Applying Climate Compatible Development and Economic Valuation to Coastal Management: A Case Study of Kenya's Mangrove Forests

Article *in* Journal of Environmental Management · April 2015

DOI: 10.1016/j.jenvman.2015.04.018

Journal of Environmental Management 157 (2015) 168–181

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Applying Climate Compatible Development and economic valuation to coastal management: A case study of Kenya's mangrove forests

Mark Huxham [a,*], Lucy Emerton [b], James Kairo [c], Fridah Munyi [c], Hassan Abdirizak [c], Tabitha Muriuki [c], Fiona Nunan [d], Robert A. Briers [a]

[a] School of Life, Sport and Social Sciences, Edinburgh Napier University, Edinburgh, Scotland EH11 4BN, UK
[b] Environmental Management Group, 15 Havelock Road, Colombo 5, Sri Lanka
[c] Kenya Marine and Fisheries Research Institute, PO Box 81651, Mombasa, Kenya
[d] International Development Department, University of Birmingham, Edgbaston, Birmingham B15 2TT, UK

A R T I C L E I N F O

Article history:
Received 18 January 2015
Received in revised form
28 March 2015
Accepted 10 April 2015
Available online

Keywords:
Climate Compatible Development
Economic valuation
Mangrove
Ecosystem service
Coastal management

A B S T R A C T

Mangrove forests are under global pressure. Habitat destruction and degradation persist despite long-standing recognition of the important ecological functions of mangroves. Hence new approaches are needed to help stakeholders and policy-makers achieve sound management that is informed by the best science. Here we explore how the new policy concept of Climate Compatible Development (CCD) can be applied to achieve better outcomes. We use economic valuation approaches to combine socio-economic data, projections of forest cover based on quantitative risk mapping and storyline scenario building exercises to articulate the economic consequences of plausible alternative future scenarios for the mangrove forests of the South Kenya coast, as a case study of relevance to many other areas. Using data from 645 household surveys, 10 focus groups and 74 interviews conducted across four mangrove sites, and combining these with information on fish catches taken at three landing sites, a mangrove carbon trading project and published data allowed us to make a thorough (although still partial) economic valuation of the forests. This gave a current value of the South Coast mangroves of USD 6.5 million, or USD 1166 ha^{-1}, with 59% of this value on average derived from regulating services. Quantitative risk mapping, projecting recent trends over the next twenty years, suggests a 43% loss of forest cover over that time with 100% loss at the most vulnerable sites. Much of the forest lost between 1992 and 2012 has not been replaced by high value alternative land uses hence restoration of these areas is feasible and may not involve large opportunity costs. We invited thirty eight stakeholders to develop plausible storyline scenarios reflecting Business as Usual (BAU) and CCD – which emphasises sustainable forest conservation and management – in twenty years time, drawing on local and regional expert knowledge of relevant policy, social trends and cultures. Combining these scenarios with the quantitative projections and economic baseline allowed the modelling of likely value added and costs avoided under the CCD scenario. This suggests a net present value of more than US$20 million of adoption of CCD rather than BAU. This work adds to the economic evidence for mangrove conservation and helps to underline the importance of new real and emerging markets, such as for REDD + projects, in making this case for carbon-rich coastal habitats. It demonstrates a policy tool – CCD – that can be used to engage stakeholders and help to co-ordinate policy across different sectors towards mangrove conservation.

1. Introduction

Mangroves are the only woody plants to grow in the intertidal zone. They occur throughout tropical and subtropical latitudes where they may form extensive forests, particularly in sheltered bays and deltas. Their global extent, approximately 138,000 km^2 (Giri et al., 2011), is shrinking by around 0.7% per year, but this figure underestimates the problem since it applies only to complete removal of the forest and does not capture forest degradation. Causes of mangrove decline include shrimp aquaculture, conversion for tourism and coastal infrastructure, commercial extraction of timber and extensive but persistent extraction of wood for fuel

* Corresponding author.
E-mail address: m.huxham@napier.ac.uk (M. Huxham).

http://dx.doi.org/10.1016/j.jenvman.2015.04.018
0301-4797/

and building materials. Climate change, and in particular sea level rise, is likely to exacerbate these impacts over the next century (Gilman et al., 2008).

Many studies have documented the impressive array of ecosystem services provided by mangroves. These include provisioning (such as fish, timber and medicines), cultural (such as spiritual sites and tourist attractions) and regulating (such as coastal protection and carbon sequestration). The continued destruction of the forests, despite their well-documented ecological value, has become a *cause celebre* amongst conservationists and is used to illustrate irrational or short-term planning (in for example the Millennium Ecosystem Assessment: Watson et al., 2005). It is often argued that undervaluation, in particular, remains a persistent problem; that the benefits associated with mangrove ecosystem services (and conversely, the economic costs associated with their degradation and loss) have long tended to be omitted from the economic calculations that are used to inform coastal development decisions (Emerton, 2006). In consequence, markets and prices fail to adequately reflect ecosystem service values, and so they are rarely considered when resource management decisions are made. The effects of undervaluation are also manifested at the policy level: there is a long history of economic policies which aim to stimulate production and growth having also hastened the process of mangrove degradation and loss. Examples include the generous tax breaks, import duty exemptions, export credits and preferential loans offered to shrimp farming in many countries (Primavera, 1997; Bailly and Willmann, 2001). The net result is that it frequently remains more profitable for people to engage in economic activities that degrade mangroves – even if the costs and losses that arise for other groups, or to society in general, outweigh the immediate gains to the land or resource user who is causing the damage. At worst, in the absence of information about ecosystem values, substantial misallocation of resources has occurred and gone unrecognized, and immense economic costs and ecological damage have been incurred (James, 1991).

In response a growing literature exploring the economic value of mangrove ecosystem services has emerged over the last two decades or so (see for example, Barbier et al., 2011; Dixon, 1989; Conservation International, 2008; TEEB, 2012; Wattage, 2011; UNEP-WCMC, 2011). Such studies often contain impressive figures; for example Barbier et al. (2011) cite values for coastal protection in Thailand in excess of US\$10,000 ha^{-1} yr^{-1}, 1 ha of Mexican mangroves may contribute US\$37,000 yr^{-1} to the value of local fisheries (Aburto-Oropeza et al., 2008), and mangroves in Benut, Johor State in Malaysia have been estimated to generate non-use values of almost \$7500 ha^{-1} yr^{-1} – more than five times as much as the combined value of their provisioning and regulating services (Bann, 1999). The authors of such work hope that by expressing these values in monetary terms they will change how decisions are made about land use and conservation in favour of long-term sustainability of mangroves.

Critics of such 'market environmentalism' warn that it may imply a dangerously simplistic view of ecosystems (by, for example, separating out functions and services that in reality are synergistic), reinforce existing social inequalities, detract from the ethical or moral arguments for conserving wild nature and encourage the intrusion of market norms and psychology into inappropriate spheres of life (Kosoy and Corbera, 2010). Several authors also contend that there remains little evidence that providing monetary estimates of ecosystem values has actually resulted in improved conservation (King, 1998). Despite such concerns, we think valuation offers an important opportunity to improve the efficiency, equity and sustainability of land and resource management decisions. This is particularly true provided that uncertainties are explicitly acknowledged, care is taken to consider the underlying power structures that support different decisions and for goods and services that already have clearly understood market values for the poor but that may not have been fully assessed in ways accessible to policy makers.

However, the continued destruction of mangroves, despite the apparently compelling case made by scientific research and valuation studies for their conservation, points to other limitations to the idea that a simple lack of information drives damaging changes. One missing component may be active engagement with policy makers and other stakeholders during and after the research; without this academic studies may be ignored entirely, or seen as abstruse or irrelevant. Coastal scientists are aware of the pressing need for this engagement; a recent study identifying research priorities amongst scientists working in the coastal zone placed a better understanding of policy, legal and institutional arrangements and how these inter-relate with management as the top global priority (Rudd and Lawton, 2013). A related problem is one of context. Whilst it might be instructive to see estimates of total economic value for an ecosystem these need to be contextualised, for example by showing how much value could be lost under different scenarios, in order for them to have obvious traction. At the same time, while there is clearly a need to demonstrate and communicate the value of coastal and marine ecosystem services to decision-makers, if better and more informed choices are to be made between different land, resource and investment options (Agardy et al., 2005; Brown et al., 2008; UNEP-WCMC, 2011), valuation is not an end in itself. However high the value of mangrove ecosystems is demonstrated to be in theory, this has little meaning unless it actually translates into shifts in real-world policy and practice, and changes the economic opportunities, prices and markets that land and resource users face as they go about their day-to-day business (Emerton, 2006, 2013). Hence there needs to be explicit consideration of the policy landscape and a concern for plausible solutions; 'plausibility' here being informed by the stakeholders who could bring about change and by ways in which theoretical values might translate into actual conservation and management cash.

There are many examples in international policy of calls for integrated management of estuarine, coastal and marine habitats, with regards to their use, conservation, restoration and in climate change mitigation and adaptation (e.g. the Convention on Biological Diversity (CBD), the Ramsar Convention on Wetlands (Ramsar) and UNEP Global Programme of Action for the Protection of the Marine Environment from Landbased Activities (GPA-Marine)). The limited success of these policies when applied to mangroves illustrates the failure of policy makers to effect new economic opportunities that support local conservation, but developments in climate change policy may open new ways to link global concerns with local action. The United Nations International Strategy for Disaster Reduction (ISDR) explicitly links ecosystem conservation with a reduction in risk factors exacerbated by climate change, implying the need to invest national risk reduction funding into ecosystem management. The United Nations Framework Convention on Climate Change (UNFCCC 1992, Article 4 (d)) now supports opportunities for forest conservation, principally through the Reduced Emissions from Deforestation and forest Degradation + (REDD+) and Nationally Appropriate Mitigation Actions (NAMAs) of the Durban Platform. Coupled with the growing recognition of the importance of coastal ecosystems as globally significant sinks for carbon (so-called Blue Carbon) and the emerging global market for carbon offsets, these developments provide new ways of linking theoretical values of two ecosystem services (risk reduction and carbon storage) with income for local people (Grimsditch, 2011); they help make conservation scenarios plausible.

Here we illustrate how an economic valuation approach can be applied within the policy context of Climate Compatible Development (CCD). CCD is a recent concept that aims for 'triple wins' in planning; change that enhances adaptation to current and anticipated climate change impacts, whilst also mitigating the production of greenhouse gases and leading to increases in human welfare (Mitchell and Maxwell, 2010). It is part of a growing international focus on developing more integrated approaches to coping with climate change and avoiding 'maladaptation', in which short term planning creates greater future problems. Such an approach may seem simultaneously obvious and utopian; if it can be done then why is it not always adopted? In fact there are many instances of policy options that could achieve CCD but that are not routinely applied, for example agro-ecological approaches leading to reduced costs, enhanced yields, better resilience and increased carbon storage (Pretty et al., 2006). Hence there are parallels here with using economic valuation as a tool for conservation and development planning; simply identifying apparently irrational policy or damaging trends is not enough to effect change without the involvement of multiple stakeholders. Many of the most important services provided by mangroves, such as coastal protection and carbon sequestration, are of direct relevance to CCD implying that wholesale loss of mangrove forests will rarely reflect a climate compatible policy direction. Using economic valuation to illustrate the benefits of a CCD scenario that is developed with key stakeholders is a new way to help conceptualise and communicate the value of mangroves.

The present paper considers the current and future value of the mangrove forests of southern Kenya. Case studies at a range of sites have demonstrated the importance of these forests to local people, in particular through the provision of services such as fuelwood and building timber (e.g. Abuodha and Kairo, 2001; Rönnbäck et al., 2007). However recent trends in the country reflect the global picture of mangrove removal and degradation. Total mangrove coverage in Kenya was 45,590 ha in 2010, which represented a loss of 18% of cover over the previous 25 years (Kirui et al., 2013). Quantitative risk mapping shows that most loss is associated with high population density and accessibility of the forests, hence forest loss is most pronounced in the southern coastline where population is highest and infrastructure most developed (Rideout et al., 2013). Simple extrapolation of these trends suggests a bleak future for mangroves in southern Kenya with concomitant impacts on the local and regional ecosystem services that they supply.

Our aim in the current paper was to use Kenyan mangroves as a case study in developing CCD processes of use more generally in coastal management, and thus to contribute to global debate and practice on CCD and on valuing 'Blue Carbon' and other coastal ecosystem services. We aimed to do this whilst informing planning and policy making in Kenya itself, and did so through the following four steps:

1) Conducting a comprehensive and up-to-date socio-economic analysis of the current value of mangrove forest services in Kwale District, southern Kenya.
2) Developing quantitative projections predicting mangrove coverage in the area in twenty years' time based on current trends and drivers.
3) Articulating the economic consequences of mangrove ecosystem change, in particular the value-added and costs-avoided that are associated with sustainable land and resource management approaches.
4) Using this information to inform a discussion with key stakeholders over Climate Compatible Development options for the Kenyan mangrove sector and to compare Business as Usual (BAU) and CCD scenarios developed collaboratively and costed to illustrate the benefits of CCD.

2. Methods

2.1. Study area and site descriptions

Our study concerns the South Coast of Kenya, lying to the south of Mombasa and covering the coastline of Kwale county (Fig. 1). Mean annual rainfall along this coast varies from 1000 to 1600 mm, relative humidity is high all year round. Maximum tidal range is ~3.9 m. The whole Kenyan coastal zone supports a population of 3.3 million, around 8.6% of the national population, and at 2.9% p.a. population growth exceeds the national average of 2.2% (KNBS, 2010). Around 40% of the coastal population lives in urban areas while the remaining 60% are rural.

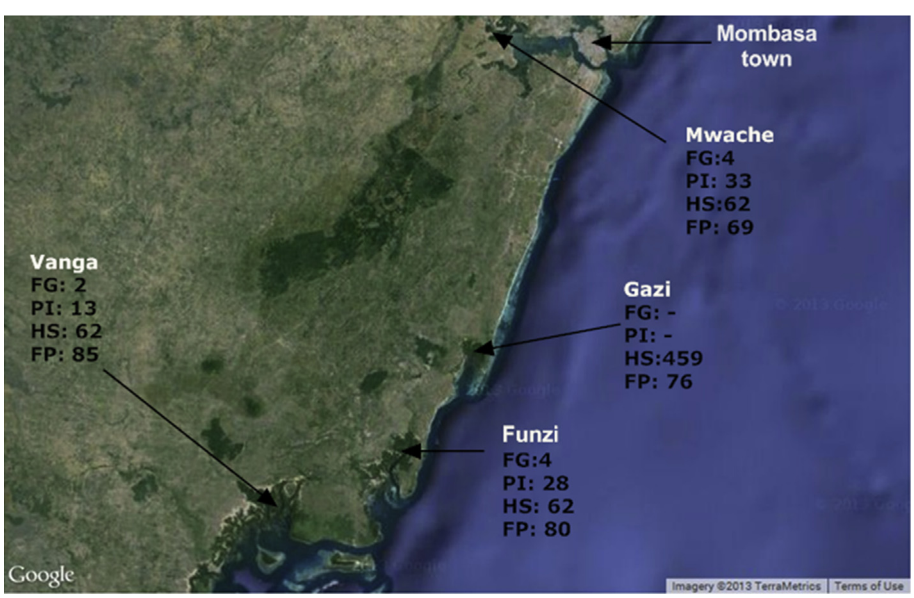

Fig. 1. Mangrove forest field sites in the South Coast of Kenya. Sample sizes taken at each area for this study are given as: FG focus groups; PI personal interviews; HS household surveys; FP forest plots.

All the nine mangrove species found in East Africa occur in Kenya, with *Rhizophora mucronata*, *Avicennia marina* and *Ceriops tagal* the dominant species at most sites. The South Coast includes four main mangrove areas, Mwache (04°3.01′S, 39.06°38.06′E), Gazi (4° 25′- 4° 27′ S, 39° 50′ E), Funzi (4°31′-4°35′S/39o23′-39°27′E) and Vanga (4°39′- 4°40′S/39°14′-39°17′E). These forests range in size from 592 to 2351 ha and all coexist with contiguous settlements, generally clustered into villages but including scattered homes and farmsteads (Table 1). The many extractive uses of these forests include exploitation for timber, wood-fuel and herbal medicines with some clearing for fish ponds, prawn farms, salt pans and port developments. Although no comprehensive forest survey, including all these sites, has been published there is abundant evidence to suggest that all these forests are heavily impacted and degraded by human use (e.g. Abuodha and Kairo, 2001; Dahdouh-Guebas et al., 2004; Mohamed et al., 2008), with resulting on-going declines in total forest area (Kirui et al., 2013).

2.2. Socio-economic and forest status analyses

The bulk of the mangrove valuation literature refers to Asia and the Americas; there remain relatively few applications in African countries (see for example McNally et al. 2011; De Wet, 2004). There are very few previous studies from Kenya, although two focus on one of our sites, Gazi (Kairo et al., 2009; UNEP, 2011). Here we use the previous work from Kenya and augment it with extensive new biological and socio-economic field data.

Forest structure data were compiled for 310 ten × ten m forest plots, covering a total of 3.1 ha (Table 1). In each plot the species, diameter at breast height (dbh) and height of trees were recorded along with indications of human impact (such as stumps, cut branches and pole quality). As one integrated measure of forest health, above ground dry weight for forest plots was calculated using an allometric equation derived from work at Gazi Bay (Pamoja, 2011):

$$LN\ Biomass = -2.29711((LNdbh) \times 2.54528)$$

Total areas of each forest were derived from recent satellite imagery (Kirui et al., 2013).

After negotiating access to villages through the appropriate local chairmen and elders socio-economic data were collected using three survey methods: household questionnaires, key informant interviews and focus groups. An interview schedule with 31 questions, covering income and employment, membership of local civic groups and use and collection of mangrove services, for family use or for sale, was piloted and then completed by local people selected by approaching households at random in each site. Semi-structured interviews were conducted with key informants, including village heads and opinion leaders, traders of mangrove goods, licensees for mangrove wood cutting, Kenya Forest Service officials and crab and prawn collectors. Interviews were informed by 7–12 questions which differed depending on the interviewee, with questions for collectors and traders focussing on rates and value of harvest and those for village heads and officials on numbers of traders/collectors and governance issues. All interviews were recorded by hand and transcribed.

Table 1
Forest size and populations at study sites.

Site	Mangroves (ha)	Number of households
Gazi Bay	592	498
Mwache/Mikindani	808	594
Bodo/Funzi	1815	500
Vanga	2351	2074

Focus groups were held separately for men and women, with people from community organisations, such as Beach Management Units and local conservation groups invited to attend. They took 30–50 min and were guided by a series of questions on who uses the mangroves and how, how access to the mangrove resources is governed, what local customs there are regarding mangrove use, any changes in mangrove status that have been noticed over the years and how well mangroves are managed. Notes were taken by hand during each focus group and transcribed shortly afterwards.

All ecosystem services provided by mangroves to local people that were identified by respondents to the surveys were included in the valuation exercise. Following the Millennium Ecosystem Assessment (Watson et al., 2005) we classified services into provisioning, cultural/aesthetic, regulating and supporting categories. Survey data provided quantitative results only for services in the first two of these categories. Relevant regulating services and appropriate economic values for these were identified from the literature, including papers on local forests by members of the current team (e.g. Huxham et al., 2004; Huxham et al., 2007; Kairo et al., 2008) and from the technical specifications developed for Mikoko Pamoja, a mangrove carbon offsetting scheme based at Gazi (Pamoja, 2011).

Village heads were asked to provide demographic data for each area, and these were augmented by literature and where necessary by direct surveys of the number of households; although population data are available from KNBS (2010) these can date rapidly given high rates of population growth and local migration.

2.3. Valuation methods

Here we provide an overview of the methods used; more detailed information is available from Emerton (2014) and Huxham (2013). Figures are expressed as net values, the costs of harvesting, producing or using ecosystem services are deducted, and at constant, 2014 prices.

2.3.1. Direct provisioning services

Monetary values for all of these services were estimated at each site by multiplying the volume per year produced or extracted by the relevant market price, taken as the sale (for traders) or purchase (for consumers) price at the site. Data were obtained from household surveys and interviews with licenced cutters. Wherever possible, harvest and production costs were subtracted so as to yield net values (for example fees payable to the Kenyan government for collection of timber and crabs and costs of fisheries gear maintenance and replacement were subtracted). Time was not taken as a cost apart from where salaries were paid (e.g. by a licenced pole trader employing cutters). Some products are extracted without licence or illegally. These values are more difficult to estimate because they do not appear in official records, and users are often reticent to disclose details of their activities. For this reason, a combination of methods was used to calculate unlicensed or illegal values, including extrapolating data obtained from legal users to a larger population. For example timber for building poles can be extracted from all the study sites under licence from the Kenyan Government and the values of this legally harvested wood were calculated following interviewees with licensees and professional cutters. However this grossly underestimates the wood value since it does not account for illegal extraction. The present surveys recorded widespread acknowledgement of illegal extraction, as has previous relevant work (Rönnbäck et al., 2007) and personal observation. Hence households were asked about collection and use of mangrove timber, and where mangrove timber was not recorded as either being bought from a licenced trader or having been collected by a household member who was themselves an employee of a licenced cutter its value was recorded as additional to that for legal sales.

2.3.2. Indirect provisioning services — fisheries

Mangroves provide nursery habitat for a wide range of fish species, many of which contribute to artisanal and commercial fisheries once they reach adult size (Mumby et al., 2004). There is therefore no contention over the importance of mangroves to fisheries in general; however there are many estimates in the literature of the extent of the mangrove contribution to fisheries, ranging from 5 to 100%. This variation reflects in part genuine differences between species and sites, but also reflects the methodological challenges in linking nursery sites with adult fish stocks and the paucity of relevant data for most field sites. Where no local evidence exists, projects often take an average value or choose one that seems appropriate for the conditions at the focal site. Fortunately, and unusually, there are a number of studies in Gazi and of sites nearby of direct relevance to this question. Kimani et al. (1996) recorded 128 teleost species in Gazi Bay during 12 months of sampling; many of these species, caught directly adjacent to the mangroves, are likely to be dependent upon them, although since sampling in this study did not occur inside the mangroves this remains an inference. Three subsequent studies (Crona and Rönnbäck, 2007; Huxham et al., 2008, 2004) demonstrated that a range of juvenile fish were indeed entering into the mangrove habitat itself, and in some cases travelling long distances from the seaward fringe (Huxham et al., 2008); there was a strong overlap with the species and families found in other mangrove sites in the region (e.g. Lugendo et al., 2007). Finally, Huxham et al. (2007) used stable isotope techniques to demonstrate that adult fish, caught offshore, had spent some of their juvenile years in mangrove habitats. Hence a strong body of evidence exists for mangrove dependence of a range of teleost species at Gazi. Here, we classified a species or group as 'mangrove dependent' if it has been recorded as a juvenile in mangrove habitat at our sites. Using this criterion an average 39% of the value of fish caught offshore came from mangrove dependent species.

Commercial catches of coastal species along the south coast are landed and traded at five major administrative landing sites: Diani, Msambweni, Shimoni, Majoreni and Vanga. Records of weights and identities of all species landed are recorded by the Kenyan Department of Fisheries. We collected landing data from all these sites for 2012. Based on their proximity to mangrove forests, we allocated catches from the Vanga and Majoreni landing sites to the Vanga forest, Shimoni landing site to Funzi and Msambweni landing site to Gazi. There is no administrative landing site (and hence no official landing records) in the Mwache area; estimates of fisheries values from that forest are taken from direct interviews with local fishers. Net values were calculated after removing annual costs; these were calculated using information obtained in interviews with fishers at the landing sites on costs of salaries, licences, fuel, gear and boat maintenance and replacement. Costs averaged 30% of income.

2.3.3. Regulating services — carbon sequestration

Gazi Bay hosts the Mikoko Pamoja project, a recently validated community-based forest conservation programme funded by carbon credits (see Pamoja (2011) and www.eafpes.org for more information). This makes it the best place in Africa to estimate the economic value of carbon sequestration. Carbon benefits can be categorised in two ways: a) the additional carbon sequestered annually by the existing forest and any new forest area, and b) the carbon emissions avoided by maintaining forest cover and quality. The second benefit is much larger than the first, but also more uncertain. Avoided emissions are potentially enormous if all the carbon present in the ecosystem is considered, but even total removal of the trees does not always imply loss of all the above-ground carbon (since wood products may retain the carbon) or the below-ground carbon (since some soil carbon is likely to remain un-oxidised). Whilst mature mangrove forests may continue to sequester carbon for hundreds to thousands of years, avoided losses of stored carbon should be priced only until depletion. Here we combine the two values since our time horizon is only 20 years.

Mature mangrove forests continue to sequester carbon in three ways: 1) Above-ground, by new growth of branches and trunks. 2) Below-ground, through new growth of roots. 3) In and on the sediment, through root exudates (carbon that is 'leaked' from roots), autochthonous production (that is, plant material such as leaves that grow in the forest and become buried) and the trapping of sediments and organic material from outside the forest. Mikoko Pamoja measurements used a conservative estimate of 4.5 t C ha^{-1} yr^{-1} for 1. A (conservative) estimate of 0.3 for root:shoot ratios (based on work at the site; Tamooh et al. (2008)) implies an additional 1.35 t C ha^{-1} yr^{-1} for root growth. Assuming a conservative 1 t C ha^{-1} yr^{-1} for sediment trapping (less than the global average: Alongi, 2014) gives total average sequestration is 6.85 t C ha^{-1} yr^{-1}.

Recent experimental work using small-scale forest cutting has shown rates of carbon losses of 4.85 t C ha^{-1} yr^{-1} from sediment following forest removal (Lang'at et al., 2014). We use this value as an estimate of annual avoided emissions. This conservatively assumes that all above-ground carbon, contained in harvested wood, is not released into the atmosphere and that these small experimental cuts are representative of larger areas. Based on realistic current market values for the voluntary carbon market we took the price of 1 tonne of CO_2 as US$10, and used these values derived from Gazi for all the forest sites. Estimated costs were subtracted from annual income to give net values; they were calculated based on the start-up and running costs incurred for Mikoko Pamoja, excluding the value of volunteer time.

2.3.4. Regulating services — protection from coastal erosion and storm surges

In most mangrove valuation studies coastal protection generates the largest single contribution to total economic value, because of mangroves' demonstrated ability to protect shorelines against gradual erosion and storm events and because of the high cost of 'hard engineering' alternatives (e.g Barbier et al., 2008). This ecosystem service is likely to become more important with climate change, as coastal areas come under greater pressure from human populations and rising sea levels; mangroves have the ability to adapt to at least modest levels of sea level rise through sediment accretion and surface elevation.

Many valuation studies make large (and in some cases unrealistic) estimates of the value of mangrove coastal protection services, applying a high per hectare figure even where there is little settlement or infrastructure in place, and only a minor likelihood of "best practice" coastal protection or remediation measures being actually implemented. The figures used here equate to a combined average of just under US$ 500 ha^{-1} yr^{-1}. It should be noted that this (deliberately) represents a fairly conservative estimate in comparison to those yielded from other studies carried out in the wider Indian Ocean region: IUCN (2006) and Ranasinghe and Kallesoe (2006) for example find average values of between US$ 3300–9500 on the east coast of Sri Lanka, Das (2007) stipulates a figure of more than US$8500 in India, and Sathirathai (1998) and Sathirathai and Barbier (2001) suggests values of between US$3000–4000 in southern Thailand.

2.3.4.1. Coastal erosion.
The beach to the south of Gazi village suffered removal of mangroves for commercial purposes some forty years ago, and the denuded areas have not recovered (Kirui et al., 2008). This degradation has resulted in obvious coastal erosion

and the longshore movement of sand (Dahdouh-Guebas et al., 2004). Hence this beach allows a locally relevant measure of coastal erosion following mangrove removal, although it does not provide a perfect opportunistic 'experiment' since a narrow ~50 m strip of mangrove remains along the beach which undoubtedly continues to provide protection. Because of this, estimates of the protective value of mangroves here are likely to be conservative.

Average rates of shoreline retreat along the degraded beach over the past 11 years were measured using satellite imagery, to provide an estimate of rates of shoreline loss after mangrove removal. Mitigative and avertive expenditure are used, looking at the cost of establishing and maintaining coastal defence structures such as groynes, breakwaters, revetments and sand replenishment that would be required to restore eroded coastal areas and continue to protect them in the future to an equivalent level should well-functioning mangroves have been in place. As no up-to-date figures were available for the Kenyan or East African Coast, cost data were transferred from a recent study carried out in a site in Sri Lanka which displays similar biophysical, ecological and socio-economic conditions to the Southern Kenyan coast (in De Mel and Weerathunge, 2011; Emerton, 2013). The transferred values were adjusted using appropriate CPI deflators and Gross Domestic Product Purchasing Power Parity (GDP PPP) conversion rates to account for real price differences over time and between Sri Lanka and Kenya. The resulting annualised figure of US$ 20.81 m^{-1} yr^{-1} was applied to the length of mangrove protected coastline, measured using satellite imagery, at each site.

2.3.4.2. Storm surge protection. Some areas on the southern coastline have invested in seawalls to protect property against the impacts of high seas and storm surges; these include Bamburi and Vanga (UNEP, 2011). The Vanga sea wall cost US$952 per metre to build (UNEP, 2011). Adding 1% maintenance costs per year gives a locally relevant replacement cost for storm protection services from mangroves. Tychsen et al. (2008) modelled the impacts of a tsunami on the south coast and found that mangroves afforded protection 300 m inland. Using satellite imagery we identified all areas on the coast that included housing or buildings within 300 m of the coastline and that were situated behind mangrove stands of at least 100 m thickness. We used replacement costs for building seawalls in these areas as a measure of the storm surge protection value of mangroves (hence we included only protection value for high value land with visible buildings).

2.3.5. Cultural services

Estimates for the market values of tourism, education and research and ritual consultation were made using a mix of published sources (UNEP, 2011) and interview data. Data were most comprehensive and reliable from Gazi, with the relevant estimates for tourism and education/research taken from this site and applied to the other sites based on interview data and estimates from those areas on the numbers of visitors. Traditional religious practices in the area involve the consultation of shaman in mangrove shrines which includes payments and offerings needed for intercession with spirits. Estimates of average market values for these were obtained through interviews and applied to sites on the basis of the estimated number of appropriate shrines.

2.4. Developing storyline scenarios

We adapted the methods described by Rounsevell and Metzger (2010) to develop qualitative storyline scenarios for the southern coast mangrove forests by 2033. The purpose of this exercise was to draw on the expertise of relevant regional stakeholders, from a wide range of backgrounds, to construct plausible scenarios. Engagement with these partners at this stage was also part of a longer term effort to ensure our science was rooted in relevant local concerns and was understood by local users. Regional stakeholders were invited to participate in a facilitated process in November 2013. Thirty eight participants drawn from relevant government departments (including the Kenya Forest Service, National Environmental Management Agency, Kenya Marine and Fisheries Research Institute and Kenya Wildlife Service), NGOs, community bodies and academia worked in four separate and independent groups through a five step process. This first identified the *state descriptors* (such as forest area and quality) and *drivers* (the causes of change in these descriptors). The second ranked these drivers in terms of perceived importance and degree of uncertainty around them. Third and fourth steps involved producing descriptive narratives of business-as-usual (BAU) and Climate Compatible Development (CCD) scenarios, based on the key drivers and what were agreed as plausible interpretations by the group participants. Groups then described the main winners and losers under each of the two scenarios. At the end of this process the results from the four independent groups were collated and two synthesis scenarios developed and circulated for approval between all participants.

2.5. Projections of forest change

In order to allow quantitative valuations of the scenarios we modelled forest cover in 20 years time (i.e. for 2033) informed by the qualitative scenarios. The BAU modelling made two key assumptions. First that national rates of mangrove forest loss recorded over the past 20 years by Kirui et al. (2013) would continue (at 0.7% per annum). Second, that the risk factors identified as important predictors of forest loss in the last two decades by Rideout et al. (2013), including population density and proximity of roads, would continue to influence the relative risks to forests. Rideout et al. (2013) validated a qualitative risk model that classified remaining forest areas into one of five risk categories. Applying this model we selectively removed forests in the highest risk category (5) first, followed by areas at lower risk, until the anticipated total national forest loss was achieved. The rank order in which forest pixels in lower risk categories were lost depended on their distance from areas of forest identified in the next higher risk category; hence those pixels in category 4 that were contiguous with category 5 areas were the first to be lost. Under this procedure total forest loss was assumed *a priori* but the distribution of this loss was determined by the risk factors, resulting in very different rates of loss between forests areas.

To inform the forest scenarios under CCD we used the satellite images described in Kirui et al. (2013) to identify all those areas at each of our sites from which mangroves have been lost in the last 20 years. We then used Google Earth to estimate the proportion of these areas that were occupied by productive alternative land uses, such as new buildings, agricultural land or coconut plantations, and therefore the proportion that were apparently left as bare, unproductive land. Because most incremental forest loss in the area has been driven by small scale cutting for timber and firewood such areas are common; whilst they often do not regenerate naturally, new mangrove forest can be established through active restoration projects (Kirui et al., 2008). Hence we classified such areas as available for restoration under CCD.

3. Results

3.1. Current forest status and quality

Analysis of the new forest structural data taken across the south coast supports previous work from individual sites that the forests

Table 2
Measures of forest quality at the four study sites along with reference comparisons. Mean and median plot dry weights are in tonnes ha^{-1}. 'Form 1' are the highest quality, straight poles (no data were available for Gazi). Reference data for plot biomass are from Lamu, in the north of Kenya, and for poles are from a protected plantation near the Gazi forest (Kairo et al., 2009).

Site	Mean (SD) plot biomass	Median plot biomass	% poles form 1
Gazi	81 (57)	71	–
Mwache	128 (102)	112	8
Funzi	121 (102)	87	0.4
Vanga	204 (230)	106	12
Reference	165 (66)	159	86

Table 4
Involvement of women in collection of provisioning services (% of harvesters who are women) and uses of each service (% of harvests used by collecting household, sold within villages or sold to traders conducting business outside). Data are from the totals for all sites.

Provisioning service	Own use	Village trade	Traders beyond village	Women
Timber	86	8	6	27
Fuel Wood	79	21	–	96
Wild Finfish	36	24	40	14
Wild Crustaceans	37	3	59	35
Honey	42	56	2	48

are degraded and heavily impacted by human activity. One measure of forest health is aboveground biomass. Median values were lowest at Gazi and highest at Mwache (Table 2). Most sites showed a marked right skew in biomass frequency distributions, with a few high value and many low value plots; this is reflected in generally large standard deviations and differences between mean and median values (Table 2). Structural data from forests near Lamu, in the north of the country, were analysed as a comparison. Forests in this area have suffered the lowest rates of loss over the past twenty years, have the lowest risk ratings and generally show the smallest human impacts (Cohen et al., 2013; Rideout et al., 2013); hence they are the closest available approximation to pristine 'control' sites in Kenya. Median biomass at Gazi is less than 50% that at Lamu. Another index of forest quality is provided by the proportion of tree

Table 3
Summary of value estimates in 2014 USD yr^{-1}.

		Gazi	Vanga	Funzi	Mwache	All sites	Mean (USD/ha)
Provisioning Services	Timber, fuelwood & honey	49,801	289,378	761,179	47,757	1,148,115	206
	Capture fisheries (finfish)	123,378	253,826	186,956	44,796	608,956	109
	Capture fisheries (crustaceans)	55,466	310,541	82,525	267,664	716,196	129
Regulating Services	Protection against coastal erosion	195,161	827,770	496,234	677,335	2,196,500	395
	Protection against extreme weather events	40,045	15,725	63,175	73,561	192,506	35
	Carbon sequestration	100,115	656,126	409,897	231,159	1,397,297	251
Cultural Services	Tourism, education & research	124,512	49,183	37,970	16,903	228,568	41
Total		688,478	2,402,549	2,037,936	1,359,177	6,488,139	1166
Mean value (USD/ha)		1163	1022	1123	1682	1166	

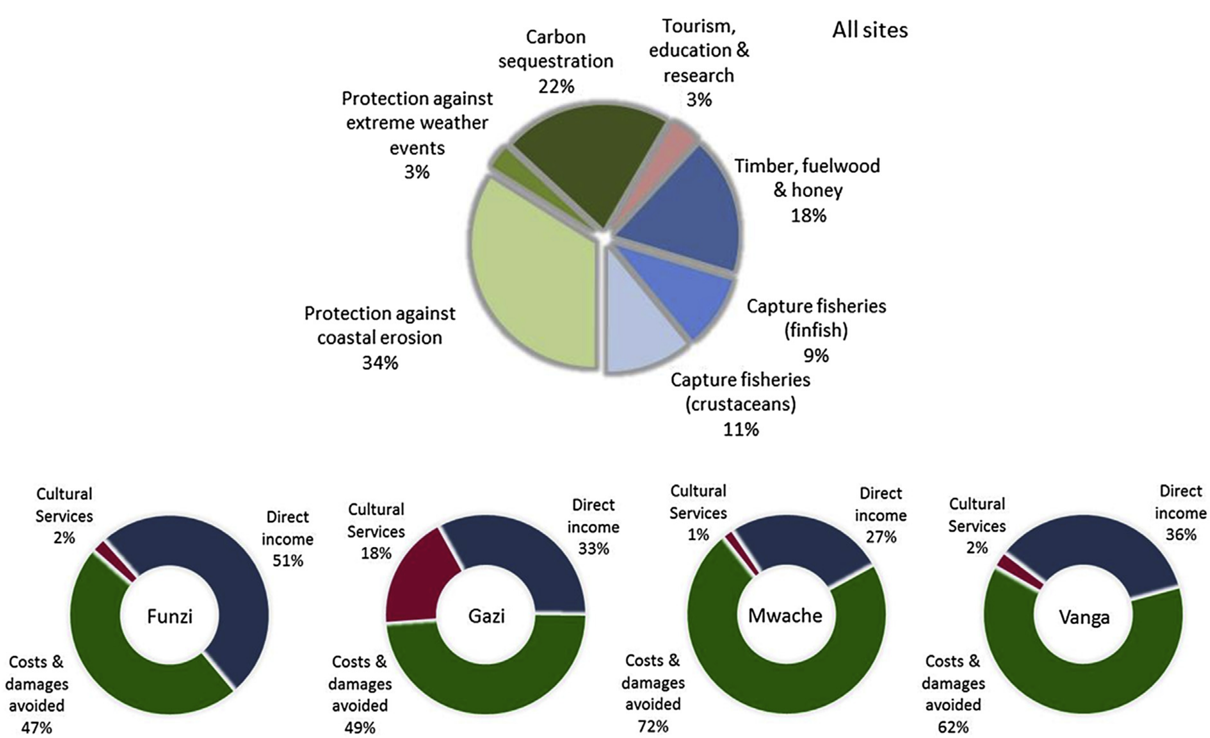

Fig. 2. Summary of value estimates in 2014 USD yr^{-1}.

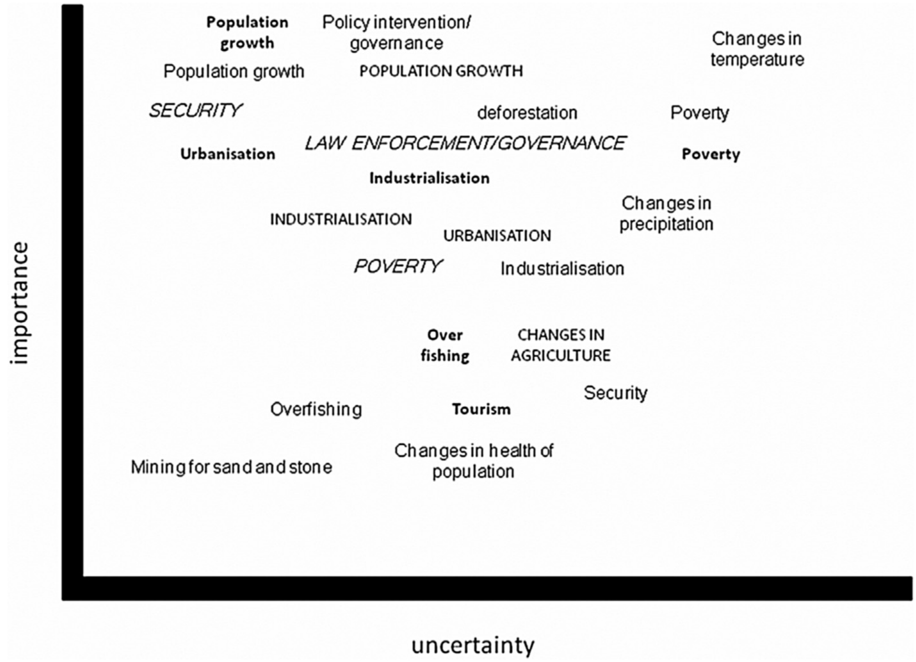

Fig. 3. Ranked drivers of change in the mangrove sector in South Coast Kenya, over a 20 year projected period. The figure summarises the discussions of four independent stakeholder groups, of eight people each. Four different fonts are used for the conclusions of each group; groups were not required to identify the same number of drivers but were asked to rank those that they did identify in terms of importance and uncertainty.

trunks that are of the highest quality form; straight and suitable for building timber. These are the poles which are most prized by cutters since they attract the highest market prices. Only 0.4% of surveyed trees at Funzi were of this form, compared with 86% of trees in an 18 year old protected plantation near to the natural Gazi forest.

3.2. The economic baseline

We randomly sampled around 18% of all households in our study areas (Fig. 1; Table 1) with saturation of topics in the surveys and focus groups suggesting we had achieved a representative sample. There were five provisioning services used by local people; in addition, three regulating and three cultural/aesthetic services were identified and valued, giving a total value of just under US$ 6.5 million or some US$1166 ha^{-1} yr^{-1} (Table 3). This list is not exhaustive. It certainly excludes some important regulating (such as the filtration of sediment and nutrients from water, which helps protect coral reefs) and cultural (such as existence value of biodiversity) services that were impossible to value here. It may also exclude other services of which we are simply ignorant. It is to be hoped that as better data become available, these estimates can be expanded and improved.

As has already been noted for studies carried out in other parts of the world, values are dominated by the damages and costs avoided associated with regulating services (Fig. 2). These account for around a half or more of total value in all four of our sites. Direct income (from wood and non-wood products and fisheries) comprises just under a third, overall, and is particularly important in Funzi (mainly due to the relatively high value of forest products harvested there).

Simply assessing the size of different types of values, without considering their distribution between users, may do nothing to illustrate the importance of services to marginal and poor groups. As one way to illustrate this, we investigated those services of most relevance to women and considered who was collecting provisioning services and where these services were used, as an indication of the relative values of services for local people. For example whilst fuel wood represented a relatively small proportion of total value at all the sites it is a very important service for local women; 96% of fuel wood collectors were women and 79% of the total harvest was used directly by those collecting, rather than for sale. This contrasts with, for example, crustaceans that are collected mostly by men and used to generate income through sale to traders (Table 4).

3.3. CCD and BAU storyline scenarios

The key drivers of change identified by the four expert stakeholder groups are shown in Fig. 3. Although each group of eight worked independently there was considerable agreement on the most important drivers: population growth, poverty, governance and urbanisation were all ranked highly.

Taking these rankings storyline scenarios under BAU and CCD were collectively developed and agreed, in an iterative process over three days, between all groups (Box 1 and 2). These scenarios were used to inform the economic modelling, with data on assumed land use change coming from the risk modelling.

Box 1
BAU scenario for south coast mangroves.

The social, ecological and economic landscape:

Rapid population growth will exceed the national average because of coastal cultural values and immigration to the region, some of which is fuelled by climate refugees from elsewhere in Kenya (as was seen in 2008 with pastoralists from the north). This will result in a near doubling in the numbers of people. Rapid urban spread, especially in Mombasa and Ukunda, and rapid industrialisation, including developments in mining and oil and gas industries, will absorb some of this new population. Hence impacts on natural resources may not be proportionate with increases in population size, as economic growth and electrification reduce the proportion of the population directly dependent on natural resources such as firewood. However the large increase in absolute numbers of people implies a growth in pressure on such resources.

Economic growth, fuelled by new industries and better education, will lift many out of poverty and generate a burgeoning middle class. However inequality is likely to increase and absolute numbers in poverty will remain the same. This growing inequality is one reason for continued insecurity; whilst better technology and new investment by government in law enforcement brings benefits, there are also domestic tensions between rich and poor and international insecurities and terrorist threats mediated by wider access to weapons. Whilst tourism continues to expand both in tourist numbers and in revenue generated it remains mostly foreign owned and with an emphasis on the long-haul luxury market with a continuing disconnect between a large tourist presence and local benefit.

Increasing use of digital media and levels of education bring local pressures for democratic accountability. Combined with the new devolved constitution this brings moderate improvement in local representation and governance. However policy implementation and law enforcement remain weak and co-ordination between sectors remains poor. There is little attempt to balance the needs of environmental protection against economic growth, as powerful new industries push environmental protection towards the bottom of local priorities.

Unabated climate change leads to increasingly erratic water supplies, with water availability for natural ecosystems further stressed by large new industrial uses, including titanium mining and sugar cane production, as well as the construction of the new Mwache dam to provide water for an expanding Mombasa.

The implications for mangroves

There is continued loss of forest area and reduction in forest quality. Mangroves near to Mombasa, which are already stressed, suffer additional problems following the Mwache dam and are mostly destroyed. Gazi mangroves are relatively protected because of high profile and existing small-scale initiatives such as Mikoko Pamoja, which are locally successful but which have a limited regional impact, despite the national Mangrove Management Plan, launched in 2014. This established a science-based approach to management but poor enforcement and policy integration limited its success. Degradation of the mangroves leads to declines in in-shore fish and crustacean species with mangrove dependence. Loss of mangrove resources, including timber and firewood, especially near large urban areas, increases pressure on terrestrial forests.

Box 2
A CCD scenario for south coast mangroves.

In 2033 government agencies dealing with environmental protection and development, including Kenya Forest Service (KFS), Kenya Wildlife Service (KWS) and the National Environment Management Agency (NEMA), have achieved harmonious and well integrated working practices to help achieve policy implementation and enforcement; this process began with the merger of KFS and KWS in 2014 following the implementation of the new constitution. This led to rigorous application of the existing progressive laws. A key change is the greatly increased empowerment and involvement of local people and communities, reflecting the enactment of commitments in the National Oceans and Fisheries Policy (2008), the Integrated Coastal Zone Management Policy (2007, draft), the Forest Act (2005), the draft Environment Policy (2012) and elsewhere – the pervasive commitment across policy and legislation in 2013 to public participation, equity and local control. This encouraged the growth of new Community Based Organisations (CBOs), including Community Forest Associations and Beach Management Units, as well as the strengthening of current ones, with appropriate and functioning community-based groups now active in all relevant locations – hence all mangrove forests and inshore fisheries have local involvement in management. The normalisation of community control of resources, especially in forestry and fisheries, has helped address poverty and promote greater equality. The growth of community organisation also led to improved transparency and governance at a local level, with technology assisting; CBOs have their own websites and use social media to communicate with members and their communities. Educational achievements improved, driven by better access to schools, improving wealth but also the investment by local groups in educational provision, and this has helped spread a message of environmental conservation. Security is improved in smaller communities because of this whilst at a county level the government has invested in policing so that security is no worse than it was in 2013, despite increased population. Whilst population growth has been rapid (at 2.6% per year there are 67% more people than in 2013) the growth is slowing and has been tempered by increasing education and empowerment of women.

Whilst the effects of climate change have become increasingly apparent their impacts have been ameliorated through new protection and restoration of key habitats, including mangroves and coral reefs. Water resources have been carefully husbanded to help with increasingly erratic supply. The new water-intensive industries that developed over the past 20 years, including mining and industrial agriculture, were required to implement strict environmental management plans that included consideration of water supply and quality. The Mwache dam became a case study in good management practice – despite initial concerns about its effects the massive afforestation of its catchment positively altered the local microclimate and reduced sedimentation, whilst flows were monitored to ensure sufficient freshwater for estuary and mangrove health down-stream.

Demand for natural products including wood and fish has increased due to urbanisation and population growth. However it was not proportionate to the increase in people since most of these new people are living in towns with increased access to electricity and regional and international markets. The expansion of industrial mining and other industries led to increased wealth which also reduced direct reliance on local ecosystem services including firewood and poles. Hence the 20% increase in demand for wood products is met through new woodlots and agroforestry. Tourist numbers increased by 50% over the past 20 years, following a long trajectory and facilitated by a stable security situation. But a key difference in the new tourist economy is the importance of the domestic market, with Kenyan tourists now making up 50% of the new visitors. The growing Kenyan economy, large Nairobi middle class and fast new rail link between Nairobi and Mombasa played important roles in stimulating this change. The overseas tourists are much more aware of the environmental impacts of their travel and all airlines are now required to consider offsetting or other mitigation efforts. Capitalising on this, the careful nurturing of links between airlines, tourist operators and local PES schemes led to a huge increase in resources for the preservation and enhancement of local sinks, including 'blue carbon' sinks.

Early encouragement for payments for ecosystem services in the draft Environment Policy (2012) combined with early success of pilot PES projects on the coast such as Mikoko Pamoja has produced a flourishing policy environment for PES. The quadrupling of carbon prices, from 6 to 24 USD per tonne CO_2, has transformed the viability of conservation schemes based on 'blue carbon' sinks, especially mangroves but including emerging schemes addressing seagrass. In combination with other sources of income from eco-tourism and new non-forest products this has increased the total economic value of natural ecosystems on the coast, and the commitments under the draft Environmental Policy (2012) to incorporate the value of natural capital in government accounting have been met, leading to increased policy awareness of ecosystem values.

A new generation of Kenyan tourist operators and managers are trained and management and ownership of tourist business shift towards Kenyans. Eco-tourism triples in importance with close links between tourist visits and community benefits, further financing good stewardship of local ecosystems.

The implications for mangroves

The historic deforestation of mangrove woods was halted in 2015 with implementation of the new national Mangrove Management plan, and the total area has increased by restoration of those areas that have not been permanently converted to alternative land uses. This followed community restoration of degraded areas which was led by the Community Forest Associations which are now well established and which benefit from the new political support for local control of resources. The quality and therefore value of the harvested mangrove wood has increased through proper husbandry, and following a period of reduced revenue from the sale of mangrove wood (which was necessary for recovery) revenue from productive mangrove services is higher than it was in 2013. All new destructive industries, especially mines, are required in their Environmental Impact Analyses to offset their biodiversity impacts including their impacts on mangroves.

The collectively produced CCD scenario storyline is given in Box 2.

3.4. Projections used for economic modelling

Modelled losses of forest cover under BAU, projecting current trends, came to 43% of the south coast forest over the next 20 years. These losses were highly variable between sites, reflecting the different proportions of each forest found in the high risk categories: percentage losses were 100, 69, 64 and 3 for the Mwache, Funzi, Gazi and Vanga sites respectively. Under the CCD scenario, in which available degraded areas are reforested, forest coverage expands by 8, 7, 9 and 13% respectively of the current forest areas of Funzi, Gazi, Mwache and Vanga.

Population trends follow the projections provided in the 2090 Census (KNBS, 2010): an annual growth rate for Coast Province of 3.05%.

Changes in the value of provisioning services were estimated based in the projected changes in user numbers, the structure and dynamics of their use, and the proportion of products that are sourced from mangroves. Regulating and cultural services relate per hectare unit values to changes in the mangrove area at each site.

Although there are grounds to suppose that the real price of mangrove services may change over the next twenty years, insufficient information exists to predict with any accuracy what these trends will be. On the one hand, continuing ecosystem degradation may result in services becoming scarcer, and their real price increasing (this relates primarily to the BAU scenario). The rising demands of a growing population, coupled with a limited resource base, may have a similar effect (under both scenarios). However, decreasing reliance on mangrove provisioning services due to the improved availability of cheap alternatives may have the opposite effect (i.e. lead to a decrease in real prices in the future). As it is impossible to predict with any certainty how these factors will play out on the southern Kenyan coast, all real prices are assumed to remain stable in both scenarios.

Average harvest or consumption rates per user were also assumed to remain stable, as there are no convincing grounds to suppose otherwise. The percentage of the population which utilises wood products was however assumed to decline, under both BAU and CCD scenarios. This reflects a continuation of the changes that are currently ongoing in the lifestyles, aspirations and demands of both coastal dwellers and the Kenyan population more generally (e.g. a shift away from reliance on wood fuel, moves towards brick-based construction, etc.). It should however be noted that these effects are counterbalanced somewhat by the increase in population size (in other words, although the percentage of households sourcing products from mangroves is assumed to decline, the absolute number of households will increase). No change was assumed in the percentage of households consuming or trading fish (although the percentage contribution of mangroves is assumed to decline in BAU and increase in CCD, as explained in the following paragraph).

Two indices were applied, so as to ensure that changes in mangrove cover and quality were reflected in value estimates. A "product availability index" was applied to provisioning services; this accounts for the change in product supply or yield that will occur as mangrove area and quality decline (under BAU) or increase

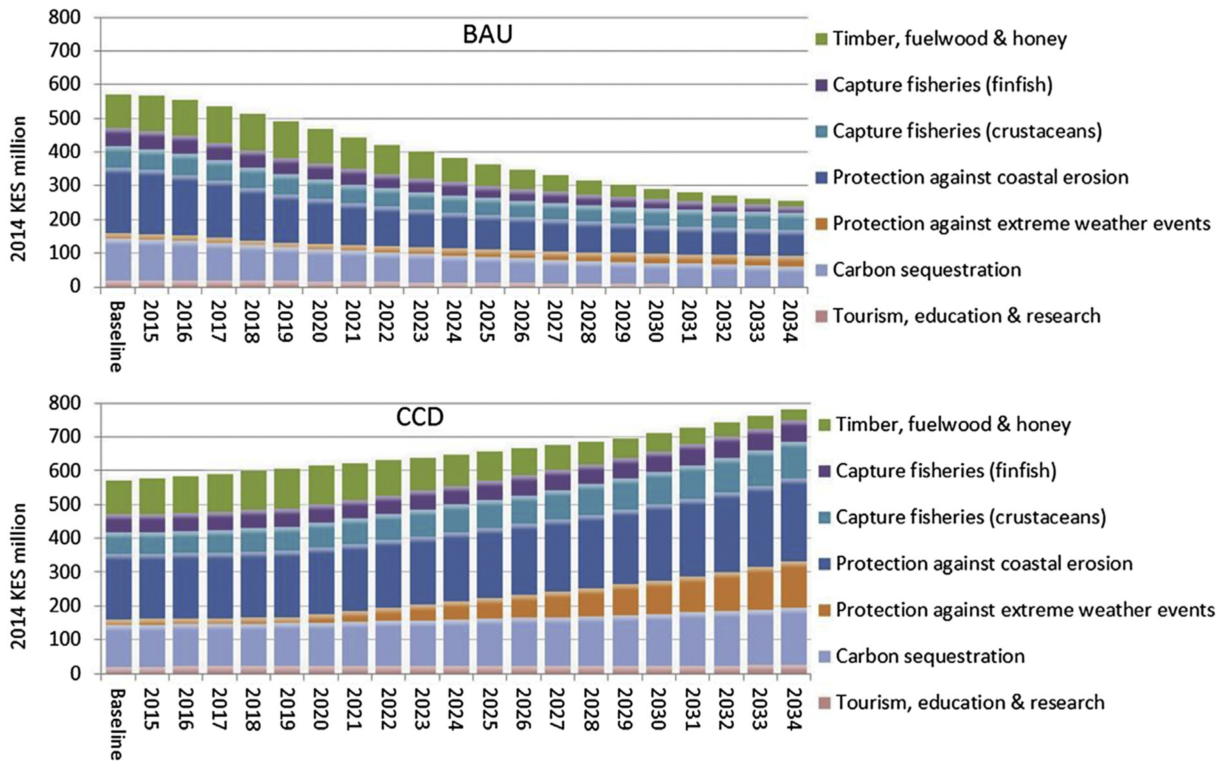

Fig. 4. Mangrove ecosystem service values under BAU and CCD in 2014 KES million.

(under CCD). This was used to estimate changes in the percentage of a product which is obtained from mangroves (as compared to other sources of, say, firewood or building poles). For regulating and cultural services, a "quality of ecosystem service index" was applied to the value of the ecosystem service per unit area. Indices were calculated based on the year-on-year change in area, multiplied by a factor that is determined by the baseline forest status and quality at each site and its assumed decline (under BAU) or improvement (under CCD) over time.

3.5. *The economic impacts of forest ecosystem change under BAU and CCD*

Running the economic scenario model shows that BAU will result in a progressive decline in mangrove values over the next 20 years, while CCD will see a sustained increase in ecosystem values over time (Fig. 4). The rate of increase in value under CCD will initially be slow, as measures to achieve CCD are set in place; it will then rise as these measures take effect, before slowing again as mangroves are restored to a healthy functioning state and area. The net present value (NPV) of mangrove services to 2034 under the BAU scenario is US$42.85 million; under CCD it is US$ 61.01 million.

Over the 20 year period modelled, BAU will incur total losses of around US$41 million as compared to a continuation of the baseline, while CCD will lead to incremental benefits worth more than US$ 20 million in total (Table 5). These figures equate to a net present cost of US$ 12.38 million under BAU over and above the baseline, and a net present value of US$ 5.77 million under CCD.

More than US$ 61 million additional value (with a NPV of US$ 18.16 million) will be generated over the next 20 years from CCD as compared to BAU (Fig. 5). This is, in effect, the return to investing in climate-compatible development measures (or, conversely, the cost of policy inaction as regards sustainable coastal ecosystem management). By the year 2034, mangrove ecosystem services will be generating values worth almost US$ 10 million a year under the CCD scenario (almost 40% more than what they are worth today), as compared to under US$ 3 million under BAU (less than half of today's value).

Table 5
Value-added by BAU and CCD in 2014 USD.

	BAU value-added over baseline		CCD value-added over baseline		CCD value-added over BAU	
	Total	NPV@10%	Total	NPV@10%	Total	NPV@10%
Timber, fuelwood & honey	−7.94	−1.76	−4.29	−0.57	3.65	1.19
Capture fisheries (finfish)	−3.54	−0.98	0.99	0.26	4.53	1.25
Capture fisheries (crustaceans)	−2.12	−0.67	4.64	1.32	6.77	1.99
Protection against coastal erosion	−17.76	−5.76	4.11	1.02	21.87	6.78
Protection against extreme weather events	1.07	0.17	10.25	2.51	9.18	2.33
Carbon sequestration	−9.24	−2.88	4.21	1.15	13.45	4.02
Tourism, education & research	−1.73	−0.50	0.33	0.08	2.06	0.59
Total	−41.27	−12.38	20.23	5.77	61.50	18.16

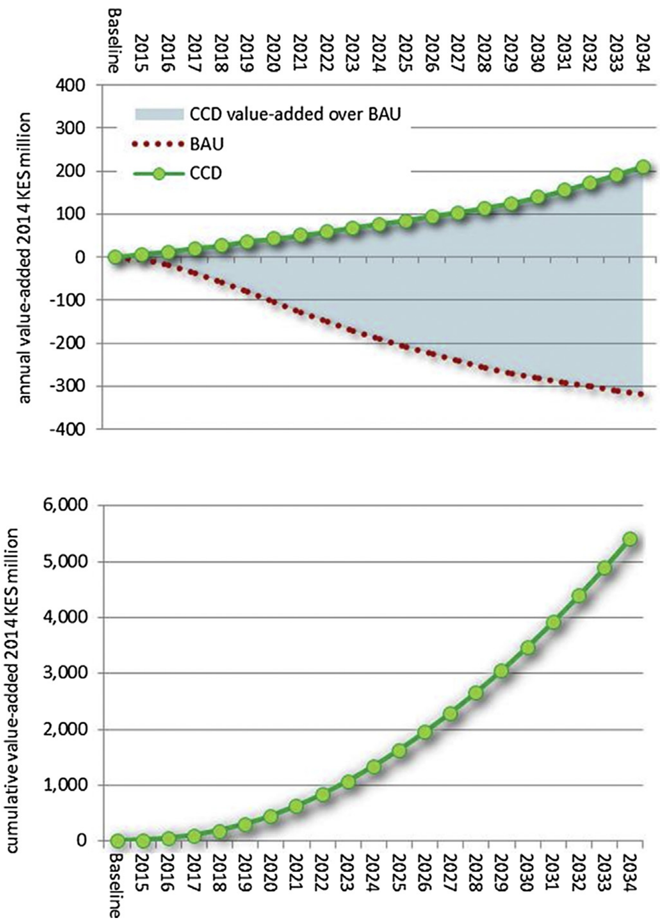

Fig. 5. CCD value-added over BAU in 2014 KES million.

4. Discussion

Our work is consistent with other studies (e.g. Barbier et al., 2011) in showing substantial value from a range of ecosystem services, with regulating services that are currently mostly without markets having the highest economic value. The average current value of USD 1166 ha $^{-1}$ yr $^{-1}$ is less than that found in most similar studies in other parts of the world. This largely represents the relative paucity of infrastructure and low cost of land in southern Kenya, as well as the conservative assumptions we used. It emphasises the importance of contextualisation in such valuation studies, since these sums are significant in the Kenyan context. Our figures are based on a large new socio-economic and ecological dataset and use real examples (such as Mikoko Pamoja) in the study area to provide estimates for market values of services. Hence we are confident that they give a useful summary of current forest values and a baseline from which to project our scenarios. However there are important omissions, including the roles of mangroves in enhancing the resilience and productivity of adjacent and connected ecosystems such as seagrass beds and coral reefs (e.g. Mumby, 2006), the existence value of biodiversity and the possible role that mangroves play in preventing saltwater intrusion into freshwater aquifers (e.g. (Ridd and Sam, 1996)). Leaving these out (because there are no reliable ways to price them) means that we capture only some of the services the forests supply. This is one reason why the figures presented here are not intended to represent the 'real' value of the forests, in some abstract way. A second is that the estimates of monetary value do not necessarily capture the importance of ecological services to key users and agents. For example services such as firewood are of great and pressing importance to the mostly poor and relatively marginalised groups (especially women) who rely on them despite being less economically important compared with regulating services, the benefits of which might accrue more broadly to society.

What the baseline economic analysis, combined with our quantitative forest modelling and qualitative scenario building, does allow is a stark illustration of the economic consequences of BAU and, conversely, of the opportunities for CCD. Recent trends are negative and our BAU scenario shows economic losses of $47 Million compared to baseline. However the policy landscape in Kenya is broadly supportive of CCD. All stakeholder participants agreed that key legislation (such as the Forest Act 2005 and the Environment Management and Co-ordination Act 1999) provided appropriate legal frameworks for forest conservation and in particular for greater community control and support. A striking feature of these storyline scenario building conversations was the level of informed optimism shown by participants, who identified plausible pathways to a better future confident in the knowledge that legislation did exist to facilitate CCD. Because much current forest degradation is caused by piecemeal incursions driven by a lack of alternatives and a failure to enforce current legislation CCD options may not require major opportunity costs or political struggles. For example most of the forest area lost over the past twenty years has not been converted to high quality alternative uses and is potentially available for restoration. Similarly the analysis of forest quality showed significant economic over-harvesting at present with most of the remaining forest timber of low quality; a properly managed harvesting regime should realise much higher returns on timber whilst also ensuring good regulating services.

We found the conceptual framework of CCD to be a useful prism through which to view current trends and options for changing them. It also provides a shared platform that can accommodate stakeholders involved in development, climate change and conservation discourses and policies and a new way to address some enduring challenges. For example integrated coastal zone management should involve a careful consideration of the costs and benefits of investment in 'hard' engineering compared with ecosystem protection as an approach to coastal protection. Although ecosystem-based approaches to hazard mitigation have a long pedigree they are still often ignored; Renaud et al. (2013, p9) conclude that 'the role of ecosystems in the context of disasters is perhaps the most overlooked component in disaster risk reduction (DRR) and development planning'. Birkmann and von Teichman (2010) reveal that differing norms and scales of impact between the climate change adaptation, development and DRR communities are partly to blame for this neglect. They recommend the wider adoption of cross-sectoral, multi-scale and integrative approaches to link DRR and climate change adaptation and to mainstream both into other activities on sustainable development. CCD combined with economic valuation provides a new way of achieving this integration. Because of its relevance to multiple sectors and scales and its topicality in policy fora, CCD was an effective focus in drawing together the wide range of stakeholders we needed to conduct the scenario building exercises and explore relevant costs and benefits.

In the context of Kenyan mangroves, 'triple wins' that combine adaptation, mitigation and development are relatively easy to identify. Because of the demonstrable importance of mangroves as carbon sinks and in the face of climate change impacts, such as sea level rise, conserving them will usually enhance mitigation and adaptation. The economic analysis of their range of services helps show how mangrove conservation can aid development too. Using the CCD concept helped make these links and synergies explicit during our stakeholder conversations; this contrasts with how development and 'conservation' are often framed as in opposition.

Hence the broad message from our work is clear; there are large economic and environmental gains to be had from a CCD scenario, in contrast to BAU, and such a scenario is plausible at least in outline and could command widespread support from appropriate stakeholders. The challenge therefore is how to help steer policy and practice towards CCD and away from BAU. Engagement with the full range of stakeholders must be a key part of the answer, and we hope the approach described here illustrates a start in that process, explicitly in Kenya and as a possible model for other areas and ecosystems. Whilst resources for the top-down enforcement of legislation will remain limited there are many opportunities for local control and empowerment that already exist in Kenyan legislation and local examples of how this approach is starting to improve mangrove conservation. The analysis presented here shows why it is in the economic, as well as social, interests of the Kenyan government to work with NGOs, civil society and others to help realise those opportunities.

Acknowledgements

We are grateful to the people of Gazi, Funzi, Vanga and Mwache communities who gave their time and expertise to this study. The participants in the East African Forum for Payments for Ecosystem Services November 2013 workshop collectively developed the CCD and BAU scenarios presented here and we thank them for their time and knowledge, and are particularly grateful to Caroline Wanjiru for her organisation of that meeting. This work was funded partly with support from the Ecosystem Services for Poverty Alleviation Programme (ESPA, Grant numbers NE/I003401/1). The ESPA programme is funded by the Department for International Development (DFID), the Economic and Social Research Council (ESRC) and the Natural Environment Research Council (NERC). It also represents an output from the iCOAST project funded by the UK Department for International Development (DFID) and the Netherlands Directorate-General for International Cooperation (DGIS) for the benefit of developing countries. However, the views expressed and information contained in it are not necessarily those of or endorsed by DFID, DGIS or the entities managing the delivery of the Climate and Development Knowledge Network, which can accept no responsibility or liability for such views, completeness or accuracy of the information or for any reliance placed on them.

References

Abuodha, P.A.W., Kairo, J.G., 2001. Human-induced stresses on mangrove swamps along the Kenyan coast. Hydrobiologia 458, 255–265.

Aburto-Oropeza, O., Ezcurra, E., Danemann, G., Valdez, V., Murray, J., Sala, E., 2008. Mangroves in the Gulf of California increase fishery yields. Proc. Natl. Acad. Sci. U. S. A. 105, 10456–10459. http://dx.doi.org/10.1073/pnas.0804601105.

Agardy, T., Alder, J., Dayton, P., Curran, S., Kitchingman, A., Wilson, M., Catenazzi, A., Restrepo, J., Birkeland, C., Blaber, S., Saifullah, S., Branch, G., Boersma, D., Nixon, S., Dugan, P., Davidson, N., Vörösmarty, C., 2005. Coastal systems, 2005. In: Hassan, R., Scholes, R., Ash, N. (Eds.), Ecosystems and Human Well-being : Current State and Trends : Findings of the Condition and Trends Working Group. Island Press, Washington DC.

Alongi, D.M., 2014. Carbon cycling and storage in mangrove forests. Ann. Rev. Mar. Sci. 6, 195–219. http://dx.doi.org/10.1146/annurev-marine-010213-135020.

Bailly, D., Willmann, R., 2001. Promoting sustainable aquaculture through economic and other incentives. In: Subasinghe, R., Phillips, M., Bueno, P., Hough, C., McGladdery, S., Arthur, J. (Eds.), Aquaculture in the Third Millennium. Technical Proceedings of the Conference on Aquaculture in the Third Millennium, Bangkok. Food and Agriculture Organisation of the United Nations (FAO), Rome.

Bann, C., 1999. Contingent Valuation of the Mangroves of Benut, Johor State, Malaysia (Paper developed as part of the preparation of an Integrated Management Plan for the Sustainable Use of the Johor Mangrove Forest, Johor State Forestry Department/DANCED/Darudec, Johor Bahru).

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L. a, Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo, G.M.E., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319, 321–323. http://dx.doi.org/10.1126/science.1150349.

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. Ecol. Monogr. 81, 169–193. http://dx.doi.org/10.1890/10-1510.1.

Birkmann, J., von Teichman, K., 2010. Integrating disaster risk reduction and climate change adaptation: key challenges – scales, knowledge, and norms. Sust. Sci. 5, 171–184.

Brown, K., Daw, T., Rosendo, S., Bunce, M., Cherrett, M., 2008. Ecosystem Services for Poverty Alleviation: Marine & Coastal Situational Analysis. Synthesis Report. University of East Anglia, Norwich.

Cohen, R., Kaino, J., Okello, J.A., Bosire, J.O., Kairo, J.G., Huxham, M., Mencuccini, M., 2013. Propagating uncertainty to estimates of above-ground biomass for Kenyan mangroves: a scaling procedure from tree to landscape level. For. Ecol. Manag. 310, 968–982. http://dx.doi.org/10.1016/j.foreco.2013.09.047.

Conservation International, 2008. Economic Values of Coral Reefs, Mangroves, and Seagrasses: a Global Compilation. Center for Applied Biodiversity Science, Conservation International, Arlington.

Crona, B.I., Rönnbäck, P., 2007. Community structure and temporal variability of juvenile fish assemblages in natural and replanted mangroves, Sonneratia alba Sm., of Gazi Bay, Kenya. Estuar. Coast. Shelf Sci. 74, 44–52. http://dx.doi.org/10.1016/j.ecss.2007.03.023.

Dahdouh-Guebas, F., Van Pottelbergh, I., Kairo, J., Cannicci, S., Koedam, N., 2004. Human-impacted mangroves in Gazi (Kenya): predicting future vegetation based on retrospective remote sensing, social surveys, and tree distribution. Mar. Ecol. Prog. Ser. 272, 77–92. http://dx.doi.org/10.3354/meps272077.

Das, S., 2007. Valuation of the Storm Protection Services of the Mangroves of Orissa. SANDEE Working Paper 25. South Asia Network for Development and Environmental Economics (SANDEE), Kathmandu.

De Mel, M., Weerathunge, C., 2011. Valuation of Ecosystem Services of the Maha Oya. Environmental Foundation, Colombo.

De Wet, J., 2004. Establishing an Economic Value for the Mangroves of the Mngazana Estuary in the Eastern Cape. Thesis (M.Sc.). University of KwaZulu-Natal, Pietermaritzburg.

Dixon, J.A., 1989. The value of mangrove ecosystems. Trop. Coast. Area Manag. Newsl. 4, 5–8.

Emerton, L., 2006. Counting Coastal Ecosystems as an Economic Part of Development Infrastructure, Ecosystems and Livelihoods Group Asia. World Conservation Union (IUCN), Colombo.

Emerton, L., 2013. Using valuation to make the case for economic incentives: promoting investments in marine and coastal ecosystems as development infrastructure. In: Essam, M. (Ed.), Economic Incentives for Marine and Coastal Conservation: Prospects, Challenges and Policy Implications. Earthscan Press, London.

Emerton, L., 2014. Valuing & Investing in Ecosystems as Development Infrastructure: Economic Analysis of Options for Climate-compatible Development in Coastal Zones of Kenya and Sri Lanka. iCOAST Technical Report. LTS International Ltd, Penicuik.

Gilman, E.L., Ellison, J., Duke, N.C., Field, C., 2008. Threats to mangroves from climate change and adaptation options: a review. Aquat. Bot. 89, 237–250. http://dx.doi.org/10.1016/j.aquabot.2007.12.009.

Giri, C., Ochieng, E., Tieszen, L.L., Zhu, Z., Singh, A., Loveland, T., Masek, J., Duke, N., 2011. Status and Distribution of Mangrove Forests of the World Using Earth, pp. 154–159. http://dx.doi.org/10.1111/j.1466-8238.2010.00584.x.

Grimsditch, G., 2011. Options for blue carbon within the international climate change framework. Sustain. Dev. Law Policy 11, 22–24.

Huxham, M., 2013. Economic Valuation of Mangrove Ecosystem Services in Southern Kenya. Report prepared for iCOAST Project. Napier University, Edinburgh.

Huxham, M., Kimani, E., Augley, J., 2004. Mangrove fish: a comparison of community structure between forested and cleared habitats. Estuar. Coast. Shelf Sci. 60, 637–647.

Huxham, M., Kimani, E., Newton, J., Augley, J., 2007. Stable isotope records from otoliths as tracers of fish migration in a mangrove system. J. Fish. Biol. 70, 1554–1567. http://dx.doi.org/10.1111/j.1095-8649.2007.01443.x.

Huxham, M., Kimani, E., Augley, J., 2008. The fish community of an east african mangrove : effects of turbidity and distance from the sea. West. Indian Ocean J. Mar. Sci. 7, 1–7.

IUCN., 2006. Valuation, Rehabilitation and Conservation of Mangroves in Tsunami Affected Areas: Rapid Mangrove Valuation Study, Panama Village in South Eastern Coast of Sri Lanka. The World Conservation Union (IUCN), Ecosystems and Livelihoods Group Asia, Colombo.

James, R., 1991. Wetland Valuation: Guidelines and Techniques. PHPA/AWB Sumatra Wetland Project Report No 31. Asian Wetland Bureau, Bogor.

Kairo, J.G., Lang'at, J.K.S., Dahdouh-Guebas, F., Bosire, J., Karachi, M., 2008. Structural development and productivity of replanted mangrove plantations in Kenya. For. Ecol. Manag. 255, 2670–2677. http://dx.doi.org/10.1016/j.foreco.2008.01.031.

Kairo, J.G., Wanjiru, C., Ochiewo, J., 2009. Net pay: economic analysis of a replanted mangrove plantation in Kenya. J. Sustain. For. 28, 395–414. http://dx.doi.org/10.1080/10549810902791523.

Kimani, E., Mwatha, G., Wakwabi, E.O., Ntiba, J., Okoth, B.K., 1996. Fishes of a shallow tropical mangrove estuary, Gazi, Kenya. Mar. Freshw. Res. 47, 857–868.

King, D., 1998. The dollar value of wetlands: trap set, bait taken, don't swallow. Natl.

Wetl. Newsl. 20 (4), 7–11.
Kirui, B.Y.K., Huxham, M., Kairo, J., Skov, M., 2008. Influence of species richness and environmental context on early survival of replanted mangroves at Gazi bay, Kenya. Hydrobiologia 603, 171–181. http://dx.doi.org/10.1007/s10750-007-9270-3.
Kirui, K.B., Kairo, J.G., Bosire, J., Viergever, K.M., Rudra, S., Huxham, M., Briers, R. a, 2013. Mapping of mangrove forest land cover change along the Kenya coastline using landsat imagery. Ocean Coast. Manag. 83, 19–24. http://dx.doi.org/10.1016/j.ocecoaman.2011.12.004.
KNBS, 2010. Kenya Population and Housing Census 2009. Kenya National Bureau of Statistics, Nairobi.
Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecol. Econ. 69, 1228–1236.
Lang'at, J.K.S., Kairo, J.G., Mencuccini, M., Bouillon, S., Skov, M.W., Waldron, S., Huxham, M., 2014. Rapid losses of surface elevation following tree girdling and cutting in tropical mangroves. PLoS One 9, e107868. http://dx.doi.org/10.1371/journal.pone.0107868.
Lugendo, B.R., Nagelkerken, I. Van, Jiddawi, N., Mgaya, Y.D., Velde, G.V.A.N.D.E.R., 2007. Fish community composition of a tropical non-estuarine embayment in Zanzibar. Tanzania 1213–1223. http://dx.doi.org/10.1111/j.1444-2906.2007.01458.x.
McNally, C., Uchida, E., Gold, A., 2011. The effect of a protected area on the tradeoffs between short-run and long-run benefits from mangrove ecosystems. PNAS 108 (34), 13945–13950.
Mitchell, T., Maxwell, S., 2010. Defining Climate Compatible Development. Policy Brief November 2010/A. Climate Development Knowledge Network, London.
Mohamed, M.O.S., Neukermans, G., Kairo, J.G., Dahdouh-Guebas, F., Koedam, N., 2008. Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). Wetl. Ecol. Manag. 17, 243–255. http://dx.doi.org/10.1007/s11273-008-9104-8.
Mumby, P.J., 2006. Connectivity of reef fish between mangroves and coral reefs: algorithms for the design of marine reserves at seascape scales. Biol. Conserv. 128, 215–222. http://dx.doi.org/10.1016/j.biocon.2005.09.042.
Mumby, P.J., Edwards, A.J., Arias-González, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H., Wabnitz, C.C.C., Llewellyn, G., 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427, 533–536. http://dx.doi.org/10.1038/nature02286.
Pamoja, Mikoko, 2011. Mikoko Pamoja Technical Specification. Plan Vivo, available at: http://www.planvivo.org/projects/registeredprojects/mikoko-pamoja-kenya/.
Pretty, J.N., Noble, A.D., Bossio, D., Dixon, J., Hine, R.E., Penning De Vries, F.W.T., Morison, J.I.L., 2006. Resource-conserving agriculture increases yields in developing countries. Environ. Sci. Technol. 40, 1114–1119.
Primavera, J., 1997. Socio-economic impacts of shrimp culture. Aquac. Res. 28, 815–827.
Ranasinghe, T., Kallesoe, M., 2006. Valuation, Rehabilitation and Conservation of Mangroves in Tsunami Affected Areas of Hambantota, Sri Lanka: Economic Valuation of Tsunami Affected Mangroves. The World Conservation Union (IUCN), Ecosystems and Livelihoods Group Asia, Colombo.
Renaud, F.G., Sudmeier-Rieux, K., Estrella, M., 2013. The relevance of ecosystems in disaster risk reduction. In: Renaud, F.G., Sudmeier-Rieux, K., Estrella, M. (Eds.), The Role of Ecosystems in Disaster Risk Reduction. United Nations University Press, New York, ISBN 978-92-808-1221-3.
Ridd, P.V., Sam, R., 1996. Profiling groundwater salt concentrations in mangrove swamps and tropical salt flats. Estuar. Coast. Shelf Sci. 43, 627–635.
Rideout, A.J.R., Joshi, N.P., Viergever, K.M., Huxham, M., Briers, R.A., 2013. Making predictions of mangrove deforestation: a comparison of two methods in Kenya. Glob. Change Biol. 19, 3493–3501. http://dx.doi.org/10.1111/gcb.12176.
Rönnbäck, P., Crona, B., Ingwall, L., 2007. The return of ecosystem goods and services in replanted mangrove forests: perspectives from local communities in Kenya. Environ. Conserv. 34, 313–324. http://dx.doi.org/10.1017/S0376892907004225.
Rounsevell, M.J., Metzger, M.J., 2010. Developing qualitative scenario storylines for environmental change assessment. Wiley Interdiscip. Rev. Clim. Change 1, 606–619.
Rudd, M.A., Lawton, R.N., 2013. Scientists' prioritization of global coastal research questions. Mar. Policy 39, 101–111. http://dx.doi.org/10.1016/j.marpol.2012.09.004.
Sathirathai, S., 1998. Economic Valuation of Mangroves and the Roles of Local Communities in the Conservation of Natural Resources: Case Study of Surat Thani, South of Thailand. EEPSEA Research Report. Environment and Economics Program for South East Asia, International Development Research Centre, Ottawa.
Sathirathai, S., Barbier, E., 2001. Valuing mangrove conservation in Southern Thailand. Cont. Econ. Policy 19, 109–122.
Tamooh, F., Huxham, M., Karachi, M., Mencuccini, M., Kairo, J.G., Kirui, B., 2008. Below-ground root yield and distribution in natural and replanted mangrove forests at Gazi bay, Kenya. For. Ecol. Manag. 256, 1290–1297. http://dx.doi.org/10.1016/j.foreco.2008.06.026.
TEEB., 2012. Why Value the Oceans?. A Discussion Paper. UNEP/GRID-Arendal Duke University Nicholas Institute for Environmental Policy Solutions, UNEP TEEB, UNEP Regional Seas Programme.
Tychsen, J., Geertz-hansen, O., Schjøth, F., 2008. KenSea – tsunami damage modelling for coastal areas of Kenya. Geol. Surv. Den. Greenl. Bull. 15, 85–88.
UNEP, 2011. Economic Analysis of Mangrove Forests : a Case Study in Gazi Bay, Kenya. United Nations Environment Programme, Nairobi.
UNEP-WCMC., 2011. Marine and Coastal Ecosystem Services: Valuation Methods and Their Application. In: UNEP-WCMC Biodiversity Series No. 33. United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC), Cambridge.
Watson, R.T., Rosswall, T., Steiner, A., Töpfer, K., Arico, S., Bridgewater, P., 2005. Ecosystems and human well-being. Ecosyst. Millenn. Assess. 5, 1–100. http://dx.doi.org/10.1196/annals.1439.003.
Wattage, P., 2011. Valuation of Ecosystem Services in Coastal Ecosystems: Asian and European Perspectives. In: Ecosystem Services Economics (ESE) Working Paper Series No. 8.. Division of Environmental Policy Implementation, United Nations Environment Programme (UNEP), Nairobi.