70%-92%) *Spiritual*: (44%-76%) *Existence*: (89%-91%)
Kakuru et al. (2013)	Wetland ecosystem services in Uganda	*Flood control*: (US$1.7 billion yr^{-1}). *Water regulation*: (US$7 million yr^{-1})

[a] The examples we cite are not meant to be exhaustive but rather illustrative of the different types of services and values attributed to a range of coastal wetland ecosystems, and are therefore to be seen as a guide for the range of possible valuations that may be attributed to seagrass ecosystems.

Overall, the lack of in-depth local studies spanning different continents and regions valuing the breadth of ecosystem services provided by seagrass ecosystems needs to be remedied, with particular focus on qualitative value attributions associated with the social-ecological dynamics of seagrass systems (Cullen-Unsworth and Unsworth 2013). This view supported the wider sentiment articulated by Raheem et al. (2012:1169), that "there is a dearth of spatially explicit non-market values for services provided by coastal and other ecosystems", and by the Abu Dhabi Global Environmental Data Initiative (AGEDI 2014:10), that the "option of combining Blue Carbon with other ecosystem services valuation should be kept open to provide multiple potential values that can support conservation activities". Strengthening the evidence base regarding the global economic value of oceans (ranked 48th, Rudd 2014) requires site-specific seagrass ecosystem valuation efforts that can be used to derive transfer values from meta-analyses (e.g., Brander et al. 2012; Johnston et al. 2005).

4 Policy frameworks for Blue Carbon management

Recent thinking about Blue Carbon acknowledges the special importance of the carbon storage and sequestration capability of coastal and marine wetlands and organisms in global climate change scenarios and policies (Sifleet et al. 2011; Vaidyanathan 2011). Blue Carbon sinks capture and store amounts of carbon equivalent to up to half of global transport emissions (~ 400 Tg C yr^{-1}) yet their inclusion in current mitigation and adaptation programs has been very limited (UNEP 2009; Tommaso et al., 2014). Developments could occur in the regulated (compliance) or the unregulated (voluntary) carbon sectors. We take each in turn.

4.1 The regulated sector

4.1.1 Policies and processes

Collectively, the United Nations Framework Convention on Climate Change (UNFCCC 1992, Article 4 (d)), Manado Ocean Declaration (2009), Cancún Agreement (2010) and Rio Ocean Declaration (2012) provide opportunities for development of Blue Carbon initiatives. In practice, however, current policy processes inadequately account for the restoration and protection of Blue Carbon systems (Grimsditch 2011; Murray et al. 2011). This is due, in part, to the initial bias towards terrestrial climate change mitigation and adaptation activities within the UNFCCC, alongside the acknowledgment that practical expansion to coastal and marine systems (from principled intentions) would require further international agreement (Murray et al. 2012). However, as a recent report indicates (UNEP and CIFOR 2014: x) 'climate change mitigation frameworks developed for terrestrial ecosystems can be extended to include coastal wetlands'.

There are clear points of entry for Blue Carbon funded activities under the parallel pathways of the UNFCCC, specifically: the Land Use and Land-Use Change and Forestry (LULUCF) and the clean development mechanism (CDM) of the Kyoto Protocol; and the Reduced Emissions from Deforestation and forest Degradation + (REDD+) and Nationally Appropriate Mitigation Actions (NAMAs) of the Durban Platform. In many cases these entry points require altering or reinterpreting definitions (Gordon et al. 2011; Grimsditch 2011; Murray et al. 2011; 2012). Nevertheless, some argue that by the Paris COP 21 meeting in 2015 negotiations are likely to have reached a consensus for including an approach for Blue Carbon accounting under the UNFCCC (UNEP and CIFOR 2014).

4.1.2 Kyoto protocol opportunities

Limited possibilities exist within the Kyoto Protocol (Murray et al. 2012). However, some progress has been made through the recently updated Intergovernmental Panel for Climate Change (IPCC) guidelines. The so-called 'Wetlands Supplement' includes guidance for national governments to report carbon emissions and removals for specific management activities in coastal wetlands (e.g., mangroves, tidal marshes and seagrass meadows) (IPCC 2014). The activities that national governments will be able include in their national inventories for greenhouse gases covers forest management in mangroves, certain aspects of aquaculture, drainage and restoration or creation of coastal wetlands. However, this supplementary regulation is 'encouraged but not mandatory in context of any other activities under Article 3, paragraphs 3 and 4, of the Kyoto Protocol' (UNFCCC 2014).

Moreover, extension of current LULUCF definitions to cover wetland ecosystems is lacking (Murray et al. 2012). However, with the publication of the IPCC Wetland Supplement the case for not including a broader set of definitions that specifically mention wetlands is harder to justify. Furthermore, activities under LULUCF could include avoided wetland degradation via alternative use or prohibiting disturbance (Herr et al. 2012). With regards to baseline credit mechanisms such as the CDM, in 2011 a mangrove project was approved as an afforestation and reforestation activity. However, the methodology applied is specifically for mangroves and not (so far at least) transferable to tidal marshes or seagrass meadows (Lovelock & McAllister 2013). Moreover, the much more substantial avoided emissions resulting from protecting Blue Carbon pools remain outside this mechanism (Murray et al. 2011; 2012).

4.1.3 Durban platform opportunities

The Durban Platform provides more scope for Blue Carbon activities. Mangroves are now covered by REDD+ (Grimsditch 2011). However, seagrass inclusion remains some way off: this would require a broader definition of 'forests' as well as an extension of emission and reduction activities across all land-uses (i.e., Agriculture, Forestry and Other Land Uses, AFOLU) (Murray et al. 2011; 2012; Siikamaki et al. 2013). Nevertheless, AFOLU projects do include a variety of carbon accounting protocols relating to biomass, C_{org} and greenhouse gas emissions (UNEP and CIFOR 2014). There have been calls to decouple carbon sequestration and emissions arising from habitat degradation (Grimsditch 2011). This is particularly important for seagrass meadows where the 'real' carbon of interest is buried in the sediment. Under REDD+, deciding what aspects of the Blue Carbon pool (i.e. sediment/soil-carbon or above-ground biomass) count would be especially important (Murray et al. 2011). Extension of REDD+ to seagrass meadows could easily see them contributing to reduced emissions via the degradation pathway, through a focus on management strategies linked to tackling the negative impacts of nutrient loading for example (Seifert-Granzin 2010). Developments to include tidal wetland restoration and conservation under REDD+ are currently on-going (UNEP and CIFOR 2014).

NAMAs offer the most direct route for funding Blue Carbon enterprises because countries have autonomy over the activities that form part of their national strategies, and could reasonably protect and restore wetland and coastal ecosystems (Grimsditch 2011; Herr et al. 2012; Murray et al. 2012). Furthermore, the green climate fund provides finances for programs in accordance with NAMAs that could be directed towards Blue Carbon activities (Herr et al. 2012). However, the challenge remains that inclusion of these activities under a national framework would still require measurement, reporting and verification approval (Murray et al. 2012).

4.2 The voluntary sector

4.2.1 The global voluntary carbon market

The voluntary carbon market (VCM) accounts for 0.1% and 0.02% of the value and volume of the regulated global carbon market respectively (Benessaiah 2012). Yet rapid sector expansion has led to increasing interest from governments, particularly in relation to carbon standards and registries (Peters-Stanley and Yin 2013). The principal attraction of the VCM is its deregulated nature, which helps to reduce transaction costs and stimulate innovation. However, the trade-off to this regulatory flexibility is market uncertainty and depression of the carbon price, which can have serious implications for expected project returns (Benessaiah 2012; Thompson et al. 2014). Project size is also a determinant of offset price, with smaller projects garnering higher carbon prices for carbon dioxide equivalent (CO_2e). The average carbon price for micro projects (i.e., those generating less than 5 Kt $CO_2e\ yr^{-1}$) was recently US$10/t$CO_2e$, whereas the mean carbon price for mega projects (i.e. those generating more than 1 Mt $CO_2e\ yr^{-1}$) was US$5.8/t$CO_2e$ (Peters-Stanley and Yin 2013).

Worldwide carbon standards have expanded from concentrating purely on carbon accounting to emphasising co-benefits (Peters-Stanley and Hamilton, 2012). This has been driven, particularly in the private sector, by an increasing interest in measuring and verifying non-carbon project outcomes (Peters-Stanley and Yin 2013). Programs are progressively focusing on climate change adaptation, public health, gender issues and biodiversity as additional attributes to non-carbon benefits (Peters-Stanley and Yin 2013) (Table 4). For example, the verified carbon standard (VCS), which accounts for 55% of market share, considers climate, community and biodiversity (16%) and Social Carbon (2%) co-benefits (Peters-Stanley and Yin 2013). This is important for ecosystems such as seagrass meadows that provide multiple benefits in addition to carbon storage as those benefits might be captured via broader standard attributes.

Table 4 Carbon standards appropriate for joint environmental and development projects

Carbon Standard and Credits	Description
Gold Standard *(acquired Carbon Fix Standard)*	Carbon accounting + embedded co-benefits
Plan Vivo	Carbon accounting + embedded co-benefits
VCS	Carbon accounting + tagged co-benefits
VCS and CCB	This joint process is premised on the notion that forestry and land-use projects with be better able to meet emission reduction targets and achieve co-benefits if validation/verification costs are lowered
Social Carbon	Co-benefits (needs to be accompanied by a carbon accounting standard)
Global Conservation Standard	Developed for the purposes of ensuring conservation can deliver

	payments to local landholders, the accounting system is based on the 'stock' amount of identifiable and measurable ecosystem service benefits – credited through the use of Conservation Credit Units (CCUs). The first protocol established CCUs based on carbon stocks in vegetation.
Women's Carbon Standard	Certifying the role, engagement and leadership of women in carbon projects. Jointly administered by Women Organising for Change in Agriculture and Natural Resource Management – WOCAN
Vulnerability Reduction Credits	Acknowledges and qualifies reduction in community vulnerability arising from adaptation efforts. Administered by the Higher Ground Foundation
The Poverty Alleviation Criteria Tool	Measures the poverty alleviation outcomes resulting from forestry and other land-use projects implemented under the Panda Standard. Developed jointly by ACR (American Carbon Registry) and the China Beijing Environmental Exchange

Another important development for coastal wetland systems such as seagrass meadows is that the VCM has highlighted the special connections between carbon and water. Both VCS and the American Carbon Registry (ACR) have coastal wetland accredited carbon accounting methodologies (Peters-Stanley and Hamilton 2012; Thomas 2014). For example, in the Mississippi Delta the ACR has developed a wetland restoration protocol (UNEP and CIFOR 2014). Furthermore, VCS has also developed a soil carbon sampling methodology that could be transferred to wetland and peatland ecosystems (Peters-Stanley and Yin 2013). Indeed, VCS methodologies cover the full array of Blue Carbon activities, from restoration and re-vegetation to conservation and management, and in late 2013, the 'Greenhouse Gas Accounting Methods for Tidal Wetlands and Seagrass Restoration' methodology was submitted to VCS and is currently awaiting approval (UNEP and CIFOR 2014).

Although the increasing alignment between livelihood development and carbon management is welcomed, several challenges exist. Specifically, a lack of appropriate markets, negotiating trade-offs between maximizing economic efficiency and ensuring equity in benefit flows, and adequately socially embedding payment schemes. These challenges relate to broader issues of the transaction costs of ocean management (ranked 57th, Rudd 2014). Developing inclusive sustainable livelihood VCM projects depends on the provision of secure property rights and tenure arrangements regarding the ownership and use of resources. However, providing secure property rights alongside certification can be prohibitively expensive (e.g., CCB certification is estimated at US$4000 – US$8000) even though adequately accounting for costs and securing financial streams is essential (Benessaiah 2012). Negotiating investment risk and return uncertainty are significant challenges in community-based carbon projects where non-compliance and complex program arrangements are pressing issues. Likewise, the provision of 'enabling institutions' for effective administrative, operational and implementation performance remains crucial. Nevertheless, the advantages of the voluntary carbon market outweigh the downsides and present a more immediately attractive option even if in some quarters the regulated carbon market is the preferred long-term option (Benessaiah 2012; Ullman et al. 2012).

4.2.2 Multilateral environmental agreements

The sustainability of estuarine, coastal and marine habitats, with regards to their use, conservation, restoration and in climate change mitigation and adaptation have been alluded to under several regional and international multilateral agreements for example: the Convention on Biological Diversity (CBD); Ramsar Convention on Wetlands (Ramsar); UNEP Global Programme of Action for the Protection of the Marine Environment from Landbased Activities (GPA-Marine); Convention for the Protection of the Marine Environment and Coastal Areas of the South-East Pacific (Lima Convention) and the South Pacific Regional Environment Programme (SPREP).

Although predominantly management and advocacy-related, some of these programs offer financial support for Blue Carbon activities (Laffoley 2013).

4.2.3 National level policies

Research evaluating the ways in which vegetative coastal ecosystem services and carbon in particular can be included in national level statues and policies is lacking, partly as a result of the highly individual nature of national legislation. However, Pendleton et al. (2013) have identified how such 'coastal carbon' could be incorporated under a subset of existing U.S. federal statutes and policies including the National Environmental Policy Act, the Comprehensive Environment Response, Compensation and Liability Act, the Oil Pollution Act, the Clean Water Act and the Coastal Zone Management Act amongst several others. The analysis indicates that although coastal carbon services are not currently accounted for under existing federal-level legislation, to do so would be relatively straightforward and consistent with the implementation of these regulations (Pendleton et al. 2013). Nevertheless, despite this relative ease, incorporating coastal carbon into existing federal legislation would require further improvements in the availability of expertise, guidance and procedures for assessing the value of coastal carbon, quantifying the impacts of projects on carbon storage and sequestration and mapping the spatial dynamics of coastal ecosystems. The lack of precedent (i.e., the formal assessment and analysis of the benefit-costs of coastal carbon economics values in these regulations) was also recognised as an important limitation that would need to be overcome for wider 'coastal carbon functions' to be frequently included in regulatory assessments (Pendleton et al. 2013). Importantly, these considerations are equally applicable to State-level legislation as they are to other national legislative policies and statutes in other countries.

4.2.4 Blue Carbon demonstration sites and the future

Recent research, policy and financing advancements in Blue Carbon relevant to seagrass meadows include global programs. The Blue Carbon Initiative (www.thebluecarboninitiative.org) focused on integrating Blue Carbon activities into the UNFCCC and other carbon financing mechanisms (Herr et al. 2012; Thomas 2014). Charities such as The Ocean Foundation and partners (www.seagrassgrow.org) have developed a Blue Carbon calculator that determines CO_2 emission reduction offsets in terms of the protection and restoration of seagrass meadows (a method pending formal approval by the VCS). Collectively, developments such as the Blue Carbon portal (www.bluecarbonportal.org) and work by Bredbenner (2013) and Thomas (2014) have demonstrated the current global extent of Blue Carbon activities. In particular, significant work remains to establish a functioning global network of fully implemented Blue Carbon programmes involving the active transfer of carbon credits (Locatelli et al. 2014). In this regard, securing private financing of Blue Carbon activities will become increasingly important (Thomas, 2014). Presently, Blue Carbon programs are predominantly research-oriented, in the early stages of development and mangrove-focused, with few directed efforts towards seagrass ecosystems (Table 5) (Bredbenner 2013).

Table 5 Seagrass-related Blue Carbon initiatives

Blue Carbon Project	Description
Long-term ecological research in the Patos Lagoon Estuary (Brazil) – Institute of Oceanography and Federal University of Rio Grande	Spatial and temporal description of seagrass and macroalgae vegetation changes. Mapping, biomass and sedimentation sampling for carbon stock evaluation
National seagrass ecosystem mapping (Brazil) - Universidade Estadual de Rio de Janeiro, Universidade Federal do Rio Grande, Universidade Federal de Santa Catariana e Universidade Federal Rural de Pernambuco	Spatial mapping of Brazil's seagrass ecosystems, distribution and extent, and the determination of the associated carbon stock

Seagrass and Mangrove pilot assessments (Indonesia) - Agency for Research and Development of Marine and Fisheries, Ministry of Marine Affairs, Fisheries-Indonesia	Three pilot areas: Banten, East Kalimantan and North Sulawesi – field surveys, mapping and biophysical sampling of seagrass and mangrove systems to assess carbon storage and sequestration, alongside the socio-economic value of these systems for improving policy
Mangrove, saltmarsh and seagrass Blue Carbon potential (China) - Tsinghua University, Xiamen University, State Oceanic Administration	Assessment of the Blue Carbon potential of these ecosystems (i.e. carbon storage and sequestration) to provide evidence to support habitat restoration linked to carbon credit scheme

Adapted from Bredbenner (2013)

5 Seagrass habitats: prospects for PES

Here we explore opportunities for developing seagrass PES programmes. The options we describe should be seen as working in tandem with carbon-credit schemes not as mutually exclusive alternatives.

5.1 A brief explanation of PES

PES programs are marketed as win-win opportunities, supporting conservation and the sustainable use of natural resources while improving rural livelihoods (van Noordwijk et al. 2007; Muradian and Rival 2012; Pokorny et al. 2012).Yet, what constitutes PES, both in theory and practice, and PES success is open to debate (e.g., Wunder 2005; Farley and Costanza 2010; Muradian et al. 2010; Hejnowicz et al. 2014). This is largely due to the plurality of financial arrangements underpinning PES schemes, which include government-financed, user-financed or hybrid co-financed arrangements, often involving external donors, such that the ways in which they function do not conform to a single operational standard (Schomers & Matzdorf, 2013). Financially speaking, however, they can (generally) be thought of as a form of direct payment based on the beneficiary pays principle (Parker and Cranford, 2010). Within typical PES programs (Lin and Nakamura 2012; Tacconi 2012; Derissen and Latacz-Lohmann 2013, Martin-Ortega et al. 2013), ES providers (e.g., landholders, farmers or communities) voluntarily participate in a program whereby they receive payments from ES buyers (e.g., a government, a utility or private organisation). Transactions are facilitated by a single or multiple set of intermediary actors (e.g., a semi-autonomous body or non-governmental organisation). In return for payments, providers adopt alternative land-use practices and management strategies that can secure and deliver a set of important ES to a wider beneficiary population.

Institutionally, PES programs are generally framed as decentralized instruments favouring bottom-up solutions to land management issues (Landen-Mills and Porras 2002; Bond and Mayers, 2010). Despite the diversity of contexts in which PES schemes operate, they tend to adopt common modes of activity such as restricting agricultural development, proposing alternative cropping arrangements, reducing deforestation and expanding forests (e.g., reforestation and afforestation), or protecting watershed and hydrological services (e.g., Aquith et al. 2008; Bennett 2008; Muňoz-Pina et al. 2008; Porras 2010; World Bank 2010; Wunder and Alban 2008). Consequently, PES involves multiple partners across sectors as well as spanning spatial and temporal scales (Schomers and Matzdorf 2013). To function properly, schemes need to be acceptable to stakeholders, take the form of contractual obligations to which all participating parties agree, have specified objectives, be operationally transparent, and provide payments (in monetary or in-kind terms) to ES providers that account for (ideally) the full range of their opportunity costs (Wunder et al. 2008; Hejnowicz et al. 2014).

5.2 PES case studies and some considerations

Examples relevant to guiding the development of seagrass payment schemes need to involve community approaches to natural resource management, as well as the provision of multiple ES with a focus on carbon management (e.g., Table S2 Supplementary Material). Schemes seeking to deliver multiple ES via incentive mechanisms must also tackle the issue of stacking and bundling (Box 1). That is to say, determining what ES are to be provided, whether they will be paid for individually (i.e., stacked) or collectively (i.e., bundled), and what form payments will take (Bianco 2009). Additionally, PES programs need to ensure that as part of their design and implementation they maximize biodiversity and social co-benefits by adopting a decoupled approach to benefit maximization (recognizing individual ES properties and spatial attributes), ensuring management decisions account for internal and external costs, and increasing social co-benefit provision by concentrating on economic and cultural context (Greiner and Stanley 2013; Phelps et al. 2013; Potts et al. 2013).

Box 1 Stacking and bundling ecosystem services

Stacking refers to the receipt of multiple payments for different ES provided from a single plot or parcel (Bianco, 2009; Cooley and Olander 2012). Cooley and Olander (2012) recognise three forms of stacking, namely*: horizontal* (whereby individual management practices performed on spatially distinct areas each receive a payment); *vertical* (where a single management practice employed on spatially overlapping areas receives multiple payments) and *temporal* (essentially a *vertical* form of stacking where payments are disbursed over time according to the production of different ES).

Advantages of stacking: (i) delivers management that provides multiple services from programs concerned with specific services; (ii) potentially increases programme uptake rates and therefore ES provision, (iii) encourages large-scale projects that could not operate through single payments e.g., wetland restoration, (iv) may increase buyer diversification, and (v) incrementally stacking payments in an optimum way for a particular project can help raise necessary funds (Bianco 2009; Cooley and Olander 2012; Robert and Sterger 2013).

Disadvantages of stacking: (i) stacking can make it difficult to demonstrate how ES delivered by mitigation projects have abated environmental impacts allowed through offset sales; (ii) stacking may undermine project 'additionality' e.g., if payments are more than that required to initiate a project, or are for an activity that would have occurred in the absence of the project, and (iii) stacking indirectly encourages 'double counting' – paying twice for (in essence) the same service where similar services overlap e.g., water quality credits and wetland mitigation credits (Bianco 2009; Cooley and Olander 2012).

In the case of bundling, single payments are received for the provision of multiple ES from an individual parcel – importantly payment amounts are not (generally speaking) based on the summation of the individual values of each ES (Cooley and Olander 2012).

Advantages of bundling: (i) recognises the interconnectedness of ES processes and production; (ii) is beneficial for biodiversity and conservation (where broad conservation outcomes are sought); (iii) may increase the overall provision of individual ES from a parcel; (iv) can reduce administrative and transaction costs and raise price premiums, and (v) may reduce the degree of infrastructure needed to support a functioning market (Greenhalgh 2008; Wendland et al. 2010; Deal et al. 2012; Robert and Sterger 2013).

Disadvantages of bundling: (i) optimising multiple ES is difficult and given the uncertainty regarding quantification may lead to unintended trade-offs; (ii) limited knowledge concerning ES provision means accurately modelling ES spatial delivery and distribution is highly complex; (iii) regulatory requirements may mean that it is necessary to 'unbundle' specific services from the broader set; (iv) it can be difficult to demonstrate additionality and mitigate against double counting, and (v) performance related payments can be difficult to manage as ES bundle provision varies with time (Greenhalgh 2008; Wendland et al. 2010; Deal et al. 2012; Robert and Sterger 2013).

Projects that employ either stacking or bundling need to ensure they have resolved the issues of additionality and double counting before proceeding (Bianco 2009).

5.3 Seagrass PES scheme options

5.3.1 Regulating fisheries and developing protected areas

Many possible institutions are available to control and direct fishing activities along coasts and marine ecosystems (Rudd 2004). They may involve fishing gear and net restrictions, limiting fishing permits to local residents and restricting the exploitation of connected habitats while providing alternative income generating projects and ‘legal’ fishing equipment (e.g., Mnazi Bay Ruvuma Estuary Marine Park, Tanzania – Alberts et al. 2012; Mohammed 2012). Enforcing closed fishing seasons while providing wage supplements to fishers to offset opportunity costs resulting from deferred fishing activities is another approach (e.g., the defeso system in Brazil – Bergossi et al. 2011, Bergossi et al. 2012). Seagrass PES schemes may often involve creating marine protected areas (MPAs), safeguarding the underlying resource base supporting coastal communities and compensating local fishers for lost income resulting from harvesting restrictions (Table 6). Designating ‘no-take-zones’ to increase habitat cover and fish stocks, and compensating fishers for lost income is a strategy that some external non-governmental organisation (NGO) donors have used (e.g., Kuruwitu Conservation and Welfare Association in Kenya – Mohammed 2012). Setting up seagrass PES schemes requires research in a number of areas identified as priorities (Rudd 2014), including the role of MPAs on ecological resilience (ranked 30^{th}) and their effect on human well-being (ranked 45^{th}). Questions regarding compliance with rules (ranked 58^{th}) and the capacity of communities to manage their coasts (ranked 56^{th}) also demonstrate the potential value of seagrass PES development beyond the sector, as programs provide valuable opportunities to learn broad lessons about the interactions between social and ecological systems.

Table 6 Examples of marine conservation agreements securing coastal conservation and livelihood development opportunities

Country	**Project Summary**
Ecuador, Galera-San Francisco Marine Area – operating since 2008	• Established to combat issues of overfishing, pollution, habitat destruction and coastal construction. • Local communities involved in the structuring of the conservation agreement and in the management of the conservation area. • Conservation agreement covers lobster fishing, no-take areas, fishing regulations and patrol zones. • Benefits to the community include employment in patrolling, management and user rights, access to markets for alternative income streams and capacity building. • Funded by the Nature Conservancy and Conservation International (via conservation stewardship programme) and Walton Foundation (via eastern tropical pacific seascape) – requires government investment to maintain the program in the long-term.
Fiji, Bio-prospecting and Live Rock Harvesting – earliest projects since 1997	• Example of locally managed marine areas (of which 200 currently exist involving 300 communities covering 30% of inshore fisheries). • *Bio-prospecting*: External private organisations make agreements with local communities facilitated by the University of South Pacific (USP) and regulated by the government; with benefits directed to local resource owners (fees paid by these companies are channelled to a district conservation and education trust fund). • *Live Rock Harvesting*: To substitute the removal of the natural reef base with artificially created reef-bases for aquarium traders. Local users are granted management and access rights over parts of the seabed. Walt Smith International signs agreements with local villages and trains individuals to artificially culture and harvest ‘live rocks’. Villages pay US$0.25/Kg of bare rock and receive US$0.50/Kg of ‘live rock’. USP also purchases 5000Kg of material for each village on the stipulation that almost two-thirds of the proceeds are put back into the live rock harvesting process.
Indonesia, Koon Island, Maluku Marine Conservation area – 2011 to 2014 (with option for yearly renewal)	• Comprises a marine protected area, a no-take-zone (to protect spawning grounds) and a rights-based sustainable fishery (also involving a local fishery cooperative partnering with a local fishing company). • Established to protect biodiversity, maintain a sustainable fishery and enhance community development. • A community foundation has been created (TUBIRNUIATA) to implement project activities such as patrols which employ paid community members. • Funding is mainly through philanthropic sources as well as WWF-Indonesia – also attempting to establish a number of ecotourism initiatives.

Indonesia, Penemu and Bambu Islands, West Papua – Marine Conservation Area – from 2011 to 2036	• Comprises a no-take-zone and sustainable fishery, for the purposes of conservation, ecotourism and community development. • Project developed with a local non-profit organisation Taman Perlindungan Laut (TPL) and Sea Sanctuaries Trust (SST). • Marine conservation agreement is a contract between TPL/SST and the Pam Island Communities, with the purpose of developing ecotourism businesses to provide alternative livelihood revenue streams and sustain the program long-term. Benefiting local communities through employment opportunities, technical assistance and access to goods and services. • Aims to be self-funding after ten years.
Tanzania, Chumbe Island Coral Park, Zanzibar – established since 1992	• Private marine reserve, which includes 30 hectares designated as a marine reef sanctuary (coral reef and seagrass beds) plus an additional 20 hectares of coral rag forest, for the purposes of conservation, research, eco-tourism and local education. • Chumbe Island Coral Park Ltd established the park through management contracts and a lease from the Zanzibar government, and has since become an international ecotourism destination and conservation area. • The ecotourism component fully covers management costs. Several international conservation and development donors have been involved with specific local conservation and education programmes. • The Park trains and employs local people as rangers, guides and hospitality personnel. Guides and rangers also function as educators to communicate to local fisherman the importance of the reef bed and maintaining a no-take-zone. Local people have benefitted through increased incomes, access to markets for local goods, technical assistance and improved fish stocks.

Examples adapted from The Nature Conservancy's Marine Conservation Agreements: Practitioner's Toolkit (http://www.mcatoolkit.org/)

5.3.2 Ecotourism

MPA managers and coastal businesses may establish "green" levies or taxes for resort tourists and charge user-fees for diving access and licenses. Revenues generated by these charges can be re-invested to support continued management activities to enforce the operating rules and ensure compliance, conserve and restore seagrass beds, and create employment opportunities for local community members (Lutz, 2011). In this respect, participation of the private sector can be transformative for scheme development by acting as a powerful ally in conservation outreach, providing new sources of financial support and creating employment and income opportunities alongside appropriate public sector institutions (e.g., the Indonesian Yayasan Karang Lestari coral restoration project and Marin tourism park on the island of Gili Trawangan – Bottema and Bush 2012).

5.3.3 Linking farming, industry and watershed and coastal management

Eutrophication and hypoxia resulting from nutrient loading and upland pollution are significant threats to the health of seagrass ecosystems (Waycott et al. 2009; Short et al. 2011; 2014). Because upstream land-use activities can negatively affect seagrass ecosystems (Freeman 2008; Rivera-Guzmán et al. 2013) the conditions necessary for emulating watershed payment schemes are ripe (Porras et al. 2013). This may involve cross-sector collaborative partnerships between local and international NGOs, who are often project initiators and intermediary facilitators, working together with public utilities, private firms and government organisations acting as ES buyers (Porras et al. 2008; Schomers and Matzdorf 2013). Benefits to water quality and reduced water treatment costs save public utilities and private firms significant financial outlays, which may then be channelled into project start-up costs and payments for participants. Examples include the equitable PES schemes for watershed services in Tanzania and Honduras (CARE 2009; Branca et al. 2011; Kosoy et al. 2007). Collectively, these examples highlight the integrated nature of coastal and terrestrial systems and demonstrate that PES schemes which acknowledge these interactions begin to address Rudd's (2014) questions on 'upland hydrology effects on oceans' and 'integrated upland coastal management' ranked (24^{th}) and (43^{rd}) overall.

5.3.4 Biodiversity conservation

Many turtle populations nest in coastal regions supported by seagrass ecosystems (Cullen-Unsworth and Unsworth 2013). These iconic and charismatic species are increasingly threatened by poaching and so ensuring healthy nesting populations is vital (Koch et al. 2006). Protecting seagrass ecosystems may be a cost-effective and financially viable option for sea turtle conservation. Paying locals to monitor nesting sites and fisherman for releasing live turtles caught in fishing gear provides a direct and additional income stream for local communities (Ferraro 2009; Mohammed 2012). In nesting projects locals usually receive two payments: a flat fee for identifying nest locations; and a variable payment based on hatching success. Successful examples include Natamu Turtle Watch and Knunga Marine National Reserve Conservation and Development Project in Kenya and Sea Sense on Mafia Island in Tanzania (Ferraro 2009).

Due to positive willingness to pay (WTP) for sea turtle conservation among citizens of developed countries (e.g., Rudd 2009), there are also opportunities for developing international PES schemes that transfer funds from developed countries, where WTP for iconic species conservation is high, to developing countries where turtle nesting grounds and critical life stages occur. For other seagrass-dependent iconic species that enjoy an international profile, there may be similar opportunities as for sea turtles. Seagrass ecosystem conservation and management may thus provide lessons in how triage decisions for species at risk (ranked 32nd, Rudd 2014) are conceptualized and implemented.

5.3.5 Restoration

Seagrass ecosystems are declining yearly (Unsworth 2014). To reverse this global trend seagrass restoration (in suitable areas) offers an effective means to rehabilitate carbon stores and sinks (Duarte et al. 2013c) whilst enhancing other equally important ecosystem services (Greiner et al. 2013). A recent seagrass restoration CO_2 accumulation model, examining long-term trends in carbon sequestration for several commonly planted seagrass species, demonstrated that at an optimal density carbon accumulation of 177 to 1337 t CO_2 ha^{-1} after 50 years could be achieved (Duarte et al. 2013c). However, although seagrass restoration has a relatively long history, particularly in the USA, it still remains limited in scope and success (Fonseca, 2011). Nevertheless, the importance of restoration activities for coastal management has been highlighted by Rudd (2014), with the ocean priority research question addressing ‘restoration effectiveness’ ranking (29th). Restoration programs also provide opportunities to generate significant socio-economic benefits.

However, seagrass restoration costs can be expensive. In the USA, projected costs were estimated at between US$593,000 and US$970,000 (1996 US$) per hectare (author’s conversion) once mapping and ground-truthing, planting, monitoring, contracting and government oversight were included (Fonseca, 2006). In addition, restoration programs suffer from a number of challenges associated with validation (i.e., monitoring), site selection, artificial colonization methods, management processes and lack of adequate scientific knowledge regarding seagrass ecology (Fonseca, 2011). Nonetheless, with respect to restoration program outlays, recent estimates in Australia have suggested somewhat more feasible restoration costs of between AUS$10,000 and AUS$629,000 per hectare, with investments in restoration at the lower end implying pay-back times of 5 years or less (Blandon and zu Ermgassen, 2014). This is further supported by the work of Duarte et al. (2013c), which suggests that due to the value associated with the sequestered carbon restoration programs may be able to recover between US$12,000 and US$43,000 ha^{-1} (constant dollars), enabling the recovery of full program costs where a carbon tax is in place. Furthermore, most restoration programs are likely to be undertaken in developing countries where capital and labour costs are much less prohibitive (Duarte et al. 2013c).

The Swahili Seas Mikoko Pamoja project (2010-2013) provides a successful example of a wetland restoration carbon finance program operating in a developing country context. Active in Gazi Bay, Kenya, the Mikoko Pamoja project has established a mangrove conservation and restoration program focused on the carbon storage value of mangroves to benefit poor coastal communities. The program operates an accredited Plan Vivo carbon credit scheme providing US$13,000 annually, which is disbursed to conservation activities and community development projects. Moreover, since 2012 one of the project partner's, Earthwatch Institute, has employed local residents and volunteers to participate in mangrove management and restoration activities covering over 600 hectares. Finally, the project has also engaged in a number of capacity building initiatives through the provision of additional training and networking facilities (UNEP and CIFOR 2014).

6 Possibilities for implementing seagrass conservation mechanisms

Deciding on the basic operational parameters for a PES program is only half the challenge; the other is to consider how broader institutional and governance elements weave together to influence scheme developments and outcomes: issues that need to be tackled at the design and implementation stage to ensure lasting results (Lin and Nakamura 2012; Lin and Ueta 2012). Collectively, these issues are intimately linked to three of the priority research questions identified by Rudd (2014), namely: 'management capacity of human communities' (ranked 56th), transaction costs of ocean management' (ranked 57th) and 'property rights and conservation' (ranked 66th). Below we identify some of the most salient issues, incorporating insights from REDD+ and coastal resource management. As AGEDI (2012:10) note, 'Blue Carbon and PES project developers have the opportunity to learn from the challenges and successful outcomes from REDD+ projects that feature similar project elements'.

6.1 Institutions

Effective institutions are crucial to the successful implementation of incentive schemes and the resolution of coastal management problems (Rudd et al. 2003; Imperial 2005; Schomers and Matzdorf 2013; Somorin et al. 2014). In the process of establishing effective institutions the development of institutional flexibility is particularly important, as this enables programs to respond adaptively over time to changing circumstances and thus maintain their efficacy (Larson and Soto 2008; Murdiyario et al. 2012). Securing institutional flexibility requires program arrangements that foster active connections and relations between actors, strong leadership and feedbacks in learning systems (Cox et al. 2011; Legrand et al. 2013; Geist and Howlett 2014).

In order to deliver these, programs need to be based on a platform of transparency, accountability and inclusivity (Lockwood et al. 2010; Larsen et al. 2011; Ingram et al. 2014). These aspects function as enabling properties, and the evidence clearly indicates that a lack of transparency and accountability can seriously impair institutional capacity and effectiveness (Somorin et al. 2014), whilst also undermining social capital (Rudd et al. 2003; Shiferaw et al. 2008). In addition, programs that fail to consider the issue of inclusivity can ultimately disempower participant groups, and as a consequence, embed benefit sharing inequalities between households and communities (Krause et al. 2013).

6.2 Stakeholders and Participation

Devolving decision-making to stakeholder groups can be enormously beneficial (Larson and Soto 2008), at once enhancing and strengthening intra-community ties as well as a sense of common identity (Rudd et al. 2003). Conversely, centralized administration can often stifle local-scale innovations and the development of shared visions (Pokorny et al. 2013). Programs need to engage and connect with local stakeholders in order to maximise participation, which is central to providing effective management (Agrawal and Chhatre 2006). Doing so legitimises decision-making and empowers individual and collective agency enabling the design process to align with, and support, local norms, values and beliefs (Kawowski et al. 2011; Brooks et al. 2012; Corbera 2012; Bremer and Glavovic 2013). This is essential for participant commitment (Murdiyario et al. 2012; Davenport and Seekamp 2013) and acknowledges the relevance for effective governance of local users' knowledge (Andersson et al. 2014).

These processes can be supported by clarifying stakeholder roles and responsibilities and promoting leadership (Chhatre et al. 2012; Dent 2012). Leadership, and especially local leadership, has been shown to be fundamental to delivering successful coastal management (Sutton and Rudd 2014). Finally, it is important to acknowledge how participation is framed in the context of power relations, as these can represent potent forces capable of distorting the meaningful involvement, agency and legitimacy of grassroots actors (Dewulf et al. 2011; Cook et al. 2013).

6.3 Tenure and property rights

Ownership in developing countries is often complicated by overlapping formal and informal (customary) tenure and rights-based arrangements (Awono et al 2014; Resosudarmo et al 2014; Rights and Resources 2014; Sunderlin et al. 2014). Clearly defining, legitimising and enabling functioning property rights systems is essential for operationalizing incentive programs (Lockie 2013). Such clarifications are critical for conditional payments where knowing who to pay (i.e., the right holder) and who is accountable for delivering project-level outcomes is necessary (Visseren-Hamakers et al. 2012; Duchelle et al. 2014; Sunderlin et al. 2014). Functioning tenure and rights-based systems also provide the framework to enforce property rights, securing contracts (Naughton-Treves and Wendland 2014) and combating weak governance (Resosudarmo et al. 2014).

This is particularly pertinent to coastal marine environments where complications concerning tenure, rights designations and authority are a direct challenge to introducing and enforcing incentive schemes (Mohammed 2012), a state of affairs clearly linked to the ambiguities regarding property rights in coastal areas and the variety of users and user interests (Cicin-Sain, 1993). As part of the design process it is crucial to mitigate potential mismatches arising between the provision, delivery and bundle of property rights to reduce the likelihood of marine resource conflicts developing (Yandle 2007), as well as to ensure that poorer sectors are not marginalised or power asymmetries and social inequalities reinforced (WRI 2005; Fisher et al. 2008).

6.4 Benefit sharing

Distributing benefits and costs in a fair and equitable way is a fundamental aspect of delivering socially acceptable incentive schemes (McDermott et al. 2012). Traditionally, equity concerns have been side-lined in favour of a greater emphasis and focus on efficiency maximization (Pascual et al. 2010; Narloch et al. 2013). However, this trade-off can produce socially undesirable outcomes (Asquith et al. 2008). Incorporating social parameters in the targeting of schemes in order to widen access and participation whilst reducing the marginalization of poorer communities represents an important first step in reversing these potential trade-offs (Mahanty et al. 2011). These processes need to proceed in tandem with beneficiary identification and the evaluation of the potential socio-economic ramifications of ES provision and distribution (Willemen et al. 2013). Additional considerations for effective benefit sharing include legitimising decision-making processes via legal and procedural avenues (Murdiyario et al. 2012); adjusting compensation levels according to the capacity needs of individuals, households and communities (Mohammed 2012); and addressing the potential socio-economic impacts of programs on non-participants (Huang et al. 2009).

6.5 Delivering ecosystem services, monitoring and compliance

The central tenant of incentive schemes relates the provision of specified outputs to agreement obligations and payments (Ferraro 2008; Wunder et al. 2008). Consequently, monitoring and compliance represent key contractual conditions for programs to deliver their principal objectives (Hejnowicz et al. 2014). These can be distilled into four broad areas:

First, measuring ES provision (Porras et al. 2013). This reduces the likelihood of producing a false picture of service provision, and provides a scientifically robust case for PES program design (Hejnowicz et al. 2014). It has been suggested that even though coastal systems may be data poor, there is sufficient knowledge of the management activities that improve resource protection and ES provision (Lau, 2013). Second, evaluating scheme additionality and demonstrating 'added value' by addressing the links between management interventions and program delivery (Ghazoul et al. 2010). Validating additionality requires baseline data, suitable metrics and performance indicators plus the targeting of PES to locations likely to maximize program benefits (Sommerville et al. 2011; Wünscher and Engel, 2012; Lau, 2013).

Third, assessing potential of spill-over effects (i.e., leakage) resulting from program implementation that may offset additionality gains (Engel et al. 2008; Porras et al. 2013). Fourth, monitoring contract conditionality and ensuring compliance (Ferraro 2008). This requires establishing who is monitoring (i.e. users, communities or officials) and how frequently (Sommerville et al. 2011), providing sufficient payments to programme participants (Porras et al. 2013), and ensuring agreements are long-term arrangements with enforceable penalties for breaches of contract (Ferraro 2008; Wunder et al. 2008). All have substantive effects on transaction costs of governance (ranked 57th, Rudd 2014) and will influence the long-term viability of PES structures.

6.6 Costs and funding

The viability of PES programs relies upon consistent and sufficient financial flows, both in the short-term (i.e., covering costs needed to initiate and implement a project) and the long-term (i.e., securing the funds necessary to sustain an active project), without which lasting transformative change cannot be achieved (Hejnowicz et al. 2014). Programs need to be designed so that they sustain themselves through self-generated revenues (Pirard et al. 2010). An added complication for seagrass PES schemes is that monitoring and enforcement in marine and coastal environments may require extra technical and specialist equipment not needed in the terrestrial sphere, adding significantly to program outlays (Lau, 2013). Securing long-term funding that reduces fiscal constraints but is not overly reliant on external donor funding is particularly important (Bennett et al. 2013; Fauzi and Anna 2013; Hein et al. 2013). Achieving both these objectives requires adequately accounting for the full range of transaction costs, which in some cases may be prohibitive for PES development (McCann et al. 2005; Marshall 2013; McCann 2013).

7 Conclusions

Seagrass ecosystems provide an array of globally and locally significant ecosystem services. From the perspective of climate change, it is their carbon sequestration and storage potential that is most attractive. Seagrass ecosystems are also home to diverse marine life that can directly or indirectly support the artisanal and commercial fisheries that help maintain resilience in human communities. In addition, they also play an important role in the conservation and maintenance of marine biological diversity and influence national or international non-market benefits deriving from endangered species such as sea turtles (Rudd 2009). We have examined the prospects for financing seagrass conservation under a purely carbon approach and in conjunction with PES schemes that could help capture the benefits derived from multiple ecosystem services beyond carbon sequestration.

The prospects for developing a pure carbon credit scheme remain slim, especially if targeted at the regulatory carbon market. Opportunities exist, however, for voluntary carbon market schemes and these are far more promising. However, the instability of the voluntary carbon market and the impact this has on carbon prices makes a purely carbon-based approach questionable; fluctuating carbon prices mean projects cannot guarantee financial returns on investment or adequate payments to meet participants' needs. Nonetheless, voluntary carbon standards are channelling more effort into delivering co-benefits and, from this perspective, seagrass PES schemes may be highly complementary. Adopting a combined strategy would maximize conservation and livelihood outcomes so long as the design, implementation and institutional issues previously highlighted were adequately dealt with.

Providing the scientific evidence base for complex incentive schemes is challenging. This is particularly so with Blue Carbon systems where there remain many ecological, social and economic knowledge gaps that need to be negotiated in order to develop functional payment programs. However, we have mapped out what those potential knowledge gaps are in relation to seagrass ecosystems, in terms of basic ecosystem function-service information, ecosystem service valuation and research concerning the governance structures and apparatus through which incentive schemes would need to operate. In so doing we have highlighted the importance and complexity of seagrass ecosystems and the value of conserving them. At the same time we have clearly identified how by conserving these systems, particularly through the use of innovative financial incentive mechanisms, we are also contributing to a broader set of significant global ocean priority research challenges.

Overall, a wide range of opportunities exist for including seagrass meadows in local PES schemes to combat climate change, secure seagrass conservation and enhance coastal community development. However, realizing the 'true' potential of seagrass meadows requires international cooperation on two fronts: combating the threats that currently imperil the integrity of functioning seagrass ecosystems and including them in formal climate change policies such as REDD+. In this respect challenges and barriers remain but promising progress is being made; efforts to protect and rehabilitate seagrass ecosystems are crucial because of their widespread distribution, their central role in supporting functional coastal environments and the human communities that rely on those systems.

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9 References

AGEDI. (2014). Building blue carbon projects: an introductory guide. AGEDI/EAD. Published by AGEDI. (86 pp.)

Agrawal, A., Chhatre, A. (2006). Explaining success on the commons: community forest governance in the Indian Himalaya. World Development 34, 149–166. doi:10.1016/j.worlddev.2005.07.013

Albers, H.J., Robinson, E.J.Z., January, S.K. (2012). Managing marine protected areas through incentives to local people: the case of Mnazi Bay Ruvuma Estuary Marine Park. Environment for Development Initiative, University of Gothenburg, Sweden. (5 pp.)

Andersson, K., Benavides, J.P., León, R. (2014). Institutional diversity and local forest governance. Environmental Science and Policy 36, 61–72. doi:10.1016/j.envsci.2013.07.009

Aquith, N.M.; Vargas, M.V. & Wunder, S. (2008). Selling two environmental services: in-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. Ecological Economics 65, 675-684. doi 10.1016/j.ecolecon.2007.12.014

Awono, A., Somorin, O. A., Eba'a Atyi, R., Levang, P. (2014). Tenure and participation in local REDD+ projects: insights from southern Cameroon. Environmental Science and Policy 35, 76–86. doi:10.1016/j.envsci.2013.01.017

Baker, S., Paddock, J., Smith, A.M., Unsworth, R.K.F., Cullen-Unsworth, L.C., Hertler, H. (2014). An ecosystems perspective for food security in the Caribbean: seagrass meadows in the Turks and Caicos Islands. Ecosystem Services 1–10. doi:10.1016/j.ecoser.2014.07.011

Baldwin, K., Mahon, R., McConney, P. (2013). Participatory GIS for strengthening transboundary marine governance in SIDS. Natural Resources Forum 37, 257–268. doi:10.1111/1477-8947.12029

Baldwin, K., Oxenford, H. A. (2014). A participatory approach to marine habitat mapping in the Grenadine Islands. Coastal Management 42, 36–58. doi:10.1080/08920753.2013.863725

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J.H. et al. (2008). Vegetation's role in coastal protection. Science 320, 176-177. doi: 10.1126/science.320.5873.176b

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R. (2011). The value of estuarine and coastal ecosystem services. Ecological Monographs 81, 169–193. doi:10.1890/10-1510.1

Barnes, R. (2013). Spatial stability of macrobenthic seagrass biodiversity. Marine Ecology Progress Series 493, 127–139. doi:10.3354/meps10546

Begossi, A., May, P.H., Lopes, P.F., Oliveira, L.E.C., da Vinha, V., Silvano, R. A. M. (2011). Compensation for environmental services from artisanal fisheries in SE Brazil: policy and technical strategies. Ecological Economics 71, 25–32. doi:10.1016/j.ecolecon.2011.09.008

Begossi, A., Salyvonchyk, S., Nora, V., Lopes, P.F., Silvano, R. A. M. (2012). The Paraty artisanal fishery (southeastern Brazilian coast): ethnoecology and management of a social-ecological system (SES). Journal of Ethnobiology and Ethnomedicine 8, 22. doi:10.1186/1746-4269-8-22

Bell, J.D., Pollard, D.A. (1989). Ecology of fish assemblages and fisheries associated with seagrasses. In: Larkum, A.W.D., McComb, A.J., Shepherd, S.A. (Eds.), Biology of Seagrasses a Treatise on the Biology of Seagrasses with Special Reference to the Australian Region. Elsevier, Amsterdam, pp. 565-609.

Benessaiah, K. (2012). Carbon and livelihoods in post-Kyoto: assessing voluntary carbon markets. Ecological Economics 77, 1–6. doi:10.1016/j.ecolecon.2012.02.022

Benjamin, C.E. (2008). Legal pluralism and decentralization: natural resource management in Mali. World Development 36, 2255–2276. doi:10.1016/j.worlddev.2008.03.005

Bennett, M.T. (2008). China's sloping land conversion program: institutional innovations or business as usual? Ecological Economics 65, 699-711. doi: 10.1016/j.ecolecon.2007.09.017

Bianco, N. (2009). Stacking payments for ecosystem services. World Resources Institute Fact Sheet, Washington, DC. (4 pp.)

Blandon, A., zu Ermgassen, P.S.E. (2014). Quantitative estimate of commercial fish enhancement by seagrass habitat in southern Australia. Estuarine and Coastal Shelf Science 141, 1–8. doi:10.1016/j.ecss.2014.01.009

Boer, W.F. (2007). Seagrass–sediment interactions, positive feedbacks and critical thresholds for occurrence: a review. Hydrobiologia 591, 5–24. doi:10.1007/s10750-007-0780-9

Bond, I., Mayers, J. (2010). Fair deals for watershed services. Natural Resource Issues No. 13. International Institute for Environment and Development, London, UK. (122 pp.)

Bos, A.R., Bouma, T.J., de Kort, G.L.J., van Katwijk, M.M. (2007). Ecosystem engineering by annual intertidal seagrass beds: sediment accretion and modification. Estuarine Coastal Shelf Science. 74, 344–348. doi:10.1016/j.ecss.2007.04.006

Bosire, J.O., Okemwa, G., Ochiewo, J. (2012). Mangrove linkages to coral reef and seagrass ecosystem services in Mombasa and Takaungu, Kenya. Ecosystem Services Poverty Alleviation Programme, UK. (45 pp.)

Bosselmann, A.S., Lund, J.F. (2013). Do intermediary institutions promote inclusiveness in PES programs? The case of Costa Rica. Geoforum 49, 50–60. doi:10.1016/j.geoforum.2013.05.009

Bottema, M.J.M, Bush, S.R. (2012). The durability of private sector marine conservation: a case study of two entrepreneurial marine protected areas in Indonesia. Ocean and Coastal Management 61, 38-48. doi: 10.1016/j.ocecoaman.2012.01.004

Brander, L., J. Wagtendonk, A., S. Hussain, S., McVittie, A., Verburg, P.H., de Groot, R.S. et al. (2012). Ecosystem service values for mangroves in Southeast Asia: a meta-analysis and value transfer application. Ecosystem Services 1, 62–69. doi:10.1016/j.ecoser.2012.06.003

Brander, L., Brouwer, R., Wagtendonk, A. (2013). Economic valuation of regulating services provided by wetlands in agricultural landscapes: a meta-analysis. Ecological Engineering 56, 89–96. doi:10.1016/j.ecoleng.2012.12.104

Bredbenner, A. (2013). Profiles in Blue Carbon Field Work. A Report Commissioned by Conservation International. (23 pp.)

Bremer, S., Glavovic, B. (2013). Mobilizing knowledge for coastal governance: re-framing the science–policy interface for integrated coastal management. Coastal Management 41, 39–56. doi:10.1080/08920753.2012.749751

Brondizio, E. S., F. Gatzweiler, C. Zagrafos, M. Kumar. (2010). Socio-cultural context of ecosystem and biodiversity valuation.(Chapter 4) In P. Kumar (ed.) The Economics of Ecosystems and Biodiversity (TEEB). United Nations Environmental Programme and the European Commission. London, UK: Earthscan Press. (pp. 150-181).

Brooks, J.S., Waylen, K. a, Borgerhoff Mulder, M. (2012). How national context, project design, and local community characteristics influence success in community-based conservation projects. Proceedings of the National Academy of Sciences U. S. A. 109, 21265–70. doi:10.1073/pnas.1207141110

Browne, C., Milne, R., Griffiths, C., Bolton, J., Anderson, R. (2013). Epiphytic seaweeds and invertebrates associated with South African populations of the rocky shore seagrass

Thalassodendron leptocaule — a hidden wealth of biodiversity. African Journal of Marine Science 35, 523–531. doi:10.2989/1814232X.2013.864332

Bujang, J.S., Zakaria, M.H., Arshad, A. (2006). Distribution and significance of seagrass ecosystems in Malaysia. Aquatic Ecosystem Health & Management 9, 203–214. doi:10.1080/14634980600705576

Cabaço, S., Apostolaki, E.T., García-Marín, P., Gruber, R., Hernández, I., Martínez-Crego, B. et al. (2013). Effects of nutrient enrichment on seagrass population dynamics: evidence and synthesis from the biomass-density relationships. Journal of Ecology 101, 1552–1562. doi:10.1111/1365-2745.12134

Camacho-Valdez, V., Ruiz-Luna, A., Ghermandi, A., Nunes, P. A. L.D. (2013). Valuation of ecosystem services provided by coastal wetlands in northwest Mexico. Ocean & Coastal Management 78, 1–11. doi:10.1016/j.ocecoaman.2013.02.017

Chhatre, A., Lakhanpal, S., Larson, A.M., Nelson, F., Ojha, H., Rao, J. (2012). Social safeguards and co-benefits in REDD+: a review of the adjacent possible. Current Opinion in Environmental Sustainability 4, 654–660. doi:10.1016/j.cosust.2012.08.006

Chiu, S.-H., Huang, Y.-H., Lin, H.-J. (2013). Carbon budget of leaves of the tropical intertidal seagrass Thalassia hemprichii. Estuarine & Coastal Shelf Science 125, 27–35. doi:10.1016/j.ecss.2013.03.026

Christianen, M.J. A, van Belzen, J., Herman, P.M.J., van Katwijk, M.M., Lamers, L.P.M., van Leent, P.J.M. et al. (2013). Low-canopy seagrass beds still provide important coastal protection services. PLoS One 8, e62413. doi:10.1371/journal.pone.0062413

Christie, M., Fazey, I., Cooper, R., Hyde, T., Kenter, J.O. (2012). An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. Ecological Economics 83, 67–78. doi:10.1016/j.ecolecon.2012.08.012

Cicin-Sain, B. (1993). Sustainable development and integrated coastal management. Ocean & Coastal Management 21, 11-43 doi:10.1016/0964-5691(93)90019-U

Cook, B.R., Kesby, M., Fazey, I., Spray, C. (2013). The persistence of “normal” catchment management despite the participatory turn: exploring the power effects of competing frames of reference. Social Studies of Science 43, 754–779. doi:10.1177/0306312713478670

Cooley, D., Olander, L. (2012). Stacking ecosystem services payments: risks and solutions. Environmental Law Reporter, 42, 10150–10165.

Corbera, E. (2012). Problematizing REDD+ as an experiment in payments for ecosystem services. Current Opinion in Environmental Sustainability 4, 612–619. doi:10.1016/j.cosust.2012.09.010

Cox, M., Arnold, G., Tomás, S.V. (2010). A review of design principles for community-based natural resource management. Ecology & Society 15(4), 19.

Cullen-Unsworth, L., Unsworth, R. (2013). Seagrass meadows, ecosystem services, and sustainability. Environmental Science and Policy for Sustainable Development 55, 14–28. doi:10.1080/00139157.2013.785864

Cullen-Unsworth, L.C., Nordlund, L.M., Paddock, J., Baker, S., McKenzie, L.J., Unsworth, R.K.F. (2013). Seagrass meadows globally as a coupled social-ecological system: implications for human wellbeing. Marine Pollution Bulletin doi:10.1016/j.marpolbul.2013.06.001

Danielsen, F., Adrian, T., Brofeldt, S. (2013). Community monitoring for REDD+: international promises and field realities. Ecology & Society. 18 (3): 41. http://dx.doi.org/10.5751/ES-05464-180341

Davenport, M. A., Seekamp, E. (2013). A multilevel community capacity model for sustainable watershed management. Society & Natural Resources 26, 1101–1111. doi:10.1080/08941920.2012.729650

De la Torre-Castro, M., Di Carlo, G., Jiddawi, N.S. (2014). Seagrass importance for a small-scale fishery in the tropics: The need for seascape management. Marine Pollution Bulletin doi:10.1016/j.marpolbul.2014.03.034

De Koning, F.; Aguiňaga, M.; Bravo, M.; Chiv, M.; Lascano, M.; Lozada, T. et al. (2011). Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. Environmental Science & Policy 14, 531-542. doi:10.1016/j.envsci.2011.04.007

Deal, R.L., Cochran, B., LaRocco, G. (2012). Bundling of ecosystem services to increase forestland value and enhance sustainable forest management. Forest Policy & Economics 17, 69–76. doi:10.1016/j.forpol.2011.12.007

Dekker, A.G., Brando, V.E., Anstee, J.M. (2005). Retrospective seagrass change detection in a shallow coastal tidal Australian lake. Remote Sensing of Environment 97, 415–433. doi:10.1016/j.rse.2005.02.017

Dendoncker, N., Keune, H., Jacobs, S., Gómez-Baggethun, F. (2014). Inclusive ecosystem service valuation. In Ecosystem Services: Global Issues, Local Practices. 2014. (Eds). Jacobs, S., Dendoncker, N. and Keune, H. Elsevier Inc. USA. pp3-12

Dent, M. (2012). Catchment management agencies as crucibles in which to develop responsible leaders in South Africa. Water SA 38, 313–326. http://dx.doi.org/10.4314/wsa.v38i2.17

Derissen, S., Latacz-Lohmann, U. (2013). What are PES? A review of definitions and an extension. Ecosystem Services 6, 12-15.doi:10.1016/j.ecoser.2013.02.002

Dewulf, A., Mancero, M., Cardenas, G., Sucozhanay, D. (2011). Fragmentation and connection of frames in collaborative water governance: a case study of river catchment management in Southern Ecuador. International Review of Administrative Sciences 77, 50–75. doi:10.1177/0020852310390108

Duarte, C.M., Kennedy, H., Marbà, N., Hendriks, I. (2013a). Assessing the capacity of seagrass meadows for carbon burial: Current limitations and future strategies. Ocean & Coastal Management 83, 32–38. doi:10.1016/j.ocecoaman.2011.09.001

Duarte, C.M., Losada, I.J., Hendriks, I.E., Mazarrasa, I., Marbà, N. (2013b). The role of coastal plant communities for climate change mitigation and adaptation. Nature Climate Change 3, 961–968. doi:10.1038/nclimate1970

Duarte, C.M., Sintes, T., Marbá, N. (2013c). Assessing the CO_2 capture potential of seagrass restoration projects. Journal of Applied Ecology 50, 1341-1349. doi: 10.1111/1365-2664.12155

Duarte, C.M., Marbà, N., Gacia, E., Fourqurean, J.W., Beggins, J., Barrón, C. et al. (2010). Seagrass community metabolism: assessing the carbon sink capacity of seagrass meadows. Global Biogeochemcal Cycles 24, n/a–n/a. doi:10.1029/2010GB003793

Duarte, C.M. (2002). The future of seagrass meadows. Environmental Conservation 29, 192-206. http://dx.doi.org/10.1017/S0376892902000127

Duarte, C.M., Chiscano, C.L. (1999). Seagrass biomass and production: a reassessment. Aquatic Botany 65, 159-174. http://dx.doi.org/10.1016/S0304-3770(99)00038-8

Duchelle, A.E., Cromberg, M., Gebara, M.F., Guerra, R., Melo, T., Larson, A. et al. (2014). Linking forest tenure reform, environmental compliance, and incentives: lessons from REDD+ initiatives in the Brazilian Amazon. World Development 55, 53–67. doi:10.1016/j.worlddev.2013.01.014

Ducrot, R., Bueno, A. K., Barban, V., Reydon, B.P. (2010). Integrating land tenure, infrastructure and water catchment management in Sao Paulo's periphery: lessons from a gaming approach. Environment and Urbanisation 22, 543–560. doi:10.1177/0956247810380168

Engel, S., Pagiola, S., Wunder, S. (2008). Designing payments for environmental services in theory and practice: an overview of the issues. Ecological Economics 65, 663–674. doi:10.1016/j.ecolecon.2008.03.011

Erftemeijer, P.L. a, Lewis, R.R.R. (2006). Environmental impacts of dredging on seagrasses: a review. Marine Pollution Bulletin 52, 1553–72. doi:10.1016/j.marpolbul.2006.09.006

Erftemeijer, P., Shuail, D. (2012). Seagrass habitats in the Arabian Gulf: distribution, tolerance thresholds and threats. Aquatic Ecosystem Health & Management 15, sup 1, 37–41.

Farina, S., Tomas, F., Prado, P., Romero, J., Alcoverro, T. (2009). Seagrass meadow structure alters interactions between the sea urchin Paracentrotus lividus and its predators. Marine Ecology Progress Series 377, 131–137. doi:10.3354/meps07692

Farley, J., Costanza, R. (2010). Payments for ecosystem services: from local to global. Ecological Economics 69, 2060–2068.doi:10.1016/j.ecolecon.2010.06.010

Fauzi, A., Anna, Z. (2013). The complexity of the institution of payment for environmental services: A case study of two Indonesian PES schemes. Ecosystem Services 6, 54–63. doi:10.1016/j.ecoser.2013.07.003

Ferraro, P.J. (2008). Asymmetric information and contract design for payments for environmental services. Ecological Economics 65, 810–821. doi:10.1016/j.ecolecon.2007.07.029

Ferraro, P.J. (2009). A global survey of sea turtle payment incentive programs. National Oceanic and Atmospheric Administration, La Jolla, California. pp1-40. (Order No:AB133F04SE1183)

Fisher, R., Maginnis, S., Jackson, W., Barrow, E., Jeanrenaud, S. (2008). Linking conservation and poverty reduction: landscapes, people and power. Earthscan. London

Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K., Burgess, N.D. (2010). Common pool resource management and PES: lessons and constraints for water PES in Tanzania. Ecological Economics 69, 1253–1261. doi:10.1016/j.ecolecon.2009.11.008

Fonseca, M.S. (2011). Addy revisited: what has changed with seagrass restoration in 64 years? Ecological Restoration 29, 73-81. doi: 10.1353/ecr.2011.0020

Fonseca, M.S. (2006). Wrap-up of seagrass restoration: success, failure and lessons about the costs of both. In Seagrass restoration: success, failure and the costs of both. 2006. (Eds) Treat, S.F. & Lewis III, R.R. Lewis Environmental Services, Inc. (pp. 169-175)

Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A. et al. (2012). Seagrass ecosystems as a globally significant carbon stock. Nature Geosciences 5, 505–509. doi:10.1038/ngeo1477

Freeman, A. S., Short, F.T., Isnain, I., Razak, F. A., Coles, R.G. (2008). Seagrass on the edge: land-use practices threaten coastal seagrass communities in Sabah, Malaysia. Biological Conservation 141, 2993–3005. doi:10.1016/j.biocon.2008.09.018

Garbach, K., Lubell, M., DeClerck, F. A. J. (2012). Payment for Ecosystem Services: The roles of positive incentives and information sharing in stimulating adoption of silvopastoral conservation practices. Agriculture Ecosystems & Environment 156, 27–36. doi:10.1016/j.agee.2012.04.017

Ghazoul, J., Butler, R. A, Mateo-Vega, J., Koh, L.P. (2010). REDD: a reckoning of environment and development implications. Trends in Ecology & Evolution 25, 396–402. doi:10.1016/j.tree.2010.03.005

Giest, S., Howlett, M. (2014). Understanding the pre-conditions of commons governance: the role of network management. Environmental Science & Policy 36, 37–47. doi:10.1016/j.envsci.2013.07.010

Gómez-Baggethun, E., de Groot, R., Lomas, P. L., & Montes, C. (2010). The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. Ecological Economics 69, 1209–1218. doi:10.1016/j.ecolecon.2009.11.007

Gowdy, J., Baveye, P.C. (2014). Monetary valuation of ecosystem services: unresolvable problems with the standard economic model. In Ecosystem Services: Global Issues, Local Practices. 2014. (Eds.) Jacobs, S., Dendoncker, N. & Keune, H. Elsevier Inc. USA. (pp. 73-77)

Greenhalgh, S. (2008). Bundled ecosystem markets - are they the future? In: American Agricultural Economics Association Annual Meeting. Orlando, Florida, USA. (28 pp.)

Green, E.P., Short, F.T. (2003). World Atlas of Seagrasses. University of California Press, Berkeley.

Greiner, J.T., McGlathery, K.J., Gunnell, J., McKee, B. A. (2013). Seagrass restoration enhances "blue carbon" sequestration in coastal waters. PLoS One 8, e72469. doi:10.1371/journal.pone.0072469

Greiner, R., Stanley, O. (2013). More than money for conservation: exploring social co-benefits from PES schemes. Land Use Policy 31, 4–10. doi:10.1016/j.landusepol.2011.11.012

Grimsditch, G. (2011). Options for blue carbon within the international climate change framework. Sustainable Development Law & Policy 11, 22–24.

Groom, B., Palmer, C. (2012). REDD+ and rural livelihoods. Biological Conservation 154, 42–52. doi:10.1016/j.biocon.2012.03.002

Gullström, M., Bodin, M., Nilsson, P.G., Öhman, M.C. (2008). Seagrass structural complexity and landscape configuration as determinants of tropical fish assemblage composition. Marine Ecology Progress Series 363, 241-255.

Gullström, M., Lundén, B., Bodin, M., Kangwe, J., Öhman, M.C., Mtolera, M.S.P. et al. (2006). Assessment of changes in the seagrass-dominated submerged vegetation of tropical Chwaka Bay (Zanzibar) using satellite remote sensing. Estuarine & Coastal Shelf Science 67, 399–408. doi:10.1016/j.ecss.2005.11.020

Hansen, J.C.R., Reidenbach, M. A. (2013). Seasonal growth and senescence of a Zostera marina seagrass meadow alters wave-dominated flow and sediment suspension within a coastal bay. Estuaries and Coasts 36, 1099–1114. doi:10.1007/s12237-013-9620-5

Heck, K.L., Carruthers, T.J.B., Duarte, C.M., Hughes, A. R., Kendrick, G., Orth, R.J. (2008). Trophic transfers from seagrass meadows subsidize diverse marine and terrestrial consumers. Ecosystems 11, 1198–1210. doi:10.1007/s10021-008-9155-y

Hedge, R. & Bull, G.Q. (2011). Performance of an agro-forestry based payments for environmental services project in Mozambique: a household level survey. Ecological Economics 71, 122-130. doi:10.1016/j.ecolecon.2011.08.014

Hein, L., Miller, D.C., de Groot, R. (2013). Payments for ecosystem services and the financing of global biodiversity conservation. Current Opinion in Environmental Sustainability 1–7. doi:10.1016/j.cosust.2012.12.004

Hejnowicz, A.P., Raffaelli, D.G., Rudd, M.A., White, P.C.L. (2014). Evaluating the outcomes of payments for ecosystem service programmes using a capital asset framework. Ecosystem Services 9, 83-97. doi: 10.1016/j.ecoser.2014.05.001

Hemming, M.A., Duarte, C.M. (2000). Seagrass Ecology. Cambridge University Press

Hendriks, I., Sintes, T., Bouma, T., Duarte, C. (2008). Experimental assessment and modeling evaluation of the effects of the seagrass Posidonia oceanica on flow and particle trapping. Marine Ecology Progress Series 356, 163–173. doi:10.3354/meps07316

Herr, D., Pidgeon, E., Laffoley, D. (2012). Blue Carbon Policy Framework 2.0: based on the discussion of the International Blue Carbon Policy Working Group. Gland, Switzerland: IUCN and Arlington, USA: CI.

Hicks, C.C., McClanahan, T.R. (2012). Assessing gear modifications needed to optimize yields in a heavily exploited, multi-species, seagrass and coral reef fishery. PLoS One 7, e36022. doi:10.1371/journal.pone.0036022

Honda, K., Nakamura, Y., Nakaoka, M., Uy, W.H., Fortes, M.D. (2013). Habitat use by fishes in coral reefs, seagrass beds and mangrove habitats in the Philippines. PLoS One 8, e65735. doi:10.1371/journal.pone.0065735

Hughes, A.R., Williams, S.L., Duarte, C.M., Heck Jr., K.L., Waycott, M. (2009). Associations of concern: declining seagrasses and threatened dependent species. Frontiers in Ecology & Environment 7, 242-246.

Imperial, M.T. (2005). Using collaboration as a governance strategy: lessons from six watershed management programs. Administration & Society 37, 281–320. doi:10.1177/0095399705276111

Ingram, J.C. (2012). Bundling and stacking for maximizing social, ecological and economic benefits: a Framing paper for discussion at the “Bundling and Stacking Workshop.” Available at: http://rmportal.net/library/content/translinks/translinks-2012/Wildlife%20Conservation%20Society/2012%20PES%20Bundling%20and%20Stacking%20Workshop%20(Washington,%20DC,%20USA)/Paper_PESBundlingandStacking.pdf/at_download/file (accessed 15/08/2014)

Ingram, J.C., Wilkie, D., Clements, T., McNab, R.B., Nelson, F., Baur, E.H. (2014). Evidence of payments for ecosystem services as a mechanism for supporting biodiversity conservation and rural livelihoods. Ecosystem Services 7, 10–21. doi:10.1016/j.ecoser.2013.12.003

IPCC (2014) 2013 Supplement to the 2006 IPCC Guidelines for Natural Greenhouse Gas Inventories: Wetlands. (Eds.) Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. & Troxler, T. G. Published: IPCC, Switzerland. (354 pp.)

Jackson, EL; Rowden, AA; Attrill, MJ; et al. (2001). The importance of seagrass beds as a habitat for fishery species. Oceanography and Marine Biology 39, 269-303

James, G.K., Adegoke, J.O., Osagie, S., Ekechukwu, S., Nwilo, P., Akinyede, J. (2013). Social valuation of mangroves in the Niger Delta region of Nigeria. International Journal of Biodiversity Science, Ecosystem Service & Management 9, 311–323. doi:10.1080/21513732.2013.842611

Jaxion-Harm, J., Saunders, J., Speight, M.R. (2012). Distribution of fish in seagrass, mangroves and coral reefs: life-stage dependent habitat use in Honduras. Revista de Biologia Tropical 60, 683–98.

Johnston, R. J., Besedin, E. Y., Iovanna, R., Miller, C. J., Wardwell, R. F. & Ranson, M. H. (2005). Systematic variation in willingness to pay for aquatic resource improvements and implications for benefit transfer: a meta-analysis. Canadian Journal of Agricultural Economics 53(2-3), 221-248. doi: 10.1111/j.1744-7976.2005.04018.x

Kaczan, D., Swallow, B.M., Adamowicz, W.L. (Vic). (2013). Designing a payments for ecosystem services (PES) program to reduce deforestation in Tanzania: An assessment of payment approaches. Ecological Economics 95, 20–30. doi:10.1016/j.ecolecon.2013.07.011

Kakuru, W., Turyahabwe, N., Mugisha, J. (2013). Total economic value of wetlands products and services in Uganda. Scientific World Journal. Volume 2013, Article ID 192656, 13 pages doi:10.1155/2013/192656

Kamimura, Y., Kasai, A., Shoji, J. (2011). Production and prey source of juvenile black rockfish Sebastes cheni in a seagrass and macroalgal bed in the Seto Inland Sea, Japan: estimation of the economic value of a nursery. Aquatic Ecology 45, 367–376. doi:10.1007/s10452-011-9360-1

Kauffman, C.M. (2013). Financing watershed conservation: lessons from Ecuador's evolving water trust funds. Agricultural Water Management doi:10.1016/j.agwat.2013.09.013

Kennedy, H., Beggins, J., Duarte, C.M., Fourqurean, J.W., Holmer, M., Marbà, N. (2010). Seagrass sediments as a global carbon sink: Isotopic constraints. Global Biogeochemical Cycles 24, n/a–n/a. doi:10.1029/2010GB003848

Kibicho, W. (2008). Community-based tourism: a factor-cluster segmentation approach. Journal of Sustainable Tourism 16, 211–231. doi:10.2167/jost623.0

Kirwan, M. L. & Megonigal, J. P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504(7478): 53-60.

Koch, V., Nichols, W.J., Peckham, H., de la Toba, V. (2006). Estimates of sea turtle mortality from poaching and bycatch in Bahía Magdalena, Baja California Sur, Mexico. Biological Conservation 128, 327–334. doi:10.1016/j.biocon.2005.09.038

Kolinjivadi, V., Sunderland, T. (2012). A review of two payment schemes for watershed services from China and Vietnam: the interface of government control and PES theory. Ecology & Society 17 (4).10. http://dx.doi.org/10.5751/ES-05057-170410

Kosoy, N.; Martinez-Tuna, M.; Muradian, R. & Martínez-Alier, J. (2007). Payments for environmental services in watershed: insights from a comparative study of three cases in Central America. Ecological Economics 61, 446-455. doi:10.1016/j.ecolecon.2006.03.016

Krause, T., Loft, L. (2013). Benefit distribution and equity in Ecuador's Socio Bosque program. Society & Natural Resources 26, 1170-1184. doi:10.1080/08941920.2013.797529

Krause, T., Collen, W., Nicholas, K.A. (2013). Evaluating safeguards in a conservation incentive program: participation, consent, and benefit sharing in indigenous communities of the Ecuadorian Amazon. Ecology & Society 18, 4:1 http://dx.doi.org/10.5751/ES-05733-180401

Kumar, R., Horwitz, P., Milton, G.R., Sellamuttu, S.S., Buckton, S.T., Davidson, N.C. (2011). Assessing wetland ecosystem services and poverty interlinkages: a general framework and case study. Hydrological Sciences Journal 56, 1602–1621. doi:10.1080/02626667.2011.631496

Laffoley, D. d'a, (2013). The management of coastal carbon sinks in Vanuatu: realising the potential. Commonwealth Secretariat, London, UK. (50 pp.)

Landell-Mills, N., Porras, I.T. (2002). Silver bullet or fools' gold? A global review of markets for environmental services and their impact on the poor. International Institute for Environment and Development, London. (pp. 272)

Larsen, R.K., Acebes, J.M., Belen, A. (2011). Examining the assumptions of integrated coastal management: stakeholder agendas and elite co-option in Babuyan Islands, Philippines. Ocean & Coastal Management 54, 10–18. doi:10.1016/j.ocecoaman.2010.10.007

Larson, A.M., Soto, F. (2008). Decentralization of natural resource governance regimes. Annual Review of Environment & Resources 33, 213–239. doi:10.1146/annurev.environ.33.020607.095522

Lau, W.W.Y. (2013). Beyond carbon: conceptualising payments for ecosystem services in blue forests on carbon and other marine and coastal ecosystem services. Ocean & Coastal Management 83, 5-14. doi: 10.1016/j.ocecoaman.2012.03.011

Lavery, P.S., Mateo, M.-Á., Serrano, O., Rozaimi, M. (2013). Variability in the carbon storage of seagrass habitats and its implications for global estimates of blue carbon ecosystem service. PLoS One 8, e73748. doi:10.1371/journal.pone.0073748

Legrand, T., Froger, G., Le Coq, J.-F. (2013). Institutional performance of payments for environmental services: an analysis of the Costa Rican program. Forest Policy & Economics 37, 115–123. doi:10.1016/j.forpol.2013.06.016

Liekens, I., De Nocker, L. (2014). Valuation of ES: Challenges and policy use. In Ecosystem Services: Global Issues, Local Practices. 2014. (Eds.) Jacobs, S., Dendoncker, N. & Keune, H. Elsevier Inc. USA. pp. 107-119

Lin, H., Nakamura, M., (2012). Payments for watershed services: directing incentives for improving lake basin governance. Lakes & Reservoirs Research & Management 17, 191–206. doi:10.1111/lre.12004

Lin, H., Ueta, K. (2012). Lake watershed management: services, monitoring, funding and governance. Lakes & Reservoirs Research & Management Lakes 17, 207–223. doi:10.1111/lre.12003

Liu, S., Costanza, R., Farber, S., Troy, A. (2010). Valuing ecosystem services: theory, practice, and the need for a transdisciplinary synthesis. Annals of the New York Academy of Sciences 1185, 54–78. doi:10.1111/j.1749-6632.2009.05167.x

Lockie, S. (2013). Market instruments, ecosystem services, and property rights: assumptions and conditions for sustained social and ecological benefits. Land Use Policy 31, 90–98. doi:10.1016/j.landusepol.2011.08.010

Locatelli, T., Binet, T., Kairo, J. G., King, L., Madden, S., Patenaude, G. et al. (2014). Turning the tide: how blue carbon and payments for ecosystem services (PES) might help save mangrove forests. Ambio 43, 981–995. doi:10.1007/s13280-014-0530-y

Lovelock, C.E. & McAllister, R.R.J. (2013). ‘Blue Carbon’ projects for the collective good. Carbon Management 4, 477-479. doi: 10.4155/CMT.13.50

Lutz, S.J. (2011). Blue carbon – first level exploration of national coastal carbon in the Arabian Peninsula with special focus on UAE and Abu Dhabi. A rapid feasibility study. AGEDI/EAD. Published by UNEP/GRID-Arendal, Norway. (48 pp.)

McCann, L., Colby, B., Easter, K.W., Kasterine, A., Kuperan, K.V. (2005). Transaction cost measurement for evaluating environmental policies. Ecological Economics 52, 527–542. doi:10.1016/j.ecolecon.2004.08.002

McCann, L. (2013). Transaction costs and environmental policy design. Ecological Economics 88, 253–262. doi:10.1016/j.ecolecon.2012.12.012

Macreadie, P.I., Baird, M.E., Trevathan-Tackett, S.M., Larkum, A W.D., Ralph, P.J. (2013). Quantifying and modelling the carbon sequestration capacity of seagrass meadows: a critical assessment. Marine Pollution Bulletin doi:10.1016/j.marpolbul.2013.07.038

Mahanty, S., Suich, H., Tacconi, L. (2013). Access and benefits in payments for environmental services and implications for REDD+: lessons from seven PES schemes. Land Use Policy 31, 38–47. doi:10.1016/j.landusepol.2011.10.009

Marshall, G.R. (2013). Transaction costs, collective action and adaptation in managing complex social–ecological systems. Ecological Economics 88, 185–194. doi:10.1016/j.ecolecon.2012.12.030

Martin-Ortega, J., Ojea, E., Roux, C. (2013). Payments for water ecosystem services in Latin America: a literature review and conceptual model. Ecosystem Services 6, 122-132. doi:10.1016/j.ecoser.2013.09.008

McArthur, L.C., Boland, J.W. (2006). The economic contribution of seagrass to secondary production in South Australia. Ecological Modelling 196, 163–172. doi:10.1016/j.ecolmodel.2006.02.030

McDermott, M., Mahanty, S., Schreckenberg, K. (2012). Examining equity: a multidimensional framework for assessing equity in payments for ecosystem services. Environmental Science & Policy 1–12. doi:10.1016/j.envsci.2012.10.006

Mendelsohn, R., Olmstead, S. (2009). The economic valuation of environmental amenities and disamenities: methods and applications. Annual Review of Environment & Resources 34, 325–347. doi:10.1146/annurev-environ-011509-135201

Mohammed, E.Y. (2012). Payments for coastal and marine ecosystem services: prospects and principles. International Institute for Environment and Development, London. (4 pp.)

Montefalcone, M., Rovere, A., Parravicini, V., Albertelli, G., Morri, C., Bianchi, C.N. (2013). Evaluating change in seagrass meadows: a time-framed comparison of side scan sonar maps. Aquatic Botany 104, 204–212. doi:10.1016/j.aquabot.2011.05.009

Muňoz-Piňa, C.; Guevara, A.; Torres, J.M. & Braňa, J. (2008). Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. Ecological Economics 65, 725-736. doi:10.1016/j.ecolecon.2007.07.031

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H. (2010). Reconciling theory and practice: an alternative conceptual framework for understanding payments for environmental services. Ecological Economics 69, 1202–1208.doi:10.1016/j.ecolecon.2009.11.006

Muradian, R. and Rival, L. (2012). Between markets and hierarchies: the challenge of governing ecosystem services. Ecosystem Services 1, 93-100. doi:10.1016/j.ecoser.2012.07.009

Murdiyarso, D., Brockhaus, M., Sunderlin, W.D., Verchot, L. (2012). Some lessons learned from the first generation of REDD+ activities. Current Opinion in Environmental Sustainability 4, 678–685. doi:10.1016/j.cosust.2012.10.014

Murray, B. (2013). Blue Carbon: Biophysical Potential and Economic Incentives. Presentation for the 'Sustainable Ocean Summit', Washington DC, 24th April 2013. Available at: http://oceancouncil.org/site/summit_2013/Presentation%20PDFs/6-ECOSERVICES%20SECURED%20PDFS/6-ECOSERVICES_Murray_rev_20130423.pdf (accessed 10/07/2014)

Murray, B., Pendleton, L., Jenkins, W., Sifleet, S. (2011). Green payments for blue carbon: Economic incentives for protecting threatened coastal habitats, Nicholas Institute for Environmental Policy Solutions, Duke University. (52 pp.)

Narloch, U., Pascual, U., Drucker, A.G. (2011). Cost-effectiveness targeting under multiple conservation goals and equity considerations in the Andes. Environmental Conservation 38, 417–425.doi: 10.1017/S0376892911000397

Narloch, U. Pascual, U., Drucker, A.G. (2013). How to achieve fairness in payments for ecosystem services? Insights from agrobiodiversity conservation auctions. Land Use Policy 35, 107-118.doi:10.1016/j.landusepol.2013.05.002

Naughton-Treves, L., Wendland, K. (2014). Land tenure and tropical forest carbon management. World Development 55, 1–6. doi:10.1016/j.worlddev.2013.01.010

Nordlund, L., Erlandsson, J., de la Torre-Castro, M., Jiddawi, N. (2011). Changes in an East African social-ecological seagrass system: invertebrate harvesting affecting species composition and local livelihood. Aquatic Living Resources 23, 399–416. doi:10.1051/alr/2011006

Noordwijk, van M., Leimona, B., Emerton, L., Tomich, T.P., Velarde, S.J., Kallesoe, M. (2007). Criteria and indicators for environmental service compensation and reward mechanisms: realistic, voluntary, conditional and pro-poor. CES Scoping Study Issue Paper no. 2. ICRAF Working Paper no. 37. World Agroforestry Centre. Nairobi, Kenya

Norgaard, R.B. (2010). Ecosystem services: from eye-opening metaphor to complexity blinder. Ecological Economics 69, 1219-1227. doi: 10.1016/j.ecolecon.2009.11.009

Ondiviela, B., Losada, I.J., Lara, J.L., Maza, M., Galván, C., Bouma, T.J. (2014). The role of seagrasses in coastal protection in a changing climate. Coastal Engineering 87, 158–168. doi:10.1016/j.coastaleng.2013.11.005

Orth, R., Carruthers, T., Dennison, W., Duarte, C., Fourqueram, J., Heck, K., Hughes, R. (2006). A global crisis for seagrass ecosystems. Bioscience 56, 987–996. http://dx.doi.org/10.1641/0006-3568(2006)56

Parker, C., and M. Cranford. 2010. The little biodiversity finance book: a guide to proactive investment in natural capital (PINC). Oxford, UK: Global Canopy Foundation.

Pascual, U., Muradian, R., Rodríguez, L.C., Duraiappah, A. (2010). Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. Ecological Economics 69, 1237–1244. doi:10.1016/j.ecolecon.2009.11.004

Pendleton, L.H., Sutton-Grier, A.E., Gordon, D.R., Murray, B.C., Victor, B.E., Griffis, R.B. et al. (2013). Considering 'coastal carbon' in existing U.S. federal statutes and policies. Coastal Management 41, 439-456. doi: 10.1080/0820753.2013.822294

Pendleton, L.H., Donato, D.C., Murray, B.C., Crooks, S., Jenkins, A.W., Sifleet, S. et al. (2012). Estimating global 'blue carbon' emissions from conversion and degradation of vegetated coastal ecosystems. PLOS One 7(9) e43542. doi: 10..1371/journal.pone.0043542

Peters-Stanley, M., Hamilton, K., Yin, D. (2012). Leveraging the landscape: State of the forest carbon markets 2012, A Report by Forest Trends' Ecosystem Marketplace. Washington, DC.

Peters-Stanley, M., Yin, D. (2013). Manoeuvring the mosaic. State of the voluntary carbon markets 2013, a report by Forest Trends' Ecosystem Marketplace. Washington, DC.

Petus, C., Collier, C., Devlin, M., Rasheed, M., McKenna, S. (2014). Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). Marine Environmental Research 98, 68–85. doi:10.1016/j.marenvres.2014.03.006

Phelps, J., Webb, E.L., Adams, W.M. (2012). Biodiversity co-benefits of policies to reduce forest-carbon emissions. Nature Climate Change 2, 1–7. doi:10.1038/nclimate1462

Phinn, S., Roelfsema, C., Dekker, A., Brando, V., Anstee, J. (2008). Mapping seagrass species, cover and biomass in shallow waters: an assessment of satellite multi-spectral and airborne hyper-spectral imaging systems in Moreton Bay (Australia). Remote Sensing Environment 112, 3413–3425. doi:10.1016/j.rse.2007.09.017

Pillay, M., Rogerson, C.M. (2013). Agriculture-tourism linkages and pro-poor impacts: the accommodation sector of urban coastal KwaZulu-Natal, South Africa. Applied Geography 36, 49–58. doi:10.1016/j.apgeog.2012.06.005

Pirard, R. Billé, R., Sembrés, T. (2010). Questioning the theory of payments for ecosystem services (PES) in light of emerging experience and plausible developments. Institut du Développement Durable et des Relations Internationales, Paris. No.4 pp1-24

Pogoreutz, C., Kneer, D., Litaay, M., Asmus, H., Ahnelt, H. (2012). The influence of canopy structure and tidal level on fish assemblages in tropical Southeast Asian seagrass meadows. Estuarine & Coastal Shelf Science 107, 58–68. doi:10.1016/j.ecss.2012.04.022

Pokorny, B., Johnson, J., Medina, G., Hoch, L. (2012). Market-based conservation of the Amazonian forests: revisiting win–win expectations. Geoforum 43, 387–401.doi:10.1016/j.geoforum.2010.08.002

Pokorny, B., Scholz, I., Jong, W. De. (2013). REDD+ for the poor or the poor for REDD+? About the limitations of environmental policies in the Amazon and the potential of achieving environmental goals through pro-poor policies. Ecology & Society 18(2): 3. http://dx.doi.org/10.5751/ES-05458-180203

Porras, I. (2010). Fair and green? Social impacts of payments for environmental services in Costa Rica. International Institute for Environment and Development, London. (37 pp.)

Porras, I., Aylward, B., Dengel, J. (2013). Sustainable Markets Monitoring payments for watershed services schemes in developing countries. International Institute for Environment and Development, London. (36 pp.)

Potts, M.D., Kelley, L.C., Doll, H.M. (2013). Maximizing biodiversity co-benefits under REDD+: a decoupled approach. Environment Research Letters 8, 024019. doi:10.1088/1748-9326/8/2/024019

Raheem, N., Colt, S., Fleishman, E., Talberth, J., Swedeen, P., Boyle, K. J. et al. (2012). Application of non-market valuation to California's coastal policy decisions. Marine Policy 36, 1166–1171. doi:10.1016/j.marpol.2012.01.005

Resosudarmo, I.A.P., Atmadja, S., Ekaputri, A.D., Intarini, D.Y., Indriatmoko, Y., Astri, P. (2014). Does tenure security lead to REDD+ project effectiveness? Reflections from five emerging sites in Indonesia. World Development 55, 68–83. doi:10.1016/j.worlddev.2013.01.015

Rivera-Guzmán, N.E., Moreno-Casasola, P., Ibarra-Obando, S.E., Sosa, V.J., Herrera-Silveira, J. (2014). Long term state of coastal lagoons in Veracruz, Mexico: effects of land use changes in watersheds on seagrasses habitats. Ocean & Coastal Management 87, 30–39. doi:10.1016/j.ocecoaman.2013.10.007

Robert, N., Stenger, A. (2013). Can payments solve the problem of undersupply of ecosystem services? Forest Policy & Economics 35, 83–91. doi:10.1016/j.forpol.2013.06.012

Rodríguez de Francisco, J.C., Budds, J., Boelens, R. (2013). Payment for environmental services and unequal resource control in Pimampiro, Ecuador. Society & Natural Resources 26, 1217–1233. doi:10.1080/08941920.2013.825037

Rudd, M. A., (2004). An institutional framework for designing and monitoring ecosystem-based fisheries management policy experiments. Ecological Economics 48, 109-124. doi:10.1016/j.ecolecon.2003.10.002

Rudd, M.A. (2009). National values for regional aquatic species at risk in Canada. Endangered Species Research 6, 239-249.

Rudd, M. A. (2014). Scientists' perspectives on global ocean research priorities. Frontiers in Marine Science, 1(August), 1–20. doi:10.3389/fmars.2014.00036

Rudd, M. A., Tupper, M. H., Folmer, H., & Kooten, G. C. Van. (2003). Policy analysis for tropical marine reserves: challenges and directions. Fish and Fisheries 4, 65–85. doi: 10.1046/j.1467-2979.2003.00110.x

Rudd, M.A., Weigand, J. (2011). Citizens' willingness to pay for improved ecosystem services arising from wastewater clean-up in Newfoundland, Canada. International LOICZ Open Science Conference 2011, Yantai China, 14th September 2011. Available at: https://www.academia.edu/1699571/Citizens_willingness_to_pay_for_improved_ecosystem_services_arising_from_wastewater_cleanup_in_Newfoundland_Canada (accessed 10/10/2014)

Rudd, M. A., & Lawton, R. N. (2013). Scientists' prioritization of global coastal research questions. Marine Policy 39, 101–111. doi:10.1016/j.marpol.2012.09.004

Saenger, P., Gartside, D., Funge-Smith, S. (2013). A review of mangrove and seagrass ecosystems and their linkage to fisheries and fisheries management. RAP Publication, FAO, Bangkok. (75 pp.)

Salazar, N.B. (2012). Community-based cultural tourism: issues, threats and opportunities. Journal of Sustainable Tourism 20, 9–22. doi:10.1080/09669582.2011.596279

Salem, M.E., Mercer, D.E. (2012). The economic value of mangroves: a meta-analysis. Sustainability 4, 359-383. doi:10.3390/su4030359

Sánchez-Carnero, N., Rodríguez-Pérez, D., Couñago, E., Aceña, S., Freire, J. (2012). Using vertical sidescan sonar as a tool for seagrass cartography. Estuarine & Coastal Shelf Science 115, 334–344. doi:10.1016/j.ecss.2012.09.015

Schomers, S., Matzdorf, B. (2013). Payments for ecosystem services: a review and comparison of developing and industrialized countries. Ecosystem Services 6, 16-30. doi:10.1016/j.ecoser.2013.01.002

Seifert-Granzin, J. (2010). Integrating green and blue carbon management: how to make it work. Katoomba Group Meeting XVI, Palto Alto, February 9th-10th 2010. Available at: http://rmportal.net/library/content/translinks/translinks-2010/forest-trends/2010-Katoomba-Meeting-XVI-Marine/Presentation_IntegratingGreenandBlueCarbonManagement.pdf/view (accessed 12/07/2014)

Senaratna Sellamuttu, S., de Silva, S., Nguyen-Khoa, S. (2011). Exploring relationships between conservation and poverty reduction in wetland ecosystems: lessons from 10 integrated wetland conservation and poverty reduction initiatives. International Journal of Sustainable Development & World Ecology 18, 328–340. doi:10.1080/13504509.2011.560034

Shiferaw, B., Obare, G., Muricho. (2008). Rural market imperfections and the role of institutions in collective action to improve markets for the poor. Natural Resources Forum 32, 25-38. doi: 10.1111/j.1477-8947.2008.00167.x

Short, F.T., Coles, R., Fortes, M.D., Victor, S., Salik, M., Isnain, I, (2014). Monitoring in the Western Pacific region shows evidence of seagrass decline in line with global trends. Marine Pollution Bulletin doi:10.1016/j.marpolbul.2014.03.036

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S., Bujang, J.S. et al. (2011). Extinction risk assessment of the world's seagrass species. Biological Conservation 144, 1961–1971. doi:10.1016/j.biocon.2011.04.010

Sifleet, S., Pendleton, L., & Murray, B. C. (2011). State of the science on coastal Blue Carbon: a summary for policy makers. Nicholas Institute for Environmental Policy Solutions, Duke University. (43 pp.).

Siikamäki, J., Sanchirico, J.N., Jardine, S., McLaughlin, D., Morris, D. (2013). Blue Carbon: coastal ecosystems, their carbon storage, and potential for reducing emissions. Environmental Science & Policy for Sustainable Development 55, 14–29. doi:10.1080/00139157.2013.843981

Sommerville, M.M., Milner-Gulland, E.J., Jones, J.P.G. (2011). The challenge of monitoring biodiversity in payment for environmental service interventions. Biological Conservation 144, 2832–2841. doi:10.1016/j.biocon.2011.07.036

Somorin, O. A., Visseren-Hamakers, I.J., Arts, B., Sonwa, D.J., Tiani, A.-M. (2014). REDD+ policy strategy in Cameroon: Actors, institutions and governance. Environmental Science & Policy 35, 87–97. doi:10.1016/j.envsci.2013.02.004

Steinicke, E., Neuburger, M. (2012). The impact of community-based Afro-alpine tourism on regional development. Mountain Research & Development 32, 420–430.

Sunderlin, W.D., Larson, A.M., Duchelle, A.E., Resosudarmo, I.A.P., Huynh, T.B., Awono, A. et al. (2014). How are REDD+ Proponents Addressing Tenure Problems? Evidence from Brazil, Cameroon, Tanzania, Indonesia, and Vietnam. World Development 55, 37–52. doi:10.1016/j.worlddev.2013.01.013

Sutton, A. M., & Rudd, M. A. (2014). Deciphering contextual influences on local leadership in community-based fisheries management. Marine Policy 50, 261–269. doi:10.1016/j.marpol.2014.07.014

Tacconi, L. (2012). Redefining payments for environmental services. Ecological Economics 73, 29–36.doi:10.1016/j.ecolecon.2011.09.028

TEEB (2010), The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. Edited by Pushpam Kumar. Earthscan, London and Washington.

Teh, L.C.L., Teh, L.S.L., Chen Chung, F. (2008). A private management approach to coral reef conservation in Sabah, Malaysia. Biodiversity & Conservation 17, 3061-3077. doi:10.1007/s10531-007-9266-3

Thomas, S. (2014). Blue carbon: knowledge gaps, critical issues and novel approaches. Ecological Economics 107, 22-38. Doi: 10.1016/j.ecolecon.2014.07.028

Thompson, B.S., Clubbe, C.P., Primavera, J.H., Curnick, D., Koldewey, H.J. (2014). Locally assessing the economic viability of blue carbon: a case study from Panay Island, the Philippines. Ecosystem Services 1–13. doi:10.1016/j.ecoser.2014.03.004

Torres-Pulliza, D., Wilson, J.R., Darmawan, A., Campbell, S.J., Andréfouët, S. (2013). Ecoregional scale seagrass mapping: a tool to support resilient MPA network design in the coral triangle. Ocean & Coastal Management 80, 55–64. doi:10.1016/j.ocecoaman.2013.04.005

Tudela, S. (2004). Ecosystem effects of fishing in the Mediterrean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats. General Fisheries Council for the Mediterranean Studies and Reviews. 74, 1-44

Turner, R.K., Morse-Jones, S., Fisher, B. (2010). Ecosystem valuation: a sequential decision support system and quality assessment issues. Annals of the New York Academy of Sciences 1185, 79–101. doi:10.1111/j.1749-6632.2009.05280.x

Tuya, F., Haroun, R., Espino, F. (2014). Economic assessment of ecosystem services: monetary value of seagrass meadows for coastal fisheries. Ocean & Coastal Management 1–7. doi:10.1016/j.ocecoaman.2014.04.032

UK National Ecosystem Assessment (2011) The UK National Ecosystem Assessment: Synthesis of the key findings. UNEP-WCMC, Cambridge.

Ullman, R., Bilbao-Bastida, V., Grimsditch, G. (2013). Including Blue Carbon in climate market mechanisms. Ocean & Coastal Management 83, 15–18. doi:10.1016/j.ocecoaman.2012.02.009

Unsworth, R.K.F., Cullen, L.C. (2010). Recognising the necessity for Indo-Pacific seagrass conservation. Conservation Letters 3, 63–73. doi:10.1111/j.1755-263X.2010.00101.x

UNEP. (2011). Economic analysis of mangrove forest: a case study in Gazi Bay, Kenya. United Nations Environment Programme, Nairobi. (42 pp.)

UNEP and CIFOR. (2014).Guiding principle for delivering coastal wetland carbon projects. United Nations Environment Programme, Nairobi, Kenya and Center for International Forestry Research, Bogor, Indonesia. (57 pp.)

UNFCC 2014. Kyoto Protocol. Available at: http://unfccc.int/kyoto_protocol/items/2830.php (accessed 23/11/2014)

Unsworth, R.K.F., Cullen, L.C., Pretty, J.N., Smith, D.J., Bell, J.J. (2010). Economic and subsistence values of the standing stocks of seagrass fisheries: potential benefits of no-fishing marine protected area management. Ocean & Coastal Management 53, 218–224. doi:10.1016/j.ocecoaman.2010.04.002

Unsworth, R.K.F., van Keulen, M., Coles, R.G. (2014). Seagrass meadows in a globally changing environment. Marine Pollution Bulletin 1–4. doi:10.1016/j.marpolbul.2014.02.026

Vaidyanathan, G. (2011). ‘Blue Carbon’ plan takes shape. Nature. doi:10.1038/news.2011.112

Vassallo, P., Paoli, C., Rovere, A., Montefalcone, M., Morri, C., Bianchi, C.N. (2013). The value of the seagrass Posidonia oceanica: a natural capital assessment. Marine Pollution Bulletin 75, 157–67. doi:10.1016/j.marpolbul.2013.07.044

Visseren-Hamakers, I.J., Gupta, A., Herold, M., Peña-Claros, M., Vijge, M.J. (2012). Will REDD+ work? The need for interdisciplinary research to address key challenges. Current Opinion Environment Sustainability 4, 590–596. doi:10.1016/j.cosust.2012.10.006

Vo, Q.T., Kuenzer, C., Vo, Q.M., Moder, F., Oppelt, N. (2012). Review of valuation methods for mangrove ecosystem services. Ecological Indicators 23, 431–446. doi:10.1016/j.ecolind.2012.04.022

Wabnitz, C.C., Andréfouët, S., Torres-Pulliza, D., Müller-Karger, F.E., Kramer, P. A. (2008). Regional-scale seagrass habitat mapping in the wider Caribbean region using landsat sensors: applications to conservation and ecology. Remote Sensing Environment 112, 3455–3467. doi:10.1016/j.rse.2008.01.020

Watson, R.A., Coles, R.G., Lee Long, W.J. (1993). Simulation estimates of annual yield and landed value for commercial penaeid prawns from a tropical seagrass habitat, Northern Queensland, Australia. Australian Journal of Marine Freshwater Research 44, 211-219. doi:0067-1940/93/010211$05.00

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences U. S. A. 106, 12377–81. doi:10.1073/pnas.0905620106

Williams S.L. and Heck Jr K.L. (2001). Seagrass community ecology. In: Bertness MD, Gaines SD, and Hay ME (Eds). Marine community ecology. Sunderland, MA: Sinauer Associates

Wendland, K.J., Honzák, M., Portela, R., Vitale, B., Rubinoff, S., Randrianarisoa, J. (2010). Targeting and implementing payments for ecosystem services: opportunities for bundling biodiversity conservation with carbon and water services in Madagascar. Ecological Economics 69, 2093–2107. doi:10.1016/j.ecolecon.2009.01.002

Willemen, L., Drakou, E.G.,, Dunbar, M.B., Mayaux, P., Egoh, B.N. (2013). Safeguarding ecosystem services and livelihoods: understanding the impact of conservation strategies on benefit flows to society. Ecosystem Services 4, 95–103.doi:10.1016/j.ecoser.2013.02.004

World Bank. (2010). Implementation completion and results report: western Kenya integrated ecosystem management project. Report No. ICR00001533

World Resources Institute (WRI) in collaboration with United Nations Development Programme, United Nations Environment Programme, and World Bank. 2005. World Resources 2005: The Wealth of the Poor – Managing Ecosystems to fight Poverty. World Resources Institute, Washington, D.C.

Wunder, S. (2005). Payments for environmental services: some nuts and bolts. Occasional Paper No. 42. Center for International Forestry Research, Bogor, Indonesia. (32 pp.)

Wunder, S. & Albán, M. (2008). Decentralized payments for environmental services: the case of Pimampiro and PROFAFOR in Ecuador. Ecological Economics 65, 685-698. doi:10.1016/j.ecolecon.2007.11.004

Wunder, S., Engel, S., Pagiola, S. (2008). Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. Ecological Economics 65, 834–852. doi:10.1016/j.ecolecon.2008.03.010

Wünscher, T., Engel, S., Wunder, S. (2008). Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. Ecological Economics 65, 822–833. doi:10.1016/j.ecolecon.2007.11.014

Yandle, T. (2007). Understanding the consequences of property rights mismatches: a case study of New Zealand's marine resources. Ecology & Society 12(2) 27.

Zuidema, C.; Plate, R.; Dikou, A. (2011). To preserve or to develop? East Bay dredging project, South Caicos, Turks and Caicos Islands. Journal of Coastal Conservation 15, 555-563. doi:10.1007/s11852-011-0144-5

Supplementary Material

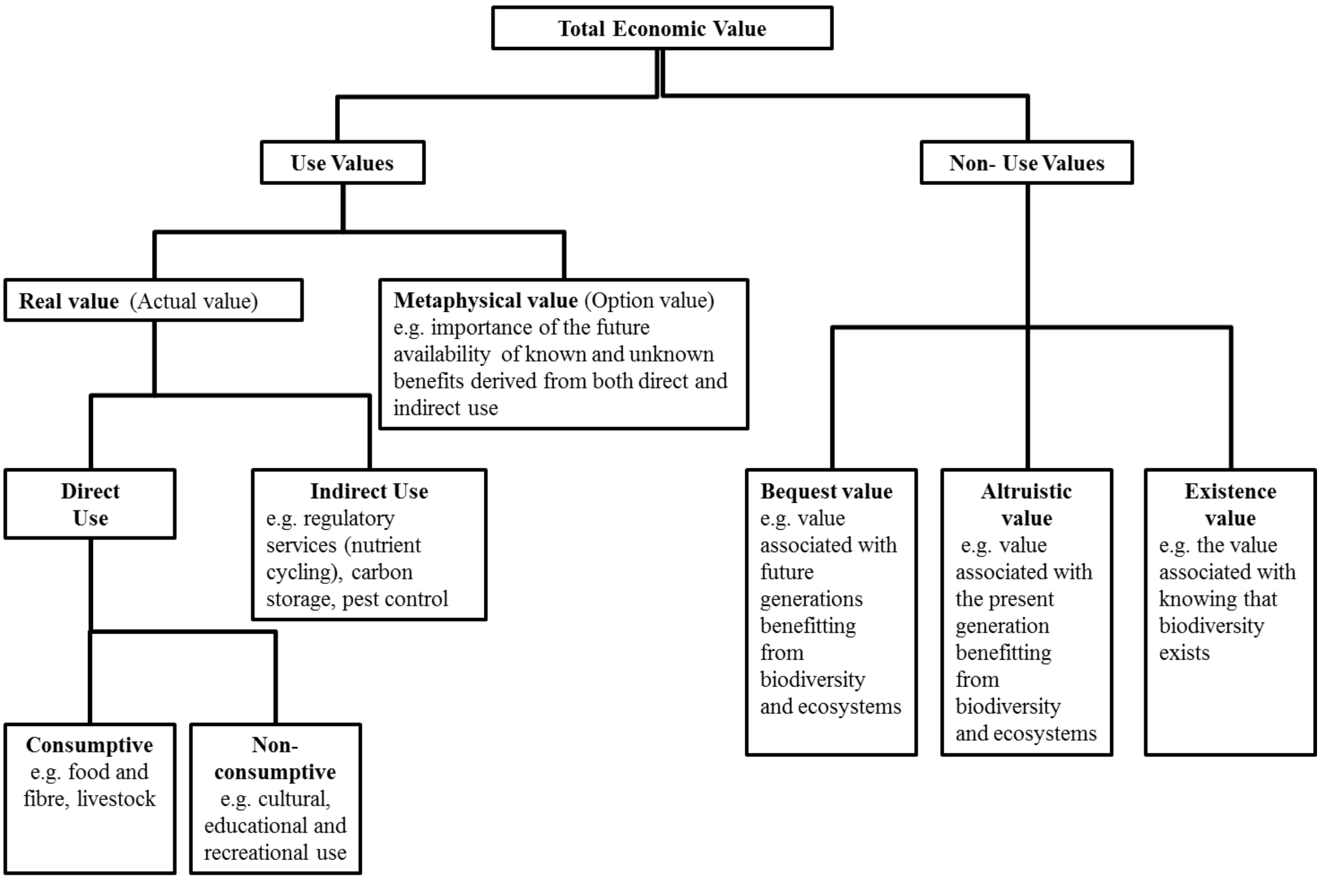

Figure S1. Diagrammatic representation of the ecosystem service economic valuation components and their relationship to each other. Adapted from Vo et al. (2012) and TEEB (2012)

Table S1. Economic valuation methodologies and their associated pros and cons

Valuation Approach	**Valuation Methodology**	**Advantages and Disadvantages of Valuation Techniques**
Market Cost	Avoided Cost: ES valued on the basis of costs avoided i.e. prohibiting the degradation or damage of environmental benefits	Mismatches can arise between the likely benefits of intervention compared to original benefits leading to misleading WTP results. Applies the precautionary principle. Can estimate indirect-use benefits
	Production Function: value of ecological function with regards to economic output effects. Changes in ES quality and quantity on human-wellbeing	Not able to assess non-use values. Difficult to derive data about changes in ES. Widely employed in the contexts of coastal and wetland ecosystems
Market Price	Market: based on Willingness to Pay (WTP)	Requires market data (questionable reliability), and policies may distort market prices. However, market prices reflect personal WTP and market price data is relatively easy to obtain
Revealed Preference	Travel Cost: survey method valuing site-based facilities. WTP for environmental benefits at particular locations	The method is data intensive, it does not estimate non-use values and complex journeys are problematical. However, it is widely used and used in developing countries for assessing ecotourism
	Hedonic Pricing: valuations based on implied WTP via purchases in related markets – mainly labour and property	The method is data intensive, it does not estimate non-use values, and income-level restricts choices whilst surrogate markets must be a good reflection of values. However it can value the impact of some ES on land values
Stated Preference	Contingent Valuation: WTP or WTA compensation for alterations in ES. Respondents can name an amount they would pay (classical CV), or are asked to say whether they would pay a specific amount (di/polychotomous choice) or select an amount from several options (Choice Modelling).	*Contingent Valuation*: this method suffers from several sources of bias, inconsistent preferences, it is costly and labour intensive to develop and implement and can miss non-trivial information. However, it is able to estimate option and existence values.
	Choice modelling: involves more elaborate sets of scenarios (or choices) from which participant select their preferred alternatives based on a set of choice attributes. Choices are constructed to reveal the marginal rate of substitution between a specific attribute and the trade-off item.	*Choice Modelling*: hypothetical bias and the choices can be complex where attribute numbers are high. However, compared to standard CV the experimenter has much more control, the statistics are more robust, attribute range is greater and the method suffers less from respondent strategic behaviour.
Value Transfer	Benefit Transfer: transference of values at one location (study site) to another location (policy site) of which there are four types: unit BT, adjusted BT, value function transfer and meta-analytic transfer	Large number of uncertainties not wholly accounted for between study and policy locations. Transfer of values from one context to another is difficult. Nevertheless, it is a quick and cheap method.
Participatory Valuation	Deliberative valuation: combines states preference methods with deliberative processes from political science, involving small groups of participants in reflective iterative dialogues.	Less bias encountered compared to standard stated preference methods. Values are constructed in a social process. Inclusive of all stakeholder groups, but depending on the power-relations of stakeholders involved some value preferences may be articulated more forcefully than others.

Non-monetary Deliberative and Participatory Approaches	Focus groups, Participatory Action Research (PAR), Health-based, Q-methodology: These are a set of group-based methods that are both participatory and deliberative, and seek to obtain information regarding human-nature relationships. PARs were developed specifically for use in developing countries to elicit local knowledge and enable local people to participate in decision-making. Health-based measures relate valuations to factors that affect quality of life and human-wellbeing. Q-methodology is a means of assessing the subjectivity of people's views and values.	Overall, these methods are able to provide values regarding biodiversity, provisioning, regulating and cultural services, and they enrich the qualitative components of value. Although they require literate participants, new data collection, trained individuals and can be affected by local nuances. However, protocols can be adjusted to illiterate individuals; values can be aggregated to the scale required and in some cases they can be relatively straightforward to undertake. Furthermore, they engage a wide-range of stakeholders and are conveyable to policy makers.

Adapted from Mendelsohn and Olmstead (2009), Lui et al. (2010), Pascual and Muradian et al. (2010), Turner et al. (2010), Christie et al. (2012) and Liekens et al. (2014)

Table S2. Examples of PES schemes that jointly focus on carbon management and the provision of additional ecosystem services

PES Study	PES Description
Carbon Livelihoods Project: Mozambique *Source*: Hedge and Bull (2011) and Groom and Palmer (2012)	The project operated across several villages and aimed to establish a viable alternative livelihood, agro-forestry and carbon credit scheme. Agroforestry was designed to generate carbon offsets alongside new 'on-farm' labour activities, whilst the alternative livelihood element promoted 'off-farm' micro-enterprises. Initially funded by the European Union, the programme became self-financing following sales of verified emissions reductions (VERs) in the voluntary carbon market. VER sales were used to establish an annual PES fund that dispensed payments to farmers over a seven year period. Remaining revenue was channelled into a community trust fund for development projects such as healthcare support. Adoption of agro-forestry practices meant households generated new 'on-farm' income by selling crops or harvesting non-timber forest products following cessation of carbon payments. Micro-enterprises such as bee-keeping, plant nurseries, carpentry and even a community sawmill provided viable and secure alternative revenue sources for farmers. In addition, some local people were hired by the project. *Criticisms*: Carbon offset payments were less important (proportionally) than income from the project's alternative revenue sources. Micro-enterprises potentially undermined the sustainability of 'on-farm' activities through changes in labour allocation. Gender discrimination contributed to uneven income distribution between male- and female-headed households, and project costs were significant; with two thirds of carbon offset sales revenue directed towards overheads.
Western Kenya Integrated Ecosystem Management Programme (WKIEMP): Kenya *Source*: World Bank (2010)	WKIEMP was initiated to provide a viable community-livelihood development model. Implemented across 15 micro-watersheds WKIEMP focused on land productivity and sustainable-use by supporting on-farm and off-farm conservation strategies and building institutional capacity; alongside promoting management interventions geared towards biodiversity and carbon sequestration and storage. Overall the project was moderately successful. Households did not receive payment; but derived income through improved land productivity, livelihood diversification and technical capacity. Estimated net present value to participating households is considered to be US$1193 to US$2844. Moreover, 60% of beneficiary households reported an increase in food production and consumption directly addressing poverty alleviation. Furthermore, the project established institutional networks to enhance the sustainability of community activities following project cessation such as basin technical committees that promoted cross-collaboration. *Criticisms:* Two problems undermined WKIEMP's notion of sustainability. First, project permanency: the project ran for only five years from 2005 to 2010. Second, the programme encountered fiscal constraints that hampered its implementation and operation leading to disjointed upstream and downstream management interventions. Overall, the failure to secure adequate co-financing of funds significantly impaired project performance.
Socio Bosque: Ecuador Source: de Koning et al. (2011) and Krasue and Loft (2013)	Socio Bosque is a nationwide government initiative designed to realise biodiversity conservation, climate mitigation and poverty alleviation benefits. Participants receive direct monetary transfers on a per hectare basis for protecting native forests and ecosystems through voluntary but monitored twenty year conservation agreements. Payments are made on a descending scale, with amounts reduced incrementally as the land enrolled increases providing a built-in equity mechanism. Participants are individual landowners or local indigenous communities, and so land is privately or communally owned. Only land that has a high probability of deforestation, sufficient carbon storage, water regulation and biodiversity capacity and is found in relatively socially-deprived areas is eligible for enrolment. Overall 260000 Ha yr-1 of forest have been protected. Remuneration is conditional, requiring compliance with a social investment plan (directing how incentives might best be used to improve social conditions) and conservation obligations. Social benefits are realised through monetary investments in health, education, household consumption, debt repayments, infrastructure and institutional capacity. *Criticisms:* Payments allocated to participants are not equal: less than a fifth of households in community agreements receive more than US$500 yr-1 compared to 92% of private landholders. The scheme has underperformed with regards to distributing individual and collective contracts in a way that accounts for the number of beneficiaries per contract and their poverty status.