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TURNING THE TIDE: HOW PAYMENTS FOR ECOSYSTEM SERVICES (PES) MIGHT HELP SAVE MANGROVE FORESTS

1. INTRODUCTION:

Slowing and reversing tropical forest loss has long been a conservation priority. Traditional concerns over the loss of habitat have been amplified by a growing awareness of the role of forests in the global carbon cycle and as carbon sinks, with tropical deforestation accounting for 8-20% of anthropogenic CO_2 emissions (Solomon, 2007). Payments for ecosystem services (PES) schemes are emerging as new market-based approaches for forest conservation, with advocates hoping that they will address some of the underlying economic and political drivers of forest loss and provide direct economic incentives for conservation. Reduced Emissions from Deforestation and Degradation (or REDD+) are a set of international policies designed to compensate land owners for demonstrable reductions in forest-based carbon emissions. Whilst the REDD+ programs currently being developed and implemented in more than 40 countries often allow only marginal roles for local communities there are many opportunities for such projects to reflect principles of social justice and local control (Danielsen et al. 2013).

Mangrove forests should be leading candidates for such schemes. Despite their limited extent (approximately 0.7% of tropical forests) they are globally important carbon sinks because of their efficiency in carbon assimilation and below-ground storage (Donato et al., 2011). The gap between the economic value of intact mangrove ecosystems and the value captured by standard market economics (i.e. the market failure) is one of the widest for any ecosystem (Balmford et al., 2002). Mangroves are recognized as providing a wide-range of provisioning, regulating, supporting and cultural services that could be combined with carbon sequestration in marketing ‘high value’ carbon payments in putative PES projects. Because these services matter most to the poor – typically marginalized subsistence and artisanal fishers – small additional sources of income to local communities could reap major human welfare rewards (Barbier, 2006). Despite the well-documented ecological, economic and social benefits they provide, mangroves continue to suffer high rates of degradation and destruction, with global losses of 1-2% per annum exceeding those of terrestrial tropical forests (Spalding et al., 2010). Traditional conservation instruments appear insufficient and new approaches are required.

The large majority of PES forestry projects, either running or in development, concern terrestrial habitats (Warren-Rhodes et al., 2011). The recognition of the importance of coastal habitats as major carbon sinks has led to calls for 'blue carbon' to be considered under international agreements (Mcleod et al., 2011). Whilst one small mangrove-based PES project exists ('Mikoko Pamoja'; see www.eafpes.org) and larger ones are under development (including controversial cases such as in the Rufiji delta in Tanzania; Beymer-Farris and Bassett, 2011), considerable technical, social, political and economic barriers remain before PES can be applied widely to mangrove ecosystems (see for example Warren-Rhodes et al. (2011) on the potential for carbon-focused PES in the mangrove ecosystems of the Solomon Islands). Our aim here is to consider the potential for carbon-focused PES in mangroves and to explore some of the current and possible impediments and objections with a "from local to global" approach. Many of the scientific uncertainties specific to mangroves, concerning measurement of above and below-ground carbon and projections of yields under different scenarios, are discussed by Alongi (2011), whilst a global economic rationale based on carbon sequestration is given by Siikamäki et al. (2012). Hence we focus primarily on regulatory, market and social issues as well as on comparing mangroves as targets for carbon-focused PES with other forest types. Our decision to focus primarily on mangroves' potential for PES based on carbon storage and sequestration, rather than on the other services that they provide, reflects the current and likely future dominance of the carbon market as a source of revenue for mangrove conservation; this is particularly true in poor nations without obvious local markets for other services. Forestry projects continue to grow in importance in the Voluntary Carbon Market (Peters-Stanley and Yin, 2013; see section 3.1 below) and a "carbocentric" approach allows for comparison of benefits and risks with non-forestry carbon projects such as those centered on renewable energy sources (Wara, 2007). Carbon credits are already considered a powerful incentive for conservation and restoration of forest biomes in the developing world (Ebeling and Yasué, 2008). Although carbon is therefore the focal ecosystem service here, the challenges we address apply equally to other services such as fisheries provision and coastal protection.

We have three key objectives:

1) To compare the relevant biophysical characteristics, including vulnerability to natural hazards and provision of alternative ecosystem services, between mangroves and terrestrial forests in the context of their potential for PES, with a primary focus on carbon storage and sequestration.
2) To review the current options for trading in carbon and how these might relate to mangroves.
3) To consider issues of local control and environmental justice in PES schemes as pertaining to mangrove systems.

2. OBJECTIVE 1: BIOPHYSICAL CHARACTERISTICS

2.1 Vulnerability to natural hazards

Forests throughout the world are subject to biotic and abiotic disturbances. Estimating the risks these pose to forestry-based PES initiatives over the expected life-time of a project is a requirement for accreditation. At present this is very difficult for mangroves, partly because of the site-specific nature of most threats but also because of a lack of data that allow comparison of mangroves with other forests. Here we qualitatively compare the exposure to biophysical hazards of mangrove forests with terrestrial forests and plantations.

The main natural threats to forests worldwide are wind, snow, fire and pests, including insect outbreaks, bacterial and fungal pathogens (Hoffmann et al., 2003; Seidl et al., 2008). Like other forests, mangroves can suffer serious damage (Alongi, 2008; Cochard et al., 2008; Gilman et al., 2008) but their highly dynamic and resilient nature and peculiar physiology and location mean they differ from other forest types in susceptibility and response to particular threats (Alongi, 2008). Snow and fire, two of the largest sources of forest damage worldwide, are irrelevant to mangroves, whilst wave action and sea-level rise are uniquely pertinent.

Table 1

2.1.1 Wind

In temperate biomes, wind is the main abiotic hazard to forests (Hanewinkel et al., 2011). Wind damage to trees includes stem breakage and overturning, the probability of each event depending on tree, stand and soil characteristics, topography and forest management strategies (e. g. Nicoll et al., 2006).

Comparing wind damage between studies is difficult due to the different scales and units used, but it is nevertheless informative to report some figures across various areas. Wind damage to European forests has been extensive, with estimates of almost 19 million m^3 of timber lost annually in the second half of the 20^{th} century (Hanewinkel et al., 2011). The major storms that have recently hit Europe with increasing frequency have had particularly large impacts in some countries. For example the storm Lothar caused the loss of 200 million m^3 of European timber in 1999, mainly in central Europe (Blennow et al., 2010). In 2005, 75% of the 100 million m^3 of European timber losses occurred in Sweden, where the equivalent of a year's harvest was lost overnight (*ibid*.). Beyond Europe, New Zealand lost more than 8 million m^3 due to wind over the last half century (Moore and

Quine, 2000), whilst timber losses in Japan exceeding 30 million m^3 over five years were attributed to typhoon events (Kamimura and Shiraishi, 2007). The scale of wind damage in the US, particularly in those states affected by tornadoes and hurricanes, is similarly large. Hurricane Hugo in 1989 damaged almost 37 million m^3 of coastal forest timber in the State of South Carolina alone, whilst Hurricanes Katrina and Rita were responsible for an estimated 63 million m^3 of timber losses in the coastal forests of the Gulf of Mexico (Stanturf et al., 2007). In total, Hurricane Katrina produced timber losses equivalent to between 50 and 140% of US annual carbon sequestration (Galik and Jackson, 2009). In South America, carbon losses in the Manau region of the Brazilian Amazon forest after a single squall line event in 2005 were almost a quarter of the Amazonian mean annual carbon accumulation (Negron-Suarez et al., 2010). Whilst there are few African studies, Munishi and Chamshama (1994) report incidences of serious wind damage in a conifer plantation in Southern Tanzania, with percentages of damaged trees ranging between 25.7% and 40.4%. These studies demonstrate that wind damage is a major and widespread threat to terrestrial forests, particularly to upland conifer plantations and in hurricane affected areas, with single storm events having frequently destroyed more than 10% of a country's annual timber production.

The literature on wind damage to mangroves is much smaller than for terrestrial forests and is mainly concerned with their role in coastal protection (section 2.2.3). This relative paucity may indicate a smaller average risk but could also reflect the smaller total area of mangroves or a relative neglect of tropical coastal habitats in the literature. Due to their location the main wind threat to mangroves arises from coastal storms, typhoons and hurricanes. Most relevant work has focused on hurricane damage in the USA and Caribbean, where major storm events with a recurrence interval of around 30 years have been reported (Doyle et al., 1997). Cyclones in the Bay of Bengal show a similar average 29 year periodicity (Singh et al., 2000). Hurricanes and cyclones can certainly cause large-scale destruction of mangrove forests; Cahoon et al. (2003) cite papers showing that "powerful storms have caused mass mortality of at least 10 Caribbean mangrove forests during the past 50 years". However there is evidence that mangroves are more resistant and resilient compared with other forest types when exposed to the same storms. Following Hurricanes Frances and Jeanne in 2004, the area of mangroves that was disturbed was much smaller than that of other coastal forest types (~14 and ~95% respectively) in Florida (Vogt et al., 2011). After 4 ½ years, 51% of lost mangrove canopy cover had regenerated, compared with 2.4% in the other forests. Imbert et al. (1998) compared the effect of Hurricane Hugo (1989) on dense tropical, semi-deciduous tropical, and mangrove forests in Guadeloupe. Mangroves were the most affected, especially in their juvenile plants, but also the most efficient in terms of re-establishment of their population and basal area. Interspecific differences are found in mangroves' susceptibility to wind damage (e.g. Baldwin et al., 2001); this may contribute to their relatively high resilience and to a stronger tendency to post-hurricane community shifts (*ibid*.; Piou et al. 2006). Following Hurricane Wilma in 2005, mangrove sites in the Florida Everglades took 2 to 4 years to approximate pre-disturbance levels of albedo, CO_2 net fluxes and soil elevation (Barr et al., 2012).

The vast majority of studies on hurricane and typhoon damage to forests, including mangroves, come from North American, Caribbean and Asian sites. This reflects the locations where hurricanes' and typhoons' frequency and intensity are highest (Cochard et al., 2008). Investors in REDD+ and A/R projects may need to identify areas that are less prone to extreme events, especially in a changing climate. Recent model simulations predict a decline in the global frequency of hurricanes but an increase in intensity, with increasing damage in North America and Asia, a minor increase in Oceania, while Europe and Africa are not expected to experience any increase (Seneviratne et al., 2012). In summary, mangroves are probably less vulnerable than other forest types to any given wind speed, but their coastal habitat may expose them to particularly high winds from hurricanes. Hence mangroves in areas at low hurricane or cyclone risk are likely to be at lower risk from wind damage than other forest types.

2.1.2 Fire

Fire is the second major abiotic disturbance to temperate forests, being responsible for the annual loss of 0.5 million ha of forested land in the Mediterranean basin alone, and is related to latitude, local climate (e.g. wind, temperature and humidity) and forest management, with low levels of moisture in forests dramatically increasing the risk of fire (Cochrane, 2011). Numerous studies have discussed the increasing risk of fire damage in forests worldwide under IPCC climate change scenarios because of increasing predicted temperature (e.g. Hanewinkel et al., 2011). High relative moisture levels generally protect tropical rainforests from fire, although areas at the forest edges and heavily patched areas close to agricultural land and human settlements are at a higher risk (Hoffmann et al., 2003; Cochrane, 2011). There are no published reports of large scale fire damage in mangroves, presumably because of their permanently wet, and regularly inundated, soils.

2.1.3 Pests

Insect outbreaks and diseases caused by microbial and fungal pathogens are common to all forest types and are a major concern for forest managers; a large body of literature considers causes and remedies and their interactions with other abiotic disturbances (e.g. Hanewinkel et al., 2011).

Reports of mass tree death following total defoliation are common in terrestrial forests, particularly plantations. Such reports are much rarer from mangroves; we know of only three papers. In their study in Southwest Florida, Rehm and Humm (1973) reported a high incidence of wood-boring crustaceans feeding on prop roots of *Rhizophora mangle*, which were then affected by bacterial and fungal attack, causing a reduction in forest area and an increase in wind and wave damage. In their study of a small forest of *Avicennia marina* in Hong Kong, Anderson and Lee (1995) reported extensive damage to the mangroves' leaf area and flowers caused by a caterpillar. Whilst damage from folivores seems to be comparatively small in mangrove forests wood borers may have a much greater impact in natural systems. *R. mangle* forests in Belize can suffer more than 50% canopy damage from wood boring insects, with important implications for small scale gap formation and ecosystem dynamics (Feller, 2002). Such impacts may be under-recorded since arthropod damage to the stems, branches and roots is harder to detect than folivory. However the current paucity of reports of large scale tree death or defoliation resulting from pest infestation in mangroves, in comparison with other forest types, does suggest that this risk is relatively smaller.

2.1.4 Sea-level rise

Mangroves are the forest type at greatest risk from sea level rise. They may adapt by shifting further inland, but this will only be possible in areas where human settlements and agriculture occur at some distance from the coastline (Gilman et al., 2008). Alternatively they may maintain surface elevation through soil building and sediment accretion, but such a response requires vigorous growth and a good supply of sediment (Kumara et al., 2010). Where adaptation is impossible the habitat available to mangrove forests will shrink and the remaining forest may become less ecologically resistant and/or resilient (Alongi, 2008). Soil quality, salinity levels, and the tolerance and reproductive quality of particular mangrove species are expected to influence colonization patterns (Alongi, 2008).

2.2 ECOSYSTEM SERVICES PROVIDED BY MANGROVES

Mangroves provide an extensive range of ecosystem services in addition to carbon sequestration, including nutrient cycling, water purification, provision of nursery habitats, coastal erosion control, moderation of extreme events and biodiversity reserves (Ruitenbeek, 1994; UNEP-WCMC, 2006; Naber et al., 2008). There are therefore many opportunities for PES schemes to market “high value” carbon credits which reflect these additional services. As well as documented examples, nursery areas for fisheries, water treatment, and coastal protection are discussed here.

2.2.1 Fisheries services

By providing a refuge from predators and, in some cases, a feeding ground for juveniles, mangroves support coastal fisheries for fish and shrimp (e.g. Rönnbäck, 1999). Kenya represents a fitting example, as most families of commercial species are present in Kenyan mangroves and mangrovefringed habitats (Kimani et al., 1996). Overall fish biomass production estimates for mangroves range from 8.2 t km^{-2} yr^{-1} for Queensland in Australia (Blaber et al., 1989) to 13.26 t km^{-2} yr^{-1} in Florida (Thayer et al., 1987). The fisheries value of mangroves has been estimated in various regions of the World and shows high values that compare well with most productive ecosystems, such as coral reefs: 2,800 USD km^{-2} yr^{-1} in Belize (Cooper et al., 2009), 7,800 USD km^{-2} yr^{-1} in Philippines (Janssen and Padilla, 1996), 8,300 USD km^{-2} yr^{-1} in Cambodia (Bann, 1997) and about 20,000 USD km^{-2} yr^{-1} in Indonesia (Ruitenbeek, 1994). A review of the size and value of commercial and subsistence fisheries in mangrove areas can be found in Walters et al. (2008).

The sale of local fishing licenses could help finance conservation actions and regulate access to mangrove areas. However, because a substantial part of fishing by local populations is subsistence fishing, this opportunity needs to be further explored in order to assess the social and economic costs and ecological benefits of such PES schemes. Rather, the commercial exploitation of offshore fisheries of species that spend part of their life cycle in mangroves is more likely to be a source of PES. In the case of Kenyan EEZ fisheries, this link could lead to the establishment of PES for an increase of fishing opportunities to be paid by shrimp fishing companies. Currently in Kenya, the community based Beach Management Units charge a small levy for every kilo of fish landed in their beach. The funds are used to construct fish landing spots as well as pay fish scouts who survey illegal fishing activities. In Tanzania on the other hand, the Marine Legacy Fund of Tanzania is revenue derived from commercial fishing licenses and paid to coastal communities to protect mangroves and other key habitats (Ruitenbeek et al., 2005).

2.2.2 Water and waste treatment services

Mangroves are able to assimilate pollutants such as heavy metals (Lacerda and Abrao, 1984), nutrients (in particular nitrogen and phosphorus) as well as suspended solids (UNEP-WCMC, 2006), playing an important role in coastal water purification and waste water treatment, and preventing pollutants of terrestrial origins from reaching deeper waters (Tann and Wong, 1999). The biofiltering value of mangroves is estimated to range between US$ 1193 $ha^{-1}year^{-1}$ and US$5820 $ha^{-1}year^{-1}$ (Walters et al., 2008). Biophysical and ecological properties of mangrove trees and their associated soils and invertebrate communities contribute to these processes.

While the coastal communities that benefit from mangroves' water and waste treatment are unlikely to financially contribute to PES schemes, commercial activities – including shrimp farms and tourism - that require good quality water may voluntarily adhere to such PES to replace or avoid costly artificial systems such as water purification plants, resanding of beaches and water filters for aquaculture. One example concerns the Bonaire Marine Park in the Netherlands Antilles (Thur, 2010), where mangroves' contribution to water treatment is recognized through payment for protection from divers' entrance fees.

2.2.3 Coastal Protection

The idea that mangroves are effective in protecting coastal areas from extreme climatic events such as tsunamis and typhoons came into prominence after the 2004 tsunami that devastatingly hit Asia, although a review of 4 widely-cited post-disaster studies shows that the contribution of mangroves to coastal protection in the specific event depended on factors such as species composition, site conditions, geographical location, depth of the mangrove belt, and health of the broader seagrass beds – mangroves – coral reefs ecosystem (Cochard et al., 2008). The intensity of the 2004 tsunami was such that little protection could have been provided to the areas worst affected. Afforestation and effective management programs in mangrove stands in Bangladesh and Vietnam have effectively reduced the costs of human-made protective structures such as sea dykes (*ibid.*). Indeed, local populations, whose ecological knowledge has been proposed as a vital component of sound management practices (Walters et al., 2008), have historically planted mangroves to protect their coastlines and stimulate sediment accretion (Cochard et al., 2008; Walters et al., 2008).

Mangroves therefore offer considerable potential for the marketing of ‘bundled’ ecosystem services. One limitation to this approach might be trade-offs; maximizing one service may diminish another. Mangroves offer considerable advantages over terrestrial forests in this regard. In terrestrial forests maximizing carbon sequestration can lead to soil salinization, acidification and reduced stream-flow (Jackson et al., 2005); none of these negative impacts come from mangroves.

Figure 1

3. Objective 2: Review of current options for carbon trading

3.1. CARBON MARKETS AND RELEVANCE FOR MANGROVES

The forest carbon market is split between compliance schemes (created and regulated by mandatory national and international agreements) and voluntary projects, in which companies and individuals choose to invest in carbon offsets. The development of regulatory frameworks has driven a fast expansion in the global carbon market which increased from 11×10^9 USD in 2005 to 141.9×10^9 USD in 2010 (Linacre et al., 2011). Hence there is enormous and growing potential to marshal funds into mitigation projects, including those concerning forests.

However, forest credits are ineligible under the largest compliant trading scheme, the European Union Emissions Trading System (EU-ETS). While forests credits (for afforestation and reforestation – A/R - projects) are permitted within the Kyoto Protocol’s Clean Development Mechanism (CDM), they have remained marginal. In 2009, only 0.2% of the total portfolio (4 out of 1665 registered projects) was for A/R projects, representing a paltry 177.6 Million USD (Diaz et al., 2011), and none of these concerned mangroves. Key impediments to investment have been the cumbersome bureaucracy and the risks of impermanence associated with CDM forest credits. The failure of the compliance market to account for forest emissions has led to more than 90% of forest carbon projects pursuing certification under the voluntary market instead (Morrison and Aubrey, 2010).

The total voluntary market, recently valued at 523.0.0 Million USD (Peters-Stanley and Yin. 2013), is an order of magnitude smaller than the compliance market, but forestry projects figure prominently within it: circa 21% of market share is taken up by A/R, REDD or avoided conversion projects (OTC values from 2012, Peters-Stanley and Yin, 2013). Addressing climate change is becoming of increased importance for the corporate sector (Patenaude, 2010) and the success of forest projects is partly due to their attraction as high profile examples of corporate social responsibility. In the voluntary carbon market, the private sector is responsible for 70% of market activity (Peters-Stanley et al. 2013). Forest credits are not only visually compelling but are also much easier to communicate than other types of credits. The top motivations behind corporate purchase of forestry credits include an interest in communicating the social and environmental benefits that these projects generate, the extent of deforestation, and the tangibility of carbon storage in tree biomass (Waage and Hamilton, 2011).

The voluntary market provides the flexibility to develop, test and implement new approaches to carbon accreditation. The most important of these alternative mechanisms is REDD+ (Lederer, 2011). This allows the recognition of (and payments for) existing carbon, in contrast to A/R schemes which require change in land use from non-forest to forested land. Hence REDD+ could stimulate the sustainable management of current forests and allow rapid payments to local people (without the uncertainties involved in awaiting tree growth). This is relevant to mangroves where up to 90% of the carbon is stored below-ground in soils. Hence the removal of mangroves may cause the rapid release of large volumes of soil carbon, whilst new plantations will assimilate carbon at much slower rates. In 2011, REDD+ projects accounted for 29% of credits transacted in the voluntary carbon market – a significant increase from the 7% observed the previous year (Peters-Stanley et al. 2011).

The nineteenth Kyoto process ‘Conference of the Parties’ (COP19) delivered some progress in the design of a framework for REDD+ action, including an agreement for tropical countries to receive financing for both readiness and results on REDD+. REDD+ will figure prominently in the 2015 global agreement on climate change which is planned to come into force in 2020. Other nascent compliant markets, such as California's compliant cap and trade take onboard REDD projects. Most observers believe that the inclusion of REDD+ into the compliance markets is necessary before carbon payments have a real chance of addressing global forest losses. As Olander and Ebeling (2011) put it: 'Let's face it, forest carbon markets will remain small, and limited to voluntary markets, until large emitters are allowed to purchase large amounts of forest carbon offsets from around the world to meet mandatory emission reduction targets'. Whilst this is probably true, it does not preclude carbon markets playing a significant role in mangrove conservation even if they are limited to voluntary schemes. The exceptional efficiency of carbon sequestration and storage combined with multiple other ecosystem services provided by mangroves make them particularly well fitted for multiple small scale schemes that, in aggregate, make a global difference.

Realizing this potential for voluntary investment in mangroves, and building the evidence and arguments for the inclusion of mangroves in compliance schemes, requires the development of methodologies and approaches suited to these ecosystems – 'off the shelf' approaches using methods developed for large terrestrial forests often do not accommodate the special biological and social features of mangroves and often involve start-up costs well beyond the means of small scale projects. The voluntary carbon market is proving a fertile testing ground for new approaches: there are already more than 14 standards within the forestry sector. Sophisticated approaches to address the issue of non-permanence of forest ecosystems have been developed, including buffers and insurance products. Hence the next steps in developing mangrove carbon markets are likely to emerge from voluntary schemes.

3.2. FOREST STANDARDS

3.2.1 Accreditation challenges common to all forests

All carbon accreditation projects must demonstrate three characteristics: *additionality* – the carbon sequestered (or saved from emission) must be additional to what would have been achieved under a 'business as usual' scenario; *permanence* – the carbon stored (or saved from emission) should remain so over long time scales (that is, the risk that a forest planted or protected today may be destroyed or degraded tomorrow); *leakage* – the carbon sequestered (or saved from emissions) should not lead to an unforeseen increase or decrease of Greenhouse gases (GHG) emissions outwith the project's boundaries, these being either geographical or operational (Watson et al., 2000). Although these requirements apply to all accredited projects the last two are usually considered to be particularly challenging for forestry schemes. Two approaches to addressing impermanence include insurance products and risk buffers. The risk of impermanence in mangrove schemes is arguably lower than that in other forest types given the importance of refractory below-ground carbon – which might be stored for millennia - and the nature of the biophysical risks experienced as described in Section 2. Addressing leakage, however, remains a major challenge for putative mangrove projects. A comprehensive review of various approaches to dealing with impermanence in forests can be found in Murray and Olander (2008).

Any carbon offsetting project is subject to the risk of leakage although this is often perceived to be higher for forestry schemes (Kindermann et al., 2008) due to the general lack of forestry data compared to that available for other sectors (Wunder, 2008). Monitoring leakage is complicated and has been thoroughly calculated only in the case study of the Noel Kempff Mercado National Park in Bolivia (Sohngen and Brown, 2004). A shift in activities releasing GHG to the atmosphere can happen at various scales, from local, to national, to international (Edwards et al., 2010) and can also happen between sectors, such as when forest products are substituted with others produced with processes not limited by GHG caps (Kindermann et al., 2008). Leakage at national and international scales cannot be currently accounted for. Most REDD+ schemes are being implemented at the project-rather than national -level (Edwards et al., 2010), and while increasing the scale of a project would likely reduce the probability of leakage, it would also increase the overall costs.

3.2.2 Implications for mangroves

While issues of permanence are similar between terrestrial forests and mangroves, the generally smaller scale of mangrove projects implies that some approaches suitable for terrestrial forests may not be suitable for mangroves. For instance, larger schemes proposed to reduce leakage will reduce the chances of small-scale community-based mangrove projects - often in densely populated areas that deal with multiple users and stakeholders –achieving accreditation. Leakage presents additional challenges for the establishment of mangrove-based REDD+ projects. A/R projects provide carbon benefits without displacing local communities, due to the fact that they are generally established on degraded land, while reduced deforestation projects prevent land-use changes (Kindermann et al., 2008). As a consequence, the provision of a number of forest products is prevented; for example less timber production could result in an increase in prices and the promotion of logging in other areas or countries. An efficient mitigation strategy would be to combine REDD+ and A/R practices within a project, so as to prevent the displacement of emissions (Wunder, 2008) such as in the Ban Sam Chong Tai village in Southern Thailand, where tree planting and forest protection have proven successful in protecting mangroves by combining community involvement and setting harvesting rules (Barbier and Cox, 2004).

Figure 2

The avoidance and management of leakage is and will remain a significant barrier for most mangrove schemes. Various certification schemes take different approaches to dealing with anticipated leakage, with forest carbon projects required to develop risk profiles of leakage during the design stage (Galik and Jackson 2009). Leakage-avoiding activities can be designed that deal with the issue spatially and/or temporally (Ewers and Rodrigues 2008). Typically a review of current forest use in the project area and identification of ways to mitigate this is required. These might include timber plantations, fuel swappages (where use of biomass for cooking is a driver of deforestation) and the implementation of alternative livelihood projects. A key issue in addressing leakage is improving the governance and local ownership of a project; this is particularly pertinent to mangroves since these are generally collectively owned and managed.

Achieving high confidence that no leakage will occur before the start of most projects is unlikely. However, such uncertainty can be accounted for through mechanisms such as applying discounts according to the level of risk. A common route is the allocation of a percentage of credits into a buffer, or reserve account. This acts as an insurance policy against unforeseen losses of carbon stocks (Plan Vivo 2012; VCS 2012). Hence the problem of leakage in mangrove projects is not insuperable, although much useful further work could be done on methods of estimating and predicting risk which could provide simple, cheap and credible criteria for project developers to apply.

Table 2

4. OBJECTIVE 3: LOCAL CONTROL AND ENVIRONMENTAL JUSTICE

4.1 LAND TENURE, COMMUNAL MANAGEMENT AND PES

Natural resource rights and access frequently underpin the livelihoods of the rural poor in ‘developing’ country contexts, including most of those relying on mangrove ecosystems (Warren-Rhodes et al., 2011). As such, the potential transformation of these rights through REDD+ and wider PES schemes are critical issues in shaping prospects not only for biodiversity conservation, but also for environmental justice and poverty/well-being. In most cases mangrove PES projects will be located on land which is collectively owned or controlled. Recent work in the Solomon Islands highlights the complexity and diversity of communal tenure arrangements in mangroves, even between adjacent villages (Warren-Rhodes et al, 2011). Kenya provides another typically complex example. Here, officially landless 'squatters' are widespread on government owned land in coastal areas, albeit often being located on their own former customary or traditional lands. *De facto* as distinct from *de jure* practices illustrate complex and creative responses amongst local communities, including land renting, leasing and sub leasing by official or unofficial 'owners', tree rental and maintenance of communal use and access rights on *de jure* state owned land (Yahya and Swazuri, 2007). Thus in coastal areas, as elsewhere in Kenya, access to land and resources typically relies on complex formal and informal rights determined in some instances through formal land title, but more often through locally variable claims to traditional rights and usage, entitlements and identity, operationalized through social networks. Recent developments in Kenya, notably the Community Land Bill currently under debate in parliament, may reshape and clarify access and entitlements in the future, although the precise nature of impacts remain uncertain at present.

Existing complex communal management and tenure arrangements present undeniably greater challenges for PES schemes than those found on privately owned or leased land. Options for dealing with this complexity include the privatization (temporary or permanent) of land or benefits, or the development of effective mechanisms for collective sharing of benefits under the continuation of communal arrangements. Arguments for individualization of land tenure are often informed by colonial and post-colonial critiques of communal tenure and the assumed primacy of private, individual land ownership (Peters, 2009). Much recent scholarship has challenged such beliefs, for example through analysis of the often highly inequitable outcomes of land titling and privatization, attendant conflicts and poverty (*ibid*). Commons scholarship has also done much to highlight the efficacy of communal resource management (e.g. Agrawal 2001). However, communal management and tenure is not immune to the critiques often leveled at land privatization programs; many communal systems are inherently inequitable, often on grounds of gender, ethnicity and tribal/ political affiliation (Peters, 2009). One key challenge for mangrove PES schemes will be how to foster genuinely equitable, fair and sustainable programs for resource management and benefit sharing under communal tenure arrangements. Another may be to recognize that local social and resource management/ tenure complexities may render PES schemes inappropriate in certain cases. 'Local participation' in PES schemes is increasingly highlighted as means to redress early problems, but is not a panacea and merits further examination, as do concepts of environmental justice in PES (Martin *et al*., 2013; Suiseeya and Caplow, 2013).

4.2 LOCAL INVOLVEMENT: ENVIRONMENTAL JUSTICE, PARTICIPATION AND PES

Where new economic values of resources, including land, come into play, institutional transformations can move towards more exclusionary, inflexible access arrangements, often to the detriment of poor local people. In recent analyses of global land grabs, biodiversity conservation and reforestation, including through REDD and comparable activities, often feature as well as more familiar 'culprits' such as cultivation of biofuels (Vermeulen and Cotula, 2010). Key considerations include changes in inter-household power relations, norms of inclusion and resource rights in participant communities, often driven by intensified resource commodification and the need for clear, equitable 'rules of engagement' (*ibid*, Peters, 2009). Questions have also been raised about the extent and nature of community consultation, with common problems including nominal local participation and consultation only/ primarily with elites, underscored by external assumptions about representation and homogeneity of communities (Suiseeya and Caplow, 2013, Vermeulen and Cotula, 2010). Such issues necessarily have implications for legitimacy and for equitable sharing of benefits over the longer term.

An environmental justice framing offers valuable insights into these various issues, as they apply to PES schemes. Contemporary scholarship emphasises the trivalent nature of environmental justice, encompassing not only concerns with distributive justice (resource rights and access) but also procedural justice and recognition. These latter dimensions denote the importance of full, fair participation in decision-making by affected parties and the acceptance and recognition of diverse values, knowledges and cultural identities therein, not least in relation to PES (e.g. Beymer-Farriss and Bassett, 2012; Suiseeya and Caplow, 2013).

With particular reference to carbon sequestration projects, Jindal *et al*. (2008) concur that typically insecure land tenure for rural African communities enhances risks of their disenfranchisement in the face of outside investment. Where clear, formal recognition of customary or group rights is lacking, evidence from East Africa suggests that prospects of increased value through carbon sequestration may prompt land seizure by powerful local elites (*ibid*). Thus distributive injustice may be enhanced. Other concerns include high transaction and opportunity costs of PES projects amongst community groups, initial investment barriers for poorer households, inequitable sharing of benefits and long term lock-in to contracts, which may not always be fully understood by local participants (*ibid*). Again, these highlight prospects for distributive injustice, but also suggest procedural injustice, where local participants are not full participants and partners in the development of PES projects (Suiseeya and Caplow, 2013). Common recommendations for reductions in transaction costs include the creation/ support of appropriate community groups who can act as managers and/or intermediaries in the processes of implementation and supervision of projects. Unfortunately, such recommendations often fail to take into account intra-group inequalities and prospects for elite capture now widely recognized in other aspects of 'commons' and devolution literature and increasingly highlighted in justice-based analyses of PES projects (Agrawal, 2001; Beymer-Farriss and Bassett, 2012; Suiseeya and Caplow, 2013). Thus, while Jindal *et al.* (2008) argue the case for suitable institutional capacity at a national scale, there is an equally pressing need at the local level in order to mediate against distributive and procedural injustices. A further issue which merits attention is heterogeneity in knowledge and values amongst stakeholders (Warren-Rhodes *et al*., 2011). Where contemporary PES interventions are attempting to assign value to aspects of ecosystem services, the need to incorporate multiple dimensions of knowledge and value becomes particularly pressing, in accordance with the demands of procedural justice and of recognition.

Case studies of community-based management of mangroves are rare, while those addressing aspects of PES in mangroves are even more elusive. However, common strands include the frequently observed lack of sustainability of externally formulated institutional arrangements where these are unfamiliar in local contexts. Describing donor-driven interventions in mangrove forests in Zanzibar, Saunders *et al*. (2010) note the destabilization of preexisting institutional arrangements and the creation of a new elite within the village, comprising those closely engaged with the donor project. This proved to be a driver of conflict and dissent and contributed to the ultimate failure of the project suggesting the need for practitioners to engage more closely with lessons on group formation and community resource management (e.g. Agrawal, 2001) and with issues of procedural environmental justice, for practical as well as ethical reasons (Suiseeya and Caplow, 2013). According to Beymer-Farris and Bassett's (2012) controversial study of a REDD+ project in mangrove forests in Tanzania, recognition as an aspect of justice is critical, where imposed environmental narratives obscure local knowledges and ultimately produce distributive injustices through dispossession. Overall, best practice in PES schemes, including in mangrove environments, indicates the need for attention to the three often mutually constitutive dimensions of environmental justice; distribution, procedure and recognition. Increasingly, contemporary research highlights procedural justice as integral to the legitimacy and long-term sustainability of PES projects and as a route to, or even pre-requisite for, distributional justice (Suiseeya and Caplow, 2013). Effective, meaningful participation of all affected actors thus becomes central. Furthermore, as Martin *et al*. (2013:10) remind us, what is considered to constitute justice (in relation to distribution, procedure and recognition) may in itself be locally specific and contrary to global norms; in other words 'context matters'. Even as justice concerns are admitted in PES design and implementation, success may be confounded where different perceptions and meanings of justice are ignored (*ibid*). In practical terms therefore attention to claims *about* justice as well as claims *to* justice emerge as critical to future development of PES in mangroves, to be realised through inclusive, flexible and adaptive engagement between all stakeholders (ibid).

5. CONCLUSIONS

In this review paper we have shown that PES schemes have generally ignored mangroves; we argue that this reflects a traditional bias towards large scale terrestrial systems rather than any inherent unsuitability of these forests. In fact, mangroves offer important attractions for PES projects. First, their potential as carbon sinks is well documented to exceed most terrestrial forests. Specific to mangroves is the amount of carbon stored below ground (Yee, 2010; Donato et al., 2011). This characteristic makes mangrove forests uniquely important and suited to avoided deforestation projects. Second, mangroves compare well against other forest types in terms of their susceptibility to damage from biophysical hazards. A notable exception, peculiar to mangroves due to their distribution in coastal habitats, is sea level rise, although flourishing mangrove forests can help in coastal protection and adaptation to rising sea levels. Third, mangroves' provision of ecosystem services (ES) is extensive, the most notable examples, other than carbon sequestration, being the supply of nursery areas for fish, water purification, provision of wood products, and coastal protection (eg, UNEP-WCMC, 2006). Beneficiaries of such ES are not restricted to local communities (Ruitenbeek *et al*., 2005), but rather extend to national and international levels (Thur, 2010). Whilst trade-offs between the supply of provisioning and regulating services must occur in any forest, trade-offs between different regulating services (such as carbon sequestration and fresh water regulation) are more common in terrestrial systems. Fourth, many coastal communities, amongst the world's poorest, rely heavily on mangroves; hence mangrove conservation can underpin human welfare.

The case for developing mangrove PES projects is therefore strong. Most of the difficulties in doing so are shared by any work devoted to establishing sustainable forestry projects in developing countries which respect the needs and aspirations of local communities whilst responding to international markets. However characteristics of mangroves make issues of governance, environmental justice and policy particularly important. The collective ownership of land typical for mangroves requires communal resource management, which needs to be clearly established early in a project. In most countries where mangroves grow, governance at national and local levels is weak, unstable and prone to inequitable resource sharing. This means clear understandings of benefit sharing that are locally supported are essential; since injustice based on gender or affiliation to local groups may traditionally exist, such negotiated benefit sharing may have to challenge local elites.

Like the forests themselves, a good mangrove PES project is well adapted to local conditions. Whilst the current exclusion of REDD+ projects from the compliance market has precluded many large scale mangrove schemes, this allows the space for smaller voluntary projects to lead the way and show good practice. As the carbon market expands the opportunity exists to change the fortunes of mangrove ecosystems; the challenge is to do this for the benefit of local people as well as for the global climate.

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