

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

Hejnowicz, Adam P.; Kennedy, Hilary; Rudd, Murray A.; Huxham, Mark R.

Frontiers in Marine Science

DOI:
[10.3389/fmars.2015.00032](https://doi.org/10.3389/fmars.2015.00032)

Published: 09/06/2015

Peer reviewed version

[Cyswllt i'r cyhoeddiad / Link to publication](#)

Dyfyniad o'r fersiwn a gyhoeddwyd / Citation for published version (APA):
Hejnowicz, A. P., Kennedy, H., Rudd, M. A., & Huxham, M. R. (2015). Harnessing the climate mitigation, conservation and poverty alleviation potential of seagrasses: prospects for developing blue carbon initiatives and payment for ecosystem service programmes. *Frontiers in Marine Science*, 2(32). <https://doi.org/10.3389/fmars.2015.00032>

Hawliau Cyffredinol / General rights

Copyright and moral rights for the publications made accessible in the public portal are retained by the authors and/or other copyright owners and it is a condition of accessing publications that users recognise and abide by the legal requirements associated with these rights.

- Users may download and print one copy of any publication from the public portal for the purpose of private study or research.
- You may not further distribute the material or use it for any profit-making activity or commercial gain
- You may freely distribute the URL identifying the publication in the public portal ?

Take down policy

If you believe that this document breaches copyright please contact us providing details, and we will remove access to the work immediately and investigate your claim.

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

Adam P Hejnowicz¹, Hilary Kennedy², Murray A Rudd¹, Mark Huxham³

¹Environment Department, University of York, Heslington, York, UK

²School of Ocean Sciences, Bangor University, Menai Bridge, Anglesey, UK

³School of Life, Sport and Social Sciences, Edinburgh Napier University, Edinburgh, UK

Article Type: Original Research Article

Words (main text): 8871

References: 244

Tables/Boxes: 6/1

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

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

13 Human actions provide a triumvirate of environmental, biological and climatological
14 stressors that act across spatial and temporal scales delivering locally-specific impacts (Orth et al.
15 2006). Drivers of seagrass ecosystem decline include: eutrophication and solid waste disposal
16 (nutrient pollution); aquaculture; thermal pollution; physical alteration or habitat damage (via
17 dredging, coastal infrastructural developments, land reclamation and mechanical destruction);
18 disease spread and invasive species introductions; climate change; over-fishing; overexploitation;
19 and land-runoff from deforestation, mining and agriculture (Duarte 2002; Erfemeijer and Robin
20 Lewis III 2006; Orth et al. 2006; Waycott et al. 2009; Short et al. 2011; Zuidema et al. 2011; Hicks
21 and McClanahan 2012; Cullen-Unsworth and Unsworth 2013; Cullen-Unsworth et al. 2013; Baker
22 et al. 2014).

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

31 Globally, twenty four percent of seagrass species are now classified as threatened or near
32 threatened on the IUCN's Red List (Short et al. 2011). Estimates of the rate of seagrass decline have
33 increased over the last 70 years, from 0.9% yr⁻¹ prior to 1940 rising to 7% yr⁻¹ since 1980 (Waycott
34 et al. 2009; Duarte et al. 2013a; Fourqurean et al. 2012). The global decline of seagrass ecosystems
35 threatens to exacerbate climate change (Duarte et al. 2010; Kennedy et al. 2010; Fourqurean et al.
36 2012; Lavery et al. 2013), undermine the supply of a range of other ecosystem services (Bujang et
37 al. 2006; Orth et al. 2006; Waycott et al. 2009; Short et al. 2011; Cullen-Unsworth and Unsworth
38 2013) and consequently detrimentally affect subsistence livelihoods (Cullen and Unsworth 2010;
39 Nordlund et al. 2010).

40 This reality reflects the complexity of seagrass ecosystems, particularly the connections
41 seagrass meadows have with marine and terrestrial systems, and therefore the difficulties and
42 challenges associated with their management, which are embedded within broader coastal and
43 ocean management issues (Rudd and Lawton 2013). For example, in a recent global ocean research
44 priorities exercise (Rudd 2014) several top-ranked priorities had implications for seagrass
45 ecosystems, including: 'greenhouse gas flux' (7th); 'climate change mitigation and manipulation'
46 (8th); 'ecosystem structure to service linkages' (16th); 'upland hydrology effects on oceans' (24th);
47 'coastal hazard management' (35th); 'ecosystem management alternatives' (40th) and 'integrated
48 upland coastal management' (43rd). Our view is that research is needed on multiple fronts to create
49 enabling conditions and the evidence base needed to craft innovative new policy tools for
50 conservation and mitigating the potential adverse effects of climate change.

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

61 We examine five key issues. In Section 2, we summarize ecosystem services (ES) provided
 62 by seagrass ecosystems and the salient information needed concerning these ES to develop
 63 incentive schemes. In Section 3, we ask how ecosystem service valuation information could be
 64 applied to design and implement new policy innovations. In Section 4, we examine the prospects
 65 for seagrass carbon finance based on current climate policy frameworks. In Section 5, we broaden
 66 the scope to financing instruments that could be developed based on the multiple ES that seagrass
 67 ecosystems provide. Lastly, in Section 6, we summarize the key design, implementation and
 68 governance issues that must be addressed to bring functioning seagrass PES schemes to fruition. In
 69 addition, we highlight the relevant ocean priority research questions that relate to each stage (Rudd
 70 2014), setting our seagrass-oriented research in the broader context of ocean research prioritization.

71 **2 Seagrass ecosystems and ecosystem services**

72 Seagrass ecosystems provide supporting, regulating, provisioning, and cultural ecosystem
 73 services (Barbier et al. 2011; Raheem et al. 2012; Cullen-Unsworth and Unsworth 2013). Here we
 74 briefly summarize the literature regarding the most widely cited ES supplied by seagrass
 75 ecosystems and the information required about each in order to include them in an incentive
 76 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: <ul style="list-style-type: none"> • Acoustic side scan sonar which is useful up to 25m depths and has been used to map seagrass communities in the Mediterranean (e.g., Montefalcone 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: <ul style="list-style-type: none"> • 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, ²¹⁰Pb, 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)

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

81 **2.1 Regulating services: climate regulation**

82 Historically, seagrass meadows had been virtually ignored in global carbon budgets (Duarte
83 et al., 2005). More recently their role in combating climate change through carbon storage and
84 sequestration has become more clearly recognised, while simultaneously the spatial extent of
85 seagrass meadows has continued to decline (Duarte et al. 2010; Kennedy et al. 2010; Fourqurean et
86 al. 2012; Duarte et al. 2013a; Lavery et al. 2013). Although a small fraction (18 to 60 x 10⁶ ha) of
87 the world’s ocean area seagrass meadows sequester 20% of global marine carbon and store 10% of
88 annual buried organic carbon (C_{org}) (Fourqurean et al. 2012; Pendleton et al. 2012). Consequently,
89 seagrass ecosystems play potentially central roles in how oceanic ecosystems can mitigate climate
90 change, a question ranked 8th in global importance by marine scientists (Rudd 2014).

91 Seagrass meadows are highly productive systems, especially in Indo-Pacific regions, and
92 provide habitat for diverse communities (Short et al. 2011). However, worldwide, seagrass standing
93 biomass is small (76-151 Tg C) relative to the biomass of the vegetation in other coastal ecosystems
94 (Fourqurean et al. 2012). Nonetheless, the high productivity of seagrass meadows, with potential

95 net community production (NCP) of $6.7 \text{ t C ha}^{-1} \text{ yr}^{-1}$ (several times higher than NCP rates
96 associated with Amazonian forests and North American wetlands), contributes significantly to their
97 carbon sink capacity (Duarte et al. 2010). Approximately 20% to 60% of this aboveground
98 productivity derives from the autotrophic epiphytes that seagrass meadows support (Duarte et al.
99 2013a). Moreover, seagrass meadows trap allochthonous material, including large amounts of
100 particulate carbon, which combined with their ability to bury carbon enables seagrass meadows to
101 store large amounts of carbon (Duarte et al. 2013a).

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

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

121 Despite the uncertainties, alongside the lack of attention given to the potential implications
122 of extensive conversion of standing carbon pools beneath vegetative coastal ecosystems more
123 generally, it is clear that seagrass meadows constitute an important global carbon sink whose
124 continued loss threatens to exacerbate climate change (Duarte et al. 2010; Pendleton et al. 2012).
125 Indeed, global carbon emissions maybe enhanced by an additional 3% to 19% from the destruction
126 of vegetative coastal ecosystems (Pendleton et al. 2012). Based on current assessments seagrass
127 biomass loss may release between 11 and 23 Tg C yr^{-1} into the ocean-atmosphere system, and a
128 further 63 to 297 Tg C yr^{-1} into the ocean-atmosphere CO_2 reservoir through the oxidization of the
129 underlying sediment (Fourqurean et al. 2012). Additionally, seagrass loss reduces the overall carbon
130 accumulation rate (equivalent to between 6 and 24 Tg C yr^{-1}). Collectively, this represents
131 considerable CO_2 emission potential (131 to $522 \text{ Mg CO}_2 \text{ ha}^{-1}$), a figure comparable to roughly 10%
132 of that emitted annually from land-use change, with associated economic costs approaching US\$1.9
133 to 13.7 billion yr^{-1} (Fourqurean et al. 2012; Pendleton et al. 2012).

134 **2.2 Regulating services: erosion and natural hazard regulation**

135 Coastal vegetated wetlands such as seagrass meadows can provide effective natural
136 protection from the destructive powers of storms and wave action (Barbier et al. 2008; Bouma and
137 Amos 2012; Duarte et al. 2013b). They are therefore important ecosystems to study in order to
138 understand the spatial extent, frequency, and risk of marine hazards affecting coastal waters and
139 how their effects can be minimized (ranked 35th in Rudd, 2014). Direct coastal protection is
140 achieved through energy dissipation resulting from wave breaking, friction and energy reflection
141 (Ondiviela et al. 2013), processes significantly influenced by seagrass shoot density and canopy
142 structure (Hansen and Reidenbach 2013). Even low biomass and heavily grazed seagrass meadows
143 can significantly reduce wave action by decreasing the hydrodynamic energy associated with
144 current flow (Christiansen et al. 2013). For example, in temperate regions current velocities have
145 been reduced by up to 60% in summer (high biomass) compared to 40% in winter (low biomass) in

146 relation to adjacent non-vegetated sites (Hansen and Reidenbach 2013). By reducing wave action
147 and current velocities seagrass habitats also protect the seafloor against hydrodynamic ‘shear
148 stresses’ (de Boer 2007).

149 Seagrass canopies act as efficient filters, stripping particles from the water column and
150 adding to sediment accumulation (Hendriks et al. 2007). Soil accretion ($\sim 1.5\text{mm yr}^{-1}$) is important
151 in helping coastal wetlands, and seagrass meadows in particular, adapt to sea level rise (Kirwan and
152 Magonigal 2013; Lavery et al. 2013), thus contributing to Rudd’s (2014) 26th ranked question on
153 sea level rise and vulnerable coasts. Below-ground seagrass biomass is particularly important for
154 sediment accretion as well as stabilization against storm erosion (Bos et al. 2007; Christiansen et al.
155 2013). By helping to immobilise sediment, seagrass meadows also reduce re-suspension and
156 increase water transparency (Duarte 2002; Ondiviela et al. 2013). In the Arabian Gulf, for example,
157 sediment stabilization and shoreline protection represent important ecosystem service functions of
158 seagrass meadows (Erftemeijer and Shuail 2012). Overall, the effectiveness and efficiency of the
159 coastal protection services provided by seagrass ecosystems varies across spatial and temporal
160 scales due to differences in species type (i.e. vegetative characteristics), coastal distribution, flow-
161 vegetation interactions and water dynamic properties (Ondiviela et al. 2013). In monetary terms, the
162 erosion control services provided by seagrass beds (inclusive of algal beds) have been estimated at
163 US\$25,000 $\text{ha}^{-1}\text{yr}^{-1}$ (Costanza et al. 2014).

164 **2.3 Provisioning services: biodiversity and fish nurseries**

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

181 **2.4 Supporting services: nutrient cycling**

182 Seagrass meadows are directly involved in nutrient cycling through their uptake of water
183 column nutrients, storage in biomass, detritus and sediment, and indirectly through the effect of
184 seagrass metabolism on water column and sediment nutrient re-cycling (Saenger et al. 2013). The
185 nutrient cycling capacity of seagrass meadows has been estimated to contribute about US\$26,000
186 $\text{ha}^{-1}\text{yr}^{-1}$, or US\$1.9 trillion in aggregate, to the global economy (Waycott et al. 2009; Costanza et al.
187 2014).

188 **2.5 Cultural services: social relations**

189 Wetland ecosystems play vital cultural, economic and ecological roles, supporting
190 livelihoods and reducing poverty (Verma and Negandhi 2011; Kumar et al. 2011; Senaratna
191 Sellamuttu et al. 2011). Frequently, the fish and marine invertebrate populations supported by intact
192 seagrass ecosystems maintain stocks of commercial and artisanal importance, and their exploitation
193 makes significant economic and food security contributions to many coastal communities (Jackson
194 et al., 2001). In some cases seagrass supported fisheries may provide a harvest value of up to
195 US\$3500 $\text{ha}^{-1}\text{yr}^{-1}$ (Waycott et al. 2009). In Tarut Bay, (Arabian Gulf), seagrass ecosystems support
196 a US\$22 million yr^{-1} fishery (Erftemeijer and Shuail 2012). Prawns are also the basis for extensive

197 fisheries, particularly along warm-temperate and tropical coastlines, and previous estimates of the
 198 potential total annual yield from seagrass ecosystems in Northern Queensland, (Australia), equated
 199 to a landed value of between AUS\$0.6 million to AUS\$2.2 million yr⁻¹ (Watson et al., 1993). In the
 200 Caribbean and Indo-Pacific region valuable species such as queen conch (*Euatomus gigas*), spiny
 201 lobster (Palinuridae), and smudgespot spinefoot (*Siganus canaliculatus*) also support local fisheries
 202 (Cullen-Unsworth and Unsworth 2013; Baker et al. 2014).

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

216 **3 The value of ecosystem services provided by seagrass ecosystems**

217 **3.1 Ecosystem service valuation**

218 Valuing ecosystem services has become an increasingly important tool for demonstrating
 219 the significance of biodiversity and ES to society and informing policy decisions (Gomez-
 220 Baggethun et al. 2010; Brondizio et al. 2010; Dendoncker et al. 2014; Liekens and De Nocker 2014).
 221 ES valuations have been criticized for focusing disproportionately on utilitarian values, overly
 222 commodifying nature and ignoring ecological complexity, potentially leading to erroneous policy
 223 decisions (Kosoy and Corbera, 2010; Norgarrd, 2010; Gowdy and Baveve, 2014). In light of these
 224 criticisms efforts to value ES have increasingly sought to focus on integrating the ecological, social
 225 and economic dimensions of ES into a unified whole (Figure S1 Supplementary Material) (TEEB,
 226 2010; UK NEA 2011; Dendoncker et al. 2014). A plethora of monetary and non-monetary
 227 techniques have recently been developed to try and capture the broadest range of 'values' across the
 228 breadth of ecosystem services (Table S1 Supplementary Material). Seagrass ecosystems provide a
 229 potentially tractable environment within which to conduct multi-faceted valuation research and
 230 address an important ocean research question (ranked 53rd, Rudd, 2014) on ecosystem service
 231 valuation implications.

232 **3.2 Seagrass and wetland valuation studies**

233 Economic valuations of seagrass ecosystems remain few in number, with most focusing on
 234 the market value of commercial fisheries as the primary ecosystem service of importance (Table 2).
 235

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 (<i>Sebastes cheni</i>) 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 (<i>Zostera marina</i>) 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 <i>Posidonia oceanica</i> 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 <i>Cymodocea nodosa</i> 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

236

237

238

239

240

241

242

243

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	<i>Coastal protection</i> : (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). <i>Maintenance of fisheries</i> : (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). <i>Carbon sequestration</i> : (US\$30.50 ha ⁻¹ yr ⁻¹ for saltmarshes and mangroves based on global sequestration rates). <i>Tourism, Recreation and Research</i> : (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	<i>Total Economic Valuation</i> : (US\$1092 ha ⁻¹ yr ⁻¹) e.g., <i>Fishery</i> : (US\$44 ha ⁻¹ yr ⁻¹) <i>Coastal protection</i> : (US\$91.7 ha ⁻¹ yr ⁻¹) <i>Carbon sequestration</i> : (US\$126 ha ⁻¹ yr ⁻¹) <i>Biodiversity</i> : (US\$5 ha ⁻¹ yr ⁻¹) <i>Existence value</i> : (US\$594.4 ha ⁻¹ yr ⁻¹)
Verma and Negandhi (2011)	Livelihood dependency and economic evaluation of the Bhopal wetland, India	<i>Fisheries production</i> : (US\$33 month ⁻¹ fisherman ⁻¹) <i>Boating activities</i> (US\$2264 yr ⁻¹ boatman ⁻¹) <i>waterchestnut cultivation</i> (US\$222 yr ⁻¹ family ⁻¹) <i>cloth washing activities</i> (US\$66 month ⁻¹ household ⁻¹) <i>secondary activities e.g., sugar cane juice sellers</i> (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	<i>Fisheries</i> : (US\$23,613 ha ⁻¹ yr ⁻¹). <i>Forestry</i> : (US\$38,115 ha ⁻¹ yr ⁻¹). <i>Recreation and Tourism</i> : (US\$37,927 ha ⁻¹ yr ⁻¹). <i>Non-Use</i> : (US\$17,373 ha ⁻¹ yr ⁻¹). <i>Water purification</i> : (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 <i>Flood control</i> : US\$6923(427) ha ⁻¹ yr ⁻¹ <i>Water supply</i> : US\$3398(57) ha ⁻¹ yr ⁻¹ <i>Nutrient cycling</i> : 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 <i>Therapeutic</i> : (14%-71%) <i>Amenity</i> : (65%-73%) <i>Heritage</i> : (70%-92%) <i>Spiritual</i> : (44%-76%) <i>Existence</i> : (89%-91%)
Kakuru et al. (2013)	Wetland ecosystem services in Uganda	<i>Flood control</i> : (US\$1.7 billion yr ⁻¹). <i>Water regulation</i> : (US\$7 million yr ⁻¹)

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

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

259 **4 Policy frameworks for Blue Carbon management**

260 Recent thinking about Blue Carbon acknowledges the special importance of the carbon
261 storage and sequestration capability of coastal and marine wetlands and organisms in global climate
262 change scenarios and policies (Sifleet et al. 2011; Vaidyanathan 2011). Blue Carbon sinks capture
263 and store amounts of carbon equivalent to up to half of global transport emissions (~ 400 Tg C yr⁻¹)
264 yet their inclusion in current mitigation and adaptation programs has been very limited (UNEP
265 2009; Tommaso et al., 2014). Developments could occur in the regulated (compliance) or the
266 unregulated (voluntary) carbon sectors. We take each in turn.

267 **4.1 The regulated sector**

268 **4.1.1 Policies and processes**

269 Collectively, the United Nations Framework Convention on Climate Change (UNFCCC
270 1992, Article 4 (d)), Manado Ocean Declaration (2009), Cancún Agreement (2010) and Rio Ocean
271 Declaration (2012) provide opportunities for development of Blue Carbon initiatives. In practice,
272 however, current policy processes inadequately account for the restoration and protection of Blue
273 Carbon systems (Grimsditch 2011; Murray et al. 2011). This is due, in part, to the initial bias
274 towards terrestrial climate change mitigation and adaptation activities within the UNFCCC,
275 alongside the acknowledgment that practical expansion to coastal and marine systems (from
276 principled intentions) would require further international agreement (Murray et al. 2012). However,
277 as a recent report indicates (UNEP and CIFOR 2014: x) ‘climate change mitigation frameworks
278 developed for terrestrial ecosystems can be extended to include coastal wetlands’.

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

288 **4.1.2 Kyoto protocol opportunities**

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

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

309 **4.1.3 Durban platform opportunities**

310 The Durban Platform provides more scope for Blue Carbon activities. Mangroves are now
311 covered by REDD+ (Grimsditch 2011). However, seagrass inclusion remains some way off: this
312 would require a broader definition of ‘forests’ as well as an extension of emission and reduction
313 activities across all land-uses (i.e., Agriculture, Forestry and Other Land Uses, AFOLU) (Murray et
314 al. 2011; 2012; Siikamaki et al. 2013). Nevertheless, AFOLU projects do include a variety of
315 carbon accounting protocols relating to biomass, C_{org} and greenhouse gas emissions (UNEP and
316 CIFOR 2014). There have been calls to decouple carbon sequestration and emissions arising from
317 habitat degradation (Grimsditch 2011). This is particularly important for seagrass meadows where
318 the ‘real’ carbon of interest is buried in the sediment. Under REDD+, deciding what aspects of the
319 Blue Carbon pool (i.e. sediment/soil-carbon or above-ground biomass) count would be especially

320 important (Murray et al. 2011). Extension of REDD+ to seagrass meadows could easily see them
 321 contributing to reduced emissions via the degradation pathway, through a focus on management
 322 strategies linked to tackling the negative impacts of nutrient loading for example (Seifert-Granzin
 323 2010). Developments to include tidal wetland restoration and conservation under REDD+ are
 324 currently on-going (UNEP and CIFOR 2014).

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

332 4.2 The voluntary sector

333 4.2.1 The global voluntary carbon market

334 The voluntary carbon market (VCM) accounts for 0.1% and 0.02% of the value and volume
 335 of the regulated global carbon market respectively (Benessaiah 2012). Yet rapid sector expansion
 336 has led to increasing interest from governments, particularly in relation to carbon standards and
 337 registries (Peters-Stanley and Yin 2013). The principal attraction of the VCM is its deregulated
 338 nature, which helps to reduce transaction costs and stimulate innovation. However, the trade-off to
 339 this regulatory flexibility is market uncertainty and depression of the carbon price, which can have
 340 serious implications for expected project returns (Benessaiah 2012; Thompson et al. 2014). Project
 341 size is also a determinant of offset price, with smaller projects garnering higher carbon prices for
 342 carbon dioxide equivalent (CO₂e). The average carbon price for micro projects (i.e., those
 343 generating less than 5 Kt CO₂e yr⁻¹) was recently US\$10/tCO₂e, whereas the mean carbon price for
 344 mega projects (i.e. those generating more than 1 Mt CO₂e yr⁻¹) was US\$5.8/tCO₂e (Peters-Stanley
 345 and Yin 2013).

346 Worldwide carbon standards have expanded from concentrating purely on carbon
 347 accounting to emphasising co-benefits (Peters-Stanley and Hamilton, 2012). This has been driven,
 348 particularly in the private sector, by an increasing interest in measuring and verifying non-carbon
 349 project outcomes (Peters-Stanley and Yin 2013). Programs are progressively focusing on climate
 350 change adaptation, public health, gender issues and biodiversity as additional attributes to non-
 351 carbon benefits (Peters-Stanley and Yin 2013) (Table 4). For example, the verified carbon standard
 352 (VCS), which accounts for 55% of market share, considers climate, community and biodiversity
 353 (16%) and Social Carbon (2%) co-benefits (Peters-Stanley and Yin 2013). This is important for
 354 ecosystems such as seagrass meadows that provide multiple benefits in addition to carbon storage as
 355 those benefits might be captured via broader standard attributes.

356
 357 **Table 4** Carbon standards appropriate for joint environmental and development projects

Carbon Standard and Credits	Description
Gold Standard (<i>acquired Carbon Fix Standard</i>)	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 will 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

Women's Carbon Standard	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. 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

358

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

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

386 **4.2.2 Multilateral environmental agreements**

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

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

396 **4.2.3 National level policies**

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

416 **4.2.4 Blue Carbon demonstration sites and the future**

417 Recent research, policy and financing advancements in Blue Carbon relevant to seagrass
418 meadows include global programs. The Blue Carbon Initiative (www.thebluecarboninitiative.org)
419 focused on integrating Blue Carbon activities into the UNFCCC and other carbon financing
420 mechanisms (Herr et al. 2012; Thomas 2014). Charities such as The Ocean Foundation and partners
421 (www.seagrassgrow.org) have developed a Blue Carbon calculator that determines CO₂ emission
422 reduction offsets in terms of the protection and restoration of seagrass meadows (a method pending
423 formal approval by the VCS). Collectively, developments such as the Blue Carbon portal
424 (www.bluecarbonportal.org) and work by Bredbenner (2013) and Thomas (2014) have
425 demonstrated the current global extent of Blue Carbon activities. In particular, significant work
426 remains to establish a functioning global network of fully implemented Blue Carbon programmes
427 involving the active transfer of carbon credits (Locatelli et al. 2014). In this regard, securing private
428 financing of Blue Carbon activities will become increasingly important (Thomas, 2014). Presently,
429 Blue Carbon programs are predominantly research-oriented, in the early stages of development and
430 mangrove-focused, with few directed efforts towards seagrass ecosystems (Table 5) (Bredbenner
431 2013).

432

433 **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 Catarina 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

434 Adapted from Bredbenner (2013)

435

436 **5 Seagrass habitats: prospects for PES**

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

440 **5.1 A brief explanation of PES**

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

458 Institutionally, PES programs are generally framed as decentralized instruments favouring
459 bottom-up solutions to land management issues (Landen-Mills and Porras 2002; Bond and Mayers,
460 2010). Despite the diversity of contexts in which PES schemes operate, they tend to adopt common
461 modes of activity such as restricting agricultural development, proposing alternative cropping
462 arrangements, reducing deforestation and expanding forests (e.g., reforestation and afforestation), or
463 protecting watershed and hydrological services (e.g., Aquith et al. 2008; Bennett 2008; Muñoz-Pina
464 et al. 2008; Porras 2010; World Bank 2010; Wunder and Alban 2008). Consequently, PES involves
465 multiple partners across sectors as well as spanning spatial and temporal scales (Schomers and
466 Matzdorf 2013). To function properly, schemes need to be acceptable to stakeholders, take the form
467 of contractual obligations to which all participating parties agree, have specified objectives, be
468 operationally transparent, and provide payments (in monetary or in-kind terms) to ES providers that
469 account for (ideally) the full range of their opportunity costs (Wunder et al. 2008; Hejnowicz et al.
470 2014).

471 **5.2 PES case studies and some considerations**

472 Examples relevant to guiding the development of seagrass payment schemes need to involve
473 community approaches to natural resource management, as well as the provision of multiple ES
474 with a focus on carbon management (e.g., Table S2 Supplementary Material). Schemes seeking to

475 deliver multiple ES via incentive mechanisms must also tackle the issue of stacking and bundling
476 (Box 1). That is to say, determining what ES are to be provided, whether they will be paid for
477 individually (i.e., stacked) or collectively (i.e., bundled), and what form payments will take (Bianco
478 2009). Additionally, PES programs need to ensure that as part of their design and implementation
479 they maximize biodiversity and social co-benefits by adopting a decoupled approach to benefit
480 maximization (recognizing individual ES properties and spatial attributes), ensuring management
481 decisions account for internal and external costs, and increasing social co-benefit provision by
482 concentrating on economic and cultural context (Greiner and Stanley 2013; Phelps et al. 2013; Potts
483 et al. 2013).

484 **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).

518 **5.3 Seagrass PES scheme options**

519 **5.3.1 Regulating fisheries and developing protected areas**

520 Many possible institutions are available to control and direct fishing activities along coasts
521 and marine ecosystems (Rudd 2004). They may involve fishing gear and net restrictions, limiting

522 fishing permits to local residents and restricting the exploitation of connected habitats while
 523 providing alternative income generating projects and ‘legal’ fishing equipment (e.g., Mnazi Bay
 524 Ruvuma Estuary Marine Park, Tanzania – Alberts et al. 2012; Mohammed 2012). Enforcing closed
 525 fishing seasons while providing wage supplements to fishers to offset opportunity costs resulting
 526 from deferred fishing activities is another approach (e.g., the defeso system in Brazil – Bergossi et
 527 al. 2011, Bergossi et al. 2012). Seagrass PES schemes may often involve creating marine protected
 528 areas (MPAs), safeguarding the underlying resource base supporting coastal communities and
 529 compensating local fishers for lost income resulting from harvesting restrictions (Table 6).
 530 Designating ‘no-take-zones’ to increase habitat cover and fish stocks, and compensating fishers for
 531 lost income is a strategy that some external non-governmental organisation (NGO) donors have
 532 used (e.g., Kuruwitu Conservation and Welfare Association in Kenya – Mohammed 2012). Setting
 533 up seagrass PES schemes requires research in a number of areas identified as priorities (Rudd
 534 2014), including the role of MPAs on ecological resilience (ranked 30th) and their effect on human
 535 well-being (ranked 45th). Questions regarding compliance with rules (ranked 58th) and the capacity
 536 of communities to manage their coasts (ranked 56th) also demonstrate the potential value of seagrass
 537 PES development beyond the sector, as programs provide valuable opportunities to learn broad
 538 lessons about the interactions between social and ecological systems.

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

Country	Project Summary
Ecuador, Galera-San Francisco Marine Area – operating since 2008	<ul style="list-style-type: none"> • 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	<ul style="list-style-type: none"> • Example of locally managed marine areas (of which 200 currently exist involving 300 communities covering 30% of inshore fisheries). • <i>Bio-prospecting</i>: 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). • <i>Live Rock Harvesting</i>: 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)	<ul style="list-style-type: none"> • 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	<ul style="list-style-type: none"> •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	<ul style="list-style-type: none"> •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.

542 Examples adapted from The Nature Conservancy’s Marine Conservation Agreements: Practitioner’s Toolkit
543 (<http://www.mcatoolkit.org/>)

544 **5.3.2 Ecotourism**

545 MPA managers and coastal businesses may establish “green” levies or taxes for resort
546 tourists and charge user-fees for diving access and licenses. Revenues generated by these charges
547 can be re-invested to support continued management activities to enforce the operating rules and
548 ensure compliance, conserve and restore seagrass beds, and create employment opportunities for
549 local community members (Lutz, 2011). In this respect, participation of the private sector can be
550 transformative for scheme development by acting as a powerful ally in conservation outreach,
551 providing new sources of financial support and creating employment and income opportunities
552 alongside appropriate public sector institutions (e.g., the Indonesian Yayasan Karang Lestari coral
553 restoration project and Marin tourism park on the island of Gili Trawangan – Bottema and Bush
554 2012).

555 **5.3.3 Linking farming, industry and watershed and coastal management**

556 Eutrophication and hypoxia resulting from nutrient loading and upland pollution are
557 significant threats to the health of seagrass ecosystems (Waycott et al. 2009; Short et al. 2011;
558 2014). Because upstream land-use activities can negatively affect seagrass ecosystems (Freeman
559 2008; Rivera-Guzmán et al. 2013) the conditions necessary for emulating watershed payment
560 schemes are ripe (Porrás et al. 2013). This may involve cross-sector collaborative partnerships
561 between local and international NGOs, who are often project initiators and intermediary facilitators,
562 working together with public utilities, private firms and government organisations acting as ES
563 buyers (Porrás et al. 2008; Schomers and Matzdorf 2013). Benefits to water quality and reduced
564 water treatment costs save public utilities and private firms significant financial outlays, which may
565 then be channelled into project start-up costs and payments for participants. Examples include the
566 equitable PES schemes for watershed services in Tanzania and Honduras (CARE 2009; Branca et
567 al. 2011; Kosoy et al. 2007). Collectively, these examples highlight the integrated nature of coastal
568 and terrestrial systems and demonstrate that PES schemes which acknowledge these interactions
569 begin to address Rudd’s (2014) questions on ‘upland hydrology effects on oceans’ and ‘integrated
570 upland coastal management’ ranked (24th) and (43rd) overall.

571

572

573 **5.3.4 Biodiversity conservation**

574 Many turtle populations nest in coastal regions supported by seagrass ecosystems (Cullen-
575 Unsworth and Unsworth 2013). These iconic and charismatic species are increasingly threatened by
576 poaching and so ensuring healthy nesting populations is vital (Koch et al. 2006). Protecting seagrass
577 ecosystems may be a cost-effective and financially viable option for sea turtle conservation. Paying
578 locals to monitor nesting sites and fisherman for releasing live turtles caught in fishing gear
579 provides a direct and additional income stream for local communities (Ferraro 2009; Mohammed
580 2012). In nesting projects locals usually receive two payments: a flat fee for identifying nest
581 locations; and a variable payment based on hatching success. Successful examples include Natamu
582 Turtle Watch and Knunga Marine National Reserve Conservation and Development Project in
583 Kenya and Sea Sense on Mafia Island in Tanzania (Ferraro 2009).

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

592 **5.3.5 Restoration**

593 Seagrass ecosystems are declining yearly (Unsworth 2014). To reverse this global trend
594 seagrass restoration (in suitable areas) offers an effective means to rehabilitate carbon stores and
595 sinks (Duarte et al. 2013c) whilst enhancing other equally important ecosystem services (Greiner et
596 al. 2013). A recent seagrass restoration CO₂ accumulation model, examining long-term trends in
597 carbon sequestration for several commonly planted seagrass species, demonstrated that at an
598 optimal density carbon accumulation of 177 to 1337 t CO₂ ha⁻¹ after 50 years could be achieved
599 (Duarte et al. 2013c). However, although seagrass restoration has a relatively long history,
600 particularly in the USA, it still remains limited in scope and success (Fonseca, 2011). Nevertheless,
601 the importance of restoration activities for coastal management has been highlighted by Rudd
602 (2014), with the ocean priority research question addressing 'restoration effectiveness' ranking
603 (29th). Restoration programs also provide opportunities to generate significant socio-economic
604 benefits.

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

620 The Swahili Seas Mikoko Pamoja project (2010-2013) provides a successful example of a
621 wetland restoration carbon finance program operating in a developing country context. Active in
622 Gazi Bay, Kenya, the Mikoko Pamoja project has established a mangrove conservation and

623 restoration program focused on the carbon storage value of mangroves to benefit poor coastal
624 communities. The program operates an accredited Plan Vivo carbon credit scheme providing
625 US\$13,000 annually, which is disbursed to conservation activities and community development
626 projects. Moreover, since 2012 one of the project partner's, Earthwatch Institute, has employed
627 local residents and volunteers to participate in mangrove management and restoration activities
628 covering over 600 hectares. Finally, the project has also engaged in a number of capacity building
629 initiatives through the provision of additional training and networking facilities (UNEP and CIFOR
630 2014).

631 **6 Possibilities for implementing seagrass conservation mechanisms**

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

643 **6.1 Institutions**

644 Effective institutions are crucial to the successful implementation of incentive schemes and
645 the resolution of coastal management problems (Rudd et al. 2003; Imperial 2005; Schomers and
646 Matzdorf 2013; Somorin et al. 2014). In the process of establishing effective institutions the
647 development of institutional flexibility is particularly important, as this enables programs to respond
648 adaptively over time to changing circumstances and thus maintain their efficacy (Larson and Soto
649 2008; Murdiyario et al. 2012). Securing institutional flexibility requires program arrangements that
650 foster active connections and relations between actors, strong leadership and feedbacks in learning
651 systems (Cox et al. 2011; Legrand et al. 2013; Geist and Howlett 2014).

652 In order to deliver these, programs need to be based on a platform of transparency,
653 accountability and inclusivity (Lockwood et al. 2010; Larsen et al. 2011; Ingram et al. 2014). These
654 aspects function as enabling properties, and the evidence clearly indicates that a lack of
655 transparency and accountability can seriously impair institutional capacity and effectiveness
656 (Somorin et al. 2014), whilst also undermining social capital (Rudd et al. 2003; Shiferaw et al.
657 2008). In addition, programs that fail to consider the issue of inclusivity can ultimately disempower
658 participant groups, and as a consequence, embed benefit sharing inequalities between households
659 and communities (Krause et al. 2013).

660 **6.2 Stakeholders and Participation**

661 Devolving decision-making to stakeholder groups can be enormously beneficial (Larson and
662 Soto 2008), at once enhancing and strengthening intra-community ties as well as a sense of
663 common identity (Rudd et al. 2003). Conversely, centralized administration can often stifle local-
664 scale innovations and the development of shared visions (Pokorny et al. 2013). Programs need to
665 engage and connect with local stakeholders in order to maximise participation, which is central to
666 providing effective management (Agrawal and Chhatre 2006). Doing so legitimises decision-
667 making and empowers individual and collective agency enabling the design process to align with,
668 and support, local norms, values and beliefs (Kawowski et al. 2011; Brooks et al. 2012; Corbera
669 2012; Bremer and Glavovic 2013). This is essential for participant commitment (Murdiyario et al.
670 2012; Davenport and Seekamp 2013) and acknowledges the relevance for effective governance of
671 local users' knowledge (Andersson et al. 2014).

672 These processes can be supported by clarifying stakeholder roles and responsibilities and
673 promoting leadership (Chhatre et al. 2012; Dent 2012). Leadership, and especially local leadership,

674 has been shown to be fundamental to delivering successful coastal management (Sutton and Rudd
675 2014). Finally, it is important to acknowledge how participation is framed in the context of power
676 relations, as these can represent potent forces capable of distorting the meaningful involvement,
677 agency and legitimacy of grassroots actors (Dewulf et al. 2011; Cook et al. 2013).

678 **6.3 Tenure and property rights**

679 Ownership in developing countries is often complicated by overlapping formal and informal
680 (customary) tenure and rights-based arrangements (Awono et al 2014; Resosudarmo et al 2014;
681 Rights and Resources 2014; Sunderlin et al. 2014). Clearly defining, legitimising and enabling
682 functioning property rights systems is essential for operationalizing incentive programs (Lockie
683 2013). Such clarifications are critical for conditional payments where knowing who to pay (i.e., the
684 right holder) and who is accountable for delivering project-level outcomes is necessary (Visseren-
685 Hamakers et al. 2012; Duchelle et al. 2014; Sunderlin et al. 2014). Functioning tenure and rights-
686 based systems also provide the framework to enforce property rights, securing contracts (Naughton-
687 Treves and Wendland 2014) and combating weak governance (Resosudarmo et al. 2014).

688 This is particularly pertinent to coastal marine environments where complications
689 concerning tenure, rights designations and authority are a direct challenge to introducing and
690 enforcing incentive schemes (Mohammed 2012), a state of affairs clearly linked to the ambiguities
691 regarding property rights in coastal areas and the variety of users and user interests (Cicin-Sain,
692 1993). As part of the design process it is crucial to mitigate potential mismatches arising between
693 the provision, delivery and bundle of property rights to reduce the likelihood of marine resource
694 conflicts developing (Yandle 2007), as well as to ensure that poorer sectors are not marginalised or
695 power asymmetries and social inequalities reinforced (WRI 2005; Fisher et al. 2008).

696 **6.4 Benefit sharing**

697 Distributing benefits and costs in a fair and equitable way is a fundamental aspect of
698 delivering socially acceptable incentive schemes (McDermott et al. 2012). Traditionally, equity
699 concerns have been side-lined in favour of a greater emphasis and focus on efficiency maximization
700 (Pascual et al. 2010; Narloch et al. 2013). However, this trade-off can produce socially undesirable
701 outcomes (Asquith et al. 2008). Incorporating social parameters in the targeting of schemes in order
702 to widen access and participation whilst reducing the marginalization of poorer communities
703 represents an important first step in reversing these potential trade-offs (Mahanty et al. 2011). These
704 processes need to proceed in tandem with beneficiary identification and the evaluation of the
705 potential socio-economic ramifications of ES provision and distribution (Willemen et al. 2013).
706 Additional considerations for effective benefit sharing include legitimising decision-making
707 processes via legal and procedural avenues (Murdiyario et al. 2012); adjusting compensation levels
708 according to the capacity needs of individuals, households and communities (Mohammed 2012);
709 and addressing the potential socio-economic impacts of programs on non-participants (Huang et al.
710 2009).

711 **6.5 Delivering ecosystem services, monitoring and compliance**

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

716 First, measuring ES provision (Porras et al. 2013). This reduces the likelihood of producing
717 a false picture of service provision, and provides a scientifically robust case for PES program design
718 (Hejnowicz et al. 2014). It has been suggested that even though coastal systems may be data poor,
719 there is sufficient knowledge of the management activities that improve resource protection and ES
720 provision (Lau, 2013). Second, evaluating scheme additionality and demonstrating ‘added value’ by
721 addressing the links between management interventions and program delivery (Ghazoul et al.
722 2010). Validating additionality requires baseline data, suitable metrics and performance indicators
723 plus the targeting of PES to locations likely to maximize program benefits (Sommerville et al.
724 2011; Wünschler and Engel, 2012; Lau, 2013).

725 Third, assessing potential of spill-over effects (i.e., leakage) resulting from program
726 implementation that may offset additionality gains (Engel et al. 2008; Porras et al. 2013). Fourth,
727 monitoring contract conditionality and ensuring compliance (Ferraro 2008). This requires
728 establishing who is monitoring (i.e. users, communities or officials) and how frequently
729 (Sommerville et al. 2011), providing sufficient payments to programme participants (Porras et al.
730 2013), and ensuring agreements are long-term arrangements with enforceable penalties for breaches
731 of contract (Ferraro 2008; Wunder et al. 2008). All have substantive effects on transaction costs of
732 governance (ranked 57th, Rudd 2014) and will influence the long-term viability of PES structures.

733 **6.6 Costs and funding**

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

746 **7 Conclusions**

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

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

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

776 mechanisms, we are also contributing to a broader set of significant global ocean priority research
777 challenges.

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

787 **8 Acknowledgements**

788 This article is based on research undertaken for ‘CESEA - Coastal Ecosystem Services in East
789 Africa NE/L001535/1research project’ and was funded with support from the Ecosystem Services
790 for Poverty Alleviation (ESPA) programme. The ESPA programme is funded by the Department for
791 International Development (DFID), the Economic and Social Research Council (ESRC) and the
792 Natural Environment Research Council (NERC). The first author (Adam P Hejnowicz) also wishes
793 to acknowledge the financial support of the ESRC and NERC (Ref: ES/I003851/1) for jointly
794 funding his PhD, which also helped to make this work possible.

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., Salyvonchik, 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., Okenwa, 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. Zografos, 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 leptocaula — 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.; Aguiñaga, 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., Marbà, N. (2013c). Assessing the CO₂ 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 Biogeochemical 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](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., Mwanjoka, 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](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.enviro.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* 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
- Muñoz-Piña, C.; Guevara, A.; Torres, J.M. & Braña, 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
- Murdiyarto, 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](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 (PINIC)*. 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
- Somarin, 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 Mediterranean: 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.)
- UNFCCC 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
- Willemsen, 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

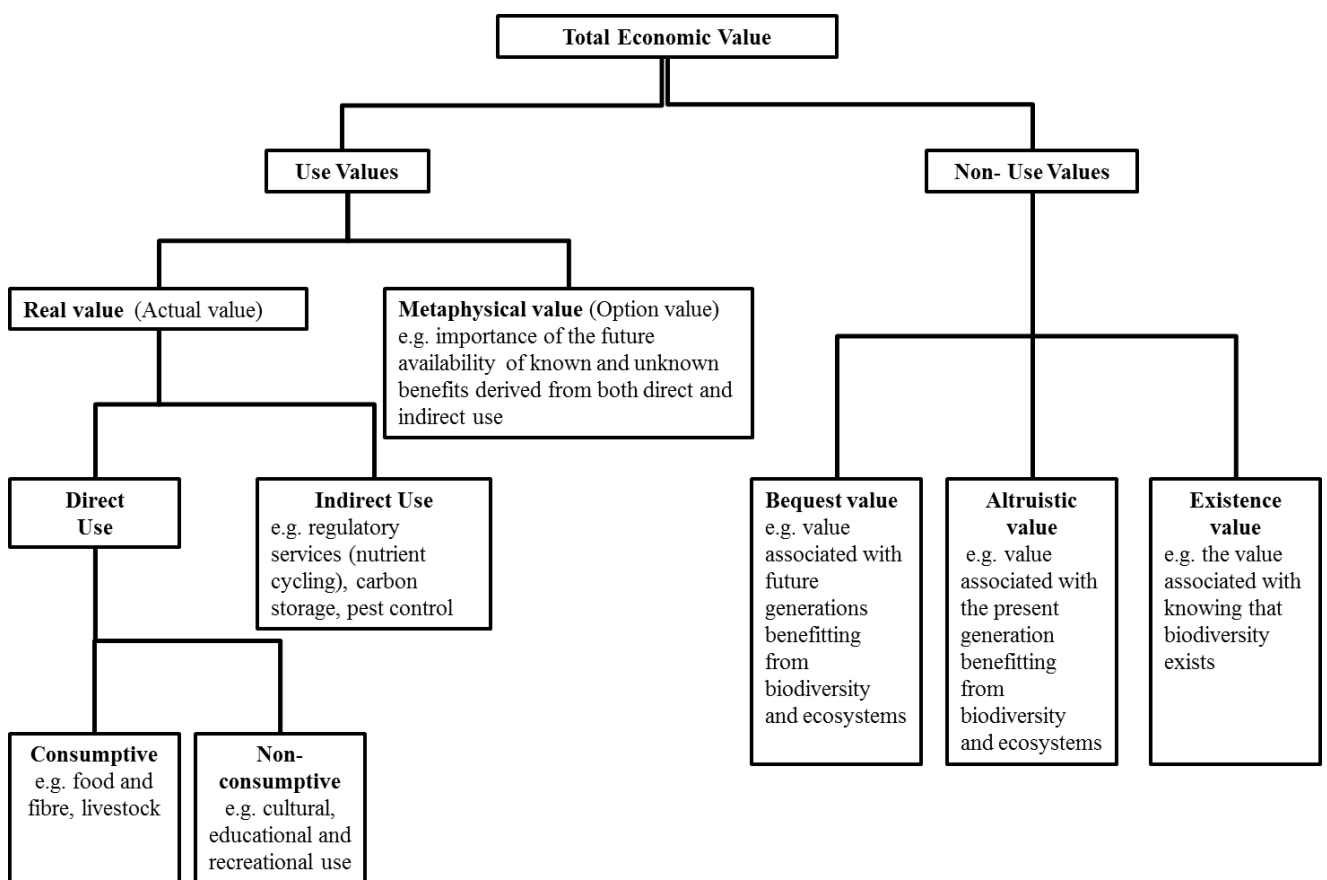


Figure S1. Diagrammatic representation of the ecosystem service economic valuation components and their relationship to each other. Adapted from Vo et al. (2012) and TEEB (2012)

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).	<i>Contingent Valuation</i> : 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.	<i>Choice Modelling</i> : 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 stated 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 <i>Source:</i> Hedge and Bull (2011) and Groom and Palmer (2012)	<p>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.</p> <p><i>Criticisms:</i> 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.</p>
Western Kenya Integrated Ecosystem Management Programme (WKIEMP): Kenya <i>Source:</i> World Bank (2010)	<p>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.</p> <p><i>Criticisms:</i> 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.</p>
Socio Bosque: Ecuador <i>Source:</i> de Koning et al. (2011) and Krasue and Loft (2013)	<p>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.</p> <p><i>Criticisms:</i> 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.</p>

