Can blue carbon contribute to clean development in West-Africa? The case of Guinea-Bissau

**Maria J. Vasconcelos · Ana I. R. Cabral · Joana B. Melo ·
Timothy R. H. Pearson · Henrique de A. Pereira ·
Viriato Cassamá · Tanya Yudelman**

Abstract Guinea-Bissau includes large extensions of mangroves (*Avicennia germinans* (L.)
L., *Rhizophora mangle* L., *Laguncularia racemosa* (L.) C.F. Gaertn, *Conocarpus erectus* L.,
Rhizophora racemosa G. Mey, *Rhizophora harrisonii* Leechm, and *Machaerium lunatum*
(L.f.) Ducke). These wetland forests are among the most carbon (C)-rich ecosystems in the
tropics, but deforestation processes may be contributing by as much as 10 % of carbon
emissions from the global forest sector. Therefore, avoiding mangrove deforestation can
contribute to mitigation of climate change in addition to preserving the many other vital
services that these ecosystems provide. The main objective of this study is to analyze the extent
to which the revenues generated by the C retained in standing mangroves can cover the cost of
avoiding their clearance in Guinea-Bissau. Moreover, this study aims at demonstrating the
feasibility of producing a spatially explicit national emissions baseline in a country where data,
technology, and capacity are still mostly absent. It also discusses the requirements and costs of
implementing the measuring, reporting and verification system needed to make C payments a
reality. The analysis relies on both the quantification of C stock dynamics in mangroves and
the calculation of expected returns from avoiding their clearance, given necessary investment

M. J. Vasconcelos · A. I. R. Cabral (✉) · J. B. Melo
Tropical Research Institute, Tropical Research Institute, Travessa Conde da Ribeira, 9, 1300-142 Lisbon,
Portugal
e-mail: anaicabral70@gmail.com

M. J. Vasconcelos
e-mail: maria.perestrelo@gmail.com

M. J. Vasconcelos
e-mail: maria.perestrelo@iict.pt

J. B. Melo
e-mail: Joana.lx.bm@gmail.com

J. B. Melo
e-mail:Joana.melo@iict.pt

J. B. Melo
RSet Non-governmental Organization, Lisbon, Portugal

and C prices in the market. Methodologies based on science and Earth observation technology are used to fulfil the first requirement and economic projections are applied to fulfil the second. In this paper, previously unavailable quantitative data on the deforestation trends and the biomass content of the mangroves of Guinea-Bissau are provided at national and local levels, and compared to values presented for other regions. Additionally, a national mangrove C reference emissions level (REL) is established and an assessment of the mitigation potential of mangroves, nationally and locally, is performed. The main conclusion of the analysis presented is that, if the price of avoided carbon dioxide (CO_2) emissions is above the United State Dollars (USD) 6.69 to USD 7.20/t range, and governance risk can be contained, it is possible to delineate cost-effective activities to avoid deforestation of mangroves and promote climate change mitigation activities in Guinea-Bissau using C revenues alone.

1 Introduction

Tropical mangrove forest ecosystems are considered one of the most important carbon (C) reservoirs on Earth and are reported to have up to five times more C per unit area than other forest ecosystems because of long-term carbon burial in sediments (Donato et al. 2011). In addition to their significant ecological value, mangroves have great influence on the lives of coastal populations and on local economies (Gevana et al. 2008; Fatoyinbo et al. 2008). Mangrove forests occur along tropical and sub-tropical coasts and are composed of the only known woody halophytes (Komiyama et al. 2008). In spite of their low floristic diversity, mangroves are one of the most productive ecosystems in the world (Conchedda et al. 2008) forming forests of salt-tolerant tree and shrub species, with complex food webs and dynamics (Ramsey and Jensen 1996). CO_2 is absorbed by photosynthetic activity in mangrove vegetation and a part of it is returned to the atmosphere through respiration and oxidation. The remaining C is accumulated in two pools: the living biomass (above and below ground) and soil. Soil C is by far the largest pool; in the first 1 m of sediments it averages 1,000 $MgCO_2e/ha$ in estuarine mangroves and 1,800 $MgCO_2e/ha$ in oceanic mangroves (Murray et al. 2011).

T. R. H. Pearson
Winrock International, Arlington, VA, USA
e-mail: hpereira@waycarbon.com

H. d. A. Pereira
WayCarbon, Belo Horizonte, Brazil
e-mail: hpereira@waycarbon.com

V. Cassamá
Secretariat of State for Environment and Tourism, Bissau, Guinea-Bissau
e-mail: casamavilus@gmail.com

T. Yudelman
The World Bank, 1818 H Street, NW, Washington, DC 20433, USAe-mail: Tyudelmanbloch@worldbank.org

M. J. Vasconcelos
University of Lisbon, Center for Forestry Studies, ISA, Lisbon, Portugal

When mangroves are destroyed, C stored in the living biomass and top soil is released, most of it immediately and part of it slowly through time.

Mangroves are facing widespread destruction as a result of natural hazards and human intervention (Codilan et al. 2009; Valiela et al. 2001; FAO 2007; Wilkie and Fortuna 2003). The C released from mangrove destruction and degradation may form as much as 10 % of global C emissions from the forest sector (MNRE-FDB 2010). Therefore, avoiding emissions by obviating the clearance of mangroves is a desirable and effective means of mitigating climate change. This fact has promoted the global-scale monitoring of mangroves and the development of spatially explicit estimations of mangrove biomass by remote sensing and modeling (e.g. Lucas et al. 2007; Hutchison et al. 2013). As a result, recent United Nations Framework Convention on Climate Change (UNFCCC) agreements led the Intergovernmental Panel on Climate Change (IPCC) to introduce guidelines for including blue carbon (i.e., C stored in wetland systems such as mangroves and salt marshes) within the National Appropriate Mitigation Adaptation Strategies (NAMAs) and Reduced Emissions from Deforestation and Degradation (REDD+) agendas. These strategies are conceived together as win-win financial instruments for promoting climate change mitigation and preserving the present and future benefits of ecosystem services (such as biodiversity) while improving local livelihoods. They are particularly relevant for least developed countries (LDC), such as Guinea-Bissau, which include globally and locally valuable ecosystems but do not yet have the financial, technical, and organizational means for preserving them (UNFCCC 2009).

On the Atlantic coast of Africa, mangroves lie from Mauritania to southern Angola (UNEP-WCMC 2007), with Guinea-Bissau (UNEP-WCMC 2006) having the second largest mangrove area (after Nigeria) comprising about 8 % of the national territory. Both estuarine and oceanic ecosystem types are present in Guinea-Bissau, where regional tidal conditions enable mangroves to grow as far as 100 km inland (AFROL 2002). The mangrove area in West Africa has been reported to be in considerable decline by UNEP-WCMC (2007) and by the Ramsar Convention, which is the main international cooperation initiative for the conservation and wise use of wetlands (Ramsar 2002). This convention recognizes the importance of obtaining wetland ecosystem spatial and structural information such as extent and biomass content, and it encourages the undertaking of national wetland inventories to support national policy and site management. However, knowledge gaps prevent the achievement of this objective, especially in regard to techniques and information management (Rosenqvist et al. 2007).

For Guinea-Bissau, there are reports of long-term mangrove destruction (Temudo 2009), especially in the regions of Bissau and Tombali. In spite of the recent international support for the establishment of several protected areas, financing their continued functioning requires that the ecological services they provide can be converted to sustainable financial flows. Whether C payments can be sufficient to incentivize mangrove protection in Guinea-Bissau remains to be assessed; thus, the work presented in this paper aims at: (1) rapidly producing a spatially explicit national emissions baseline in a country where data, technology, and capacity are still mostly absent; and (2) assessing the potential returns and costs of implementing the needed measuring, reporting and verification (MRV) system at both national and local level.

To characterize the business-as-usual scenario we use satellite imagery (current and historic), field data, ancillary data, and socio-economic knowledge to quantify deforestation, C stocks, and gross CO_2 emission trends in both the country and in two protected areas. Based on the baselines and on economic projections for selected market scenarios, break-even C prices are calculated and the possibility of paying for the investment and system maintenance is analyzed. The problems of accounting, nesting, and benefit sharing are discussed together with the mitigation potential of mangroves in Guinea-Bissau.

This study is part of the assessment of carbon dioxide (CO_2) removal capacity and quantification of C stocks in the forests of Guinea-Bissau, CARBOVEG-GB project and the Wetland C Market Development for Funding Coastal Communities' Adaptation to Climate Change in Sub-Saharan Africa initiative in Guinea-Bissau, which aim to facilitate the development of credible MRV systems as a means of supporting the country in the preparation of policies for sustainable forest management and climate change mitigation. Given the impending expiration of the funds provided by the World Bank and the Global Environment Facility (GEF) for managing the most relevant protected areas in Guinea-Bissau, and given the country's extreme poverty, there is a need to study the viability of including mangrove protection activities in the (NAMA) of Guinea-Bissau and also as a complementary source of continuing funding to support the work of the Instituto da Biodiversidade e Áreas Protegidas (IBAP) in the management of globally important protected areas.

2 Study area and data

2.1 Study area

Guinea-Bissau is located on the west coast of Africa, between 10°55′ and 12°40′ N and 13°40′ and 16°45′ W (Fig. 1). The area is about 36,125 km^2, comprised of continental mainland and a group of islands, the Bijagós Archipelago. The territory is characterized by flat relief, mostly lying below 50 m in elevation and subject to a large influence from tides (Catarino et al. 2001) and by two marked seasons: a dry season between November and May, and a wet season between June and October. The rainfall regime is influenced by the seasonal migration of the Inter-Tropical Convergence Zone (ITCZ).

The mangroves of Guinea-Bissau are often bordered by mudflats along the shoreline, and by rice fields inland (Altenburg and Spanje 1989). Three main types of mangrove environments exist in Guinea-Bissau (Bertrand 1999): (1) estuarine with low rainfall discharge,
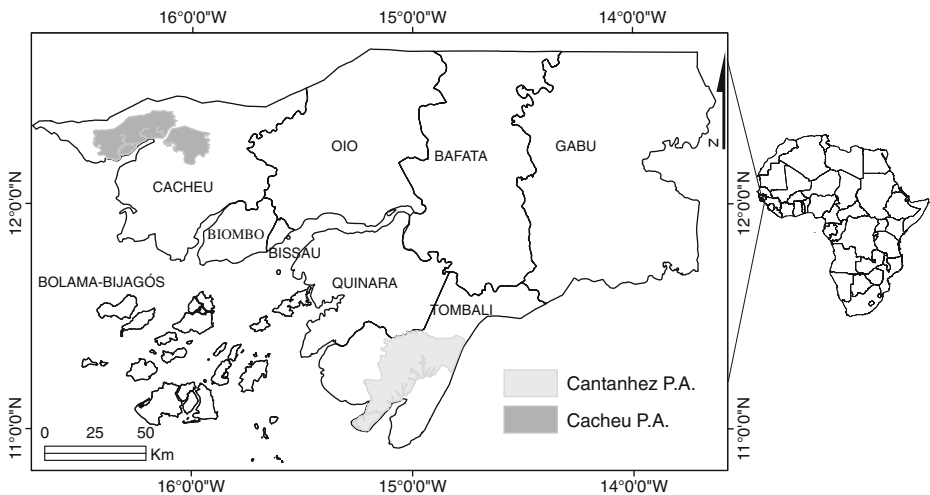

Fig. 1 Location of the study area: *on the right*, the African continent with the location of Guinea-Bissau, and *on the left* the delimitation of its administrative limits and the corresponding names. The *light gray* polygons in the Cacheu and Tombali regions show the delimitation of the Cacheu and Cantanhez protected areas

dominant in the north of the country; (2) estuarine with high rainfall discharge, mainly in the south; and (3) oceanic mangrove without rainfall discharge, present along the mainland coast and dominant in the Bijagós Archipelago. The mangrove species present are *Avicennia germinans* (L.) L., *Rhizophora mangle* L., *Laguncularia racemosa* (L.) C.F. Gaertn, *Conocarpus erectus* L., *R. racemosa* G. Mey, *R. harrisonii* Leechm, and *Machaerium lunatum* (L.f.) Ducke. The first two species are dominant in Guinea-Bissau's mangrove forests, occurring extensively in monospecific and in varying mixed stands (Catarino 2004).

The prevalent land use and farming practises associated with mangroves and the particular characteristics of the Cacheu and Cantanhez Protected Areas are summarized in the Appendix. It is important to note that cleared mangrove areas are mostly used for wetland rice production as illustrated in Fig. 2.

2.2 Data and pre-processing

2.2.1 Satellite imagery

Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) image data covering the entire territory were acquired for 1990, 1994, 2002, 2007 and 2010 (Table 7). All images were selected following the same set of criteria, namely cloud cover, quality, and date of acquisition in the late dry season. Images from 1990 were only available for the beginning of the dry season.

The Appendix describes all the imagery used and the pre-processing procedures applied to produce mosaics with a 25-m spatial resolution. This resolution was chosen for harmonizing calculations with the minimum mapping unit required for reporting forest inventory data (1 ha). The clean image mosaics of each of the 5 years above were used as input to standard image classification procedures.

2.2.2 Field data

Three national field campaigns were carried out in March/April of 2007, 2008 and 2009, covering the entire territory of Guinea-Bissau. A stratified sampling strategy based on a 250×250 m systematic grid of points with a random origin was followed. This grid was used as a basis for locating the field measurement plots for the four forest land-cover classes considered (i.e., forest, woodland, savannah-woodland and mangrove), while establishing a minimum distance between sampling points. The United Nations Food and Agriculture Organization (FAO) (2006) definition was adopted: land of more than 0.5 ha, with trees higher than 5 m and a canopy cover of more than 10 %, or trees able to reach this thresholds in situ, and which are not primarily under agricultural or urban

Fig. 2 Example of the conversion of mangrove (**a**) into wetland rice (*Oryza sativa*) paddies (**b–c**)

land use. In addition to the national field campaigns, two local data collection campaigns were undertaken in 2010 and 2012. The latter covered the regions where the Cacheu and Cantanhez Protected Areas are located with a sampling design that follows the same procedures described for the national inventory.

In each location (randomly selected over the grid) a circular plot (20 m diameter) was installed and the measurement methodology described in Pearson et al. (2005b) was followed to obtain representative data. Data collected for individual trees included species, stem height of living trees, top and basal diameter of dead trees, diameter at breast height (DBH), and total height of living trees (h). Forest inventory measurements and collection of biophysical parameters were performed in 492 forest vegetation plots, of which 83 were forest, 231 were woodland, 106 were savannah-woodland and 72 were mangrove. Figure 3 shows the distribution of measured plot locations, as well as the boundaries of the two protected areas. The Appendix includes additional information, namely the number of trees measured per species and respective DBH range.

2.2.3 Economic data

The data supporting the economic analysis were obtained from several sources and are composed of: (1) objective costs of actions related to the establishment and operation of the protected areas and which were financed by World Bank and Global Environment Facility (GEF) funds; (2) costs of operating the protected areas available from official institutions in Guinea-Bissau; (3) market information; and (4) information from the available literature. Table 1 lists the sources of all the data used for the economic analysis.
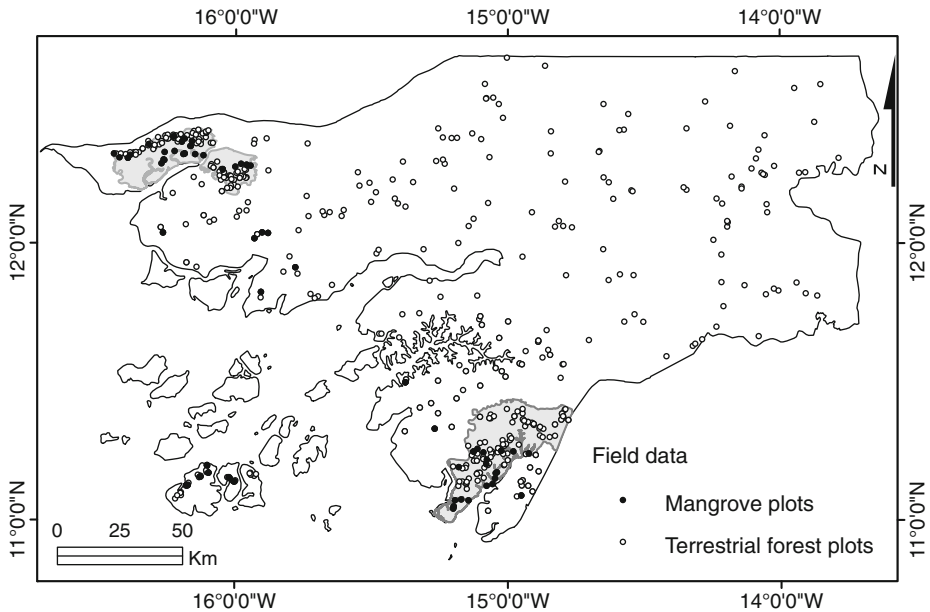

Fig. 3 Distribution of the measured plots in Guinea-Bissau during the field campaigns. The *light gray polygons* show the delimitation of the Cacheu and Cantanhez protected Areas

Table 1 Detailed input values applied in the REDD economic analysis including both expected expenditures and revenues from blue carbon

Cacheu Natural Park	Value	Unit	Reference
Total Area	74,700	Hectare	Decree 12/2000
Mangrove cover	35,467	Hectare	Calculated
Management cost	1.27	USD/ha	Vreugdenhil 2007
Implementation cost	5.46	USD/ha	Vreugdenhil 2007
Cantanhez National Park	Value	Unit	Reference
Total area	106,500	Hectare	Decree 14/2011
Mangrove cover	22,651	Hectare	Calculated
Average management cost	1.51	USD/ha	Vreugdenhil 2007
Implementation cost	3.75	USD/ha	Vreugdenhil 2007
Guinea-Bissau (national analysis)	Value	Unit	Reference
Mangrove cover	342,268	Hectare	Calculated
Average management cost	0.82	USD/ha	Vreugdenhil 2007
Average implementation cost	15.00	USD/ha	World Bank 2011
Opportunity cost of land	Value	Unit	Reference
Rice production (North Guinea-Bissau)	369.50	USD/ha	World Bank 2010
Rice production (South Guinea-Bissau)	378.60	USD/ha	World Bank 2010
REDD transaction cost	Value	Unit	Reference
Verification	35,000	USD	Cantanhez and Cacheu REDD Project[a]
BL update	200,000	USD	Cantanhez and Cacheu REDD Project[a]
Monitoring (project based)	50,000	USD	Cantanhez and Cacheu REDD Project[a]
National monitoring system	1,000,000	USD	Olsen and Bishop 2009
Monitoring (national)	0.60	USD/ha	World Bank 2011
REDD revenues			
Blue carbon price	0.60	USD/tCO$_2$e	Peters-Stanley and Yin 2013

[a] These values are based on current market prices for the afore-mentioned services. The values are derived from average prices presented by international consultancies and verification bodies to the World Bank for the development of the REDD project in Cantanhez national park and Cacheu natural park in Guinea Bissau in 2011 and 2012 (Community Based Avoided Deforestation Project in Guinea-Bissau)

3 Methods

Achieving the objectives proposed requires three main sets of activities: (1) land-cover analysis, (2) biomass/C stock assessment and establishment of emissions baselines; and (3) economic analysis of avoided mangrove conversions. The methodologies applied in these three activities cover a broad selection of scientific, technical, and technological topics, including, inter alia: models of ecological functioning and C decay in deforested mangroves; forest inventory techniques and application of allometric models; satellite image processing and map time-series analysis; statistical analysis; and economic modelling. To simplify the description of such a wide and diverse set of methods, component procedures refer to previously published work as much as possible (see supporting information in Appendix).

3.1 Land-cover analysis

An IPCC compatible legend was defined for classification of the satellite image mosaics generated for each of the addressed years: 1990, 1994, 2002, 2007 and 2010. Based on

knowledge of the main land-cover types in Guinea-Bissau and to experience gained during the project, a legend with 11 classes was established, including four forest classes: forest, woodland, savannah-woodland, and mangrove. A supervised classification technique based on a maximum likelihood classifier was applied to each image mosaic (Lillesand and Kiefer 1994) based on a training sample obtained from on-screen digitizing. Class separability was evaluated by calculating the Jeffries-Matusita (JM) pair-wise class distances (Richards and Jia 2006). After classification, clouds, cloud shadows and water were masked out in a common mask from all dates to eliminate artefacts and the effect of tides. Additionally, a 3×3 pixel median filter was applied to the classified images to eliminate spurious classified pixels, thus generating the final maps. Binary mangrove maps were obtained for each date based on reclassification of the land-cover maps produced by image classification.

Classification accuracy was assessed based on a formal comparison between a reference dataset and the corresponding image classification for 2007 and 2010. Two accuracy assessment procedures were performed. The first of these considered all forest classes separately while other grouped terrestrial forest classes (forest, woodland, and savannah-woodland) in one class, leaving mangrove (wetland forest) in a separate class. The maps produced for dates other than 2007 and 2010 were not subject to a formal accuracy assessment due to lack of ground data. However, since the entire historic image data set is subject to consistent processing, it can be assumed that the validation of the most recent maps is enough to support the validity of the historical map data set.

Mangrove cover changes correspond to areal removals and additions (loss-recovery), thus reflecting the inter-date expansion or shrinking of the total area covered by mangroves. Land-cover transitions were obtained by map algebra performed over date-paired maps (Singh 1989). In this study, mangrove transitions for the historical period (1990–2010) and sub-periods analyzed are calculated on a pixel-by-pixel basis and by producing a change detection matrix (Jensen 2004). The corresponding change rates (r) are obtained by applying the formula $r = (1/(t_2-t_1)) \times \ln (A_2/A_1)$, suggested by Puyravaud (2003), where A_1 and A_2 are the mangrove cover at time t_1 and t_2.

Land-cover transitions within the addressed protected areas were analyzed separately. This analysis was based on the land-cover maps produced for the national analysis, but for the period 2002–2010, which would be the temporal boundary required for local REDD+ projects planned to start in 2012. The national land-cover maps were cookie-cut to the required area and a separate accuracy assessment was performed to establish the local validity of the maps.

3.2 Biomass, carbon stock assessment, and emissions baseline

The above-ground biomass (AGB) of mangroves is estimated based on the collected field data and on published allometric equations applicable for these forests (see Appendix). Mangrove trees store a significant portion of their biomass in the roots, which makes it important to estimate also the below-ground biomass (BGB) based on field measurements (Komiyama et al. 2008). However, as root biomass data were not collected during the field campaigns, BGB was estimated establishing a relationship with AGB, i.e., multiplying AGB by the ratio (designated by root-to-shoot ratio - RSR) of BGB to AGB biomass (Alder and Kuijk 2009). Several RSR exist for tropical countries (Komiyama et al. 2008) but none applied to Africa. To estimate BGB in mangroves, AGB and BGB data reported by Komiyama et al. (2008) from Indonesia, Thailand, Panama, Australia and Puerto Rico (US) were compiled in order to obtain a RSR for each location. Across all available values, an average RSR of 0.61 was obtained. Total live biomass was estimated by summing AGB and BGB values.

The historical level of greenhouse gas (GHG) emissions from living biomass across the historical reference period 1990–2010 was assessed by combining the estimated CO_2

emissions per hectare of cleared mangrove (emission factor) with observed mangrove deforestation extent (activity data). This business-as-usual scenario forms the basis for projecting the rate of deforestation and associated emissions over time, thus constituting the baseline and establishing a reference emissions level (REL) (likely emissions if deforestation is not prevented). Two different approaches to project the annual area of deforestation were assessed: historical average and historical trend (i.e., linear regression of deforested area against time). The decade 2000–2010, which is more representative of recent historical changes (and thus potentially more suitable) was used to harmonize and reconcile the approaches followed nationally and across the two protected areas, integrating these two lower-level case scenarios into the national-scale assessment.

3.3 Economic analysis of avoided mangrove conversions

The economic analysis performed in this study follows the approach presented by Murray et al. (2011) with opportunity costs, implementation costs, and transaction costs assessed nationally and for the two protected areas. This approach does not take into account the value of other ecosystem services, which could surpass C monetary benefits and is thus conservative.

Following Murray et al. (2011) the C net present value (C-NPV) is established as:

$$C = \sum_{t=1}^{n} \frac{\left(GHG \text{ BenefitFlux}_t * \text{Price}\left(\$/MgCO_2e\right)_t \right) - (REDD\ Costs)}{(1+d)^t} \tag{1}$$

This expression multiplies the annual GHG benefit flux (avoided emissions) by a stream of expected C prices over a time horizon of length n ($n=25$ years) to portray currently accepted crediting periods. All incurred costs, including the opportunity cost, implementation, and transaction costs are calculated and subtracted from the expected C revenues in each year. The yearly cash flow is discounted at $d=10$ %, which is constant throughout the analysis.

The GHG benefit is defined as the emissions that would be avoided and sequestration maintained by protecting mangroves and it can be expressed as:

$$GHG \text{ benefit Flux}_t = CS_t + AvCO_{2t} - M_t \tag{2}$$

Where CS is the annual C sequestration rate, $AvCO_2$ is the CO_2 emissions avoided (t/ha), M is the annual methane emissions (t/ha), and t is time expressed in years. Methane emissions reduce GHG benefits but are comparatively very small, and are neglected in this study.

Since no C measurements were performed, we calculate the emissions avoided by adding a soil emission value obtained from the literature to the average live biomass emissions quantified in this study. Additionally, we conservatively consider all soils in the mangroves of Guinea-Bissau to be of the estuarine type. The latter average 1,000 $MgCO_2e$/ha in the top 1 m, while soils of oceanic mangroves average 1,800 $MgCO_2e$/ha in the top 1 m (Murray et al. 2011) even though they contain less C than the estuarine type at deeper levels.

The C contained in both the live biomass pool and in the sediments pool (1 m deep) is not released immediately in its entirety upon disturbance. Following Murray et al. (2011), we assume that 75 % of the C stocked in live biomass is released at deforestation time due to uprooting with exposure and burning of the mangrove root systems. The remainder decays exponentially with a half-life of 15 years. Concomitantly, in the model applied, soil C is considered to decay exponentially with a half-life of 7.5 years.

The current analysis covers the full spectrum of costs incurred by REDD initiatives: (1) the one-off establishment of protection; (2) the recurrent management costs; (3) transaction costs; (4)

the opportunity cost of land; and (5) the disbursement at the local level through a benefit sharing mechanism. The costs of establishing protection are significant and the World Bank (2011) points to a range of USD 4–15/ha. Additionally, Vreugdenhil (2007) compiled an extensive cost analysis of the establishment of these protected areas in Guinea-Bissau, discriminating both the investments demanded per park and the recurrent yearly management costs. Cacheu required investments of USD 408,200 (USD 5.46/ha) and Cantanhez USD 399,100 (USD 3.75/ha). Recurrent management costs are estimated at USD 1.27/ha in Cacheu and USD 1.51/ha in Cantanhez. These figures are very close to the lower bound values presented by the World Bank (2011) and similar to the USD 1/ha value observed in other studies (Murray et al. 2011).

The opportunity costs correspond to the income that is foregone by not converting areas of mangroves into alternative cash-producing uses. In Guinea-Bissau those alternatives are mainly wetland rice production and fuel wood collection, the latter being comparatively irrelevant (Ferreira and Faria 2013, unpublished data). Cost studies show that Guinea-Bissau has a substantial comparative advantage in rice production and also has a number of unexploited opportunities to increase both the area and the yields in the rice sector. The report (World Bank 2010) presents returns of USD 369.5 to USD 378.6/ha in 2007 for northern and southern Guinea-Bissau respectively, and provides an estimated area of potentially suitable mangrove rice production of over 106,000 ha. This value, which corresponds to approximately one third of the total mangrove area in Guinea-Bissau, is used as a cap to mangrove deforestation in the country, establishing the maximum available agricultural area in the model.

In this study, two different scenarios are analyzed: the national level and the two protected areas. Thus, with two scenarios, four different cash flows are generated: one national, one for the two protected areas handled together, and one for each protected area in a stand-alone mode. Additionally, whereas C revenues per hectare of standing mangrove are the same in all cases, eligible costs vary slightly, reflecting differences in scale and corresponding monitoring requirements. The calculations are based on the data shown in Table 3 and on the market value of REDD+ credits in 2012, which averaged USD 10.00 (Peters-Stanley and Yin 2013). In addition, a break-even analysis is performed to evaluate the minimum price for C that will keep the REDD+ initiatives operational.

4 Results

4.1 Land cover and land-cover change

The overall accuracies obtained for the 2007 and 2010 binary mangrove maps are 94.2 and 95.8 %, respectively. This was to be expected given the fact that the pair-wise spectral separability between class mangrove and all other land-cover classes showed a minimum value of 1.62. In 2010, mangroves covered a total of 310,694 ha of the territory of Guinea-Bissau, which corresponds to a 23.5 % expansion since 1990. Cacheu is the region where mangroves have the largest extent, covering about 35 % of the total area, followed by Tombali with 20 %, Bolama/Bijagós with 11 % and Quinara with 10 %.

Figure 4 illustrates the evolution of the area of mangrove forests in Guinea-Bissau between 1978 and 2010 and the net annual mangrove change rate in the reference period (1990–2010) and in the sub-periods analyzed (1990–1994, 1994–2002, 2002–2007 and 2007–2010), and Fig. 5 shows the mangrove spatial distribution in 2010 (Fig. 5a) and its evolution by region between 1990 and 2010 (Fig. 5b). The first two values (1978 and 1983) were obtained from Altenburg and Spanje (1989), who based their estimates on aerial photography and field work.
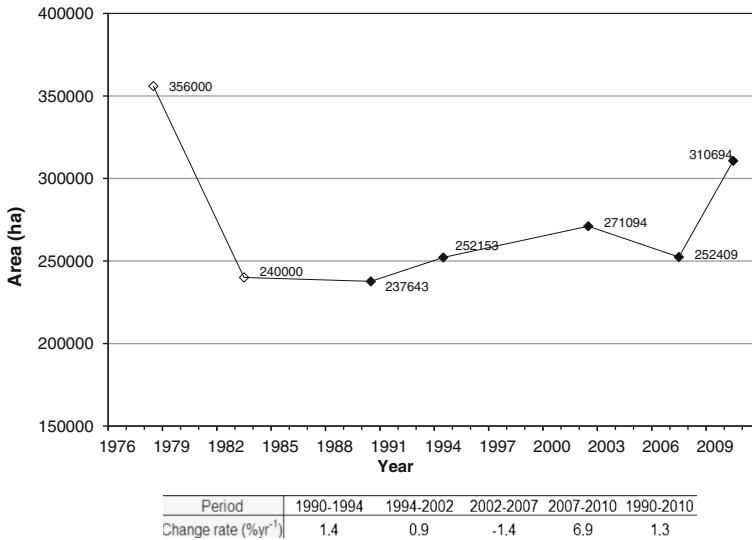
Period	1990-1994	1994-2002	2002-2007	2007-2010	1990-2010
Change rate (%yr^{-1})	1.4	0.9	-1.4	6.9	1.3

Fig. 4 Evolution of the area of mangrove forests in Guinea-Bissau in the period 1978–2010 and net change rates in the period 1990–2010. The first two areas displayed in unfilled diamonds correspond to 1978 and 1983 and were obtained from Altenburg and Spanje (1989), and the values for the other dates, shown in *solid diamonds*, were obtained in this study

4.2 Biomass, carbon stock assessment, and emissions baseline

Total mean live biomass of mangrove forests (± standard error of the mean, SEM), computed as the sum of the trees AGB (51.5±5.0 Mg/ha) and BGB (31.4±3.1 Mg/ha) in the 72 mangrove plots, was estimated to be 82.9 (±8.0) Mg/ha. The corresponding C stock is 39.0 (±3.8) MgC/ha, which results in 143.0 (±13.9) MgCO$_2$e/ha. Therefore, by adding the soil C in the top 1 m to the live biomass C stock, we obtain a value of 1,143 MgCO$_2$e/ha. This is the value used to calculate avoided emissions following the model described in point 3.3.

The total mangrove live biomass C storage across the country, assessed by combining both the afore-mentioned live biomass C stock and mangrove extent in each land-cover map produced, is presented in Table 2. The reported data show an increase of 2.9 (±0.28) MtC between 1990 and 2010. Similarly, the C stocks in the living biomass (MtC ± SEM) of the two protected areas in the historical reference period 2002–2010 are shown in Table 3.

The national gross deforestation rates in each sub-period and in the entire period (1990–2010), the corresponding number of hectares deforested, and the corresponding emissions (tCO$_2$/ha) are shown in Table 4. In this national analysis, the mangrove area existing in 1990 was used as a benchmark. Thus all areas deforested refer to the 1990 area and those areas corresponding to an expansion into new areas outside this initial polygon are not included.

As is reasonable, given a no-action scenario, future deforestation can be estimated based on a linear projection of past deforestation (historical trend). Although a clear trend in decreasing deforestation exists there are no significant regression results from this analysis (Fig. 6). Thus, the mean rate between 2002 and 2010 shown in Table 5 is used.

To encompass sub-national business-as-usual baselines across the two protected areas, the national REL presented previously was revised, and an analogous analysis was performed for the historic period 2002–2010, now using the 2002 mangrove map as a benchmark. The results for gross deforestation and associated emissions in each sub-period and in the total historical
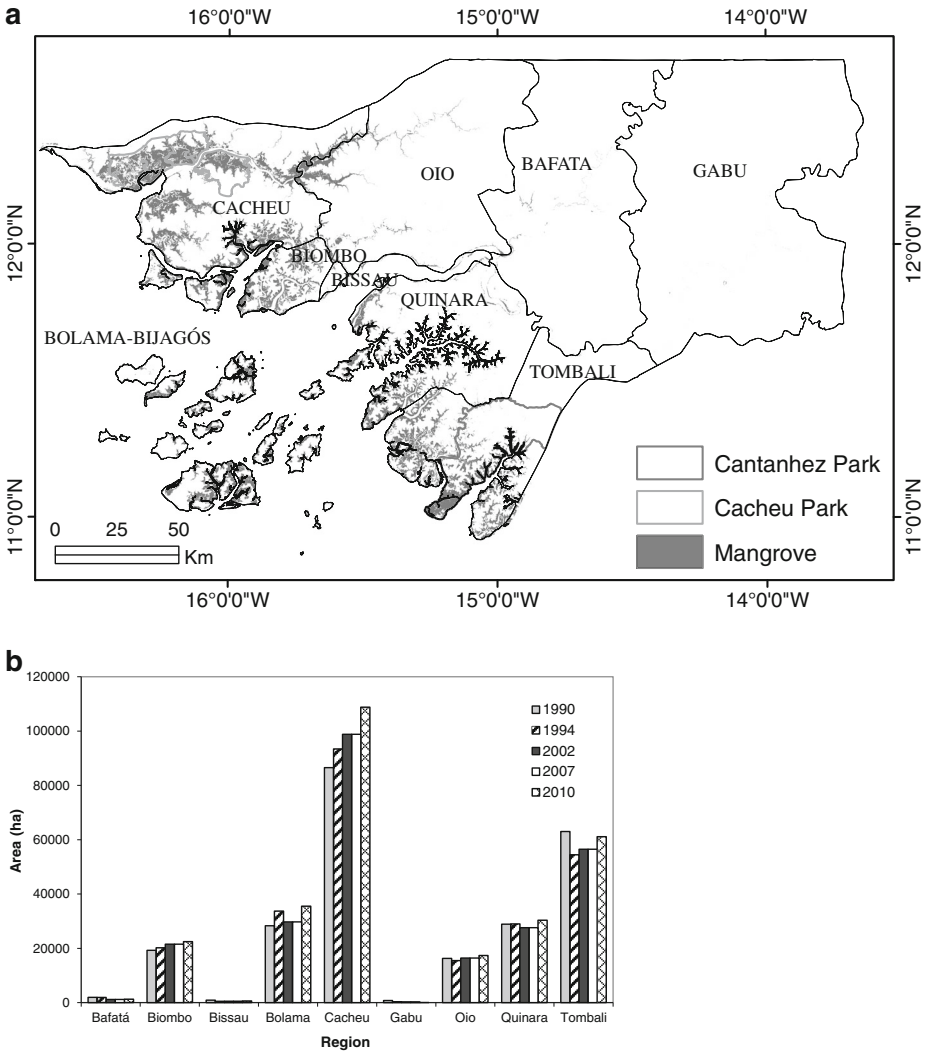
Fig. 5 (**a**) Mangrove spatial distribution in Guinea-Bissau in 2010; (**b**) Mangrove evolution by region, between 1990 and 2010

reference period are shown in Table 5. The Cacheu protected area consistently shows a lower deforestation rate than national figures. The Cantanhez protected area also shows slightly

	Year	Total area (×10³ ha)	Stock (MtC)
Table 2 C stocked in the living biomass of the mangrove forests of Guinea-Bissau across the historical reference period 1990–2010	1990	238	9.28 (±0.90)
	1994	252	9.83 (±0.95)
	2002	271	10.57 (±1.02)
	2007	252	9.83 (±0.95)
	2010	311	12.13 (±1.18)

Table 3 C stocked in the living biomass of the mangrove forests, in the two protected areas (Cacheu and Cantanhez) across the historical reference period 2002–2010

Year	Protected area	Total area (ha)	Stock (MtC)
2002		35,782	1.40 (±0.14)
2007	Cacheu	35,160	1.37 (±0.13)
2010		35,467	1.38 (±0.13)
2002		21,470	0.84 (±0.08)
2007	Cantanhez	20,554	0.80 (±0.08)
2010		22,651	0.88 (±0.09)

lower rates for the second sub-period, with a sharp decrease from the first period, when it had a higher value than the country as a whole. The same happens if we look at the projected area of baseline deforestation, calculated as the total area deforested divided by the number of years of the historical reference period.

An historical reference period with only three time points does not allow the estimation of future emissions from deforestation based on a linear projection of past emissions. Therefore, in all cases (nationally and for the two lower-level case scenarios), the historical average shall be used in subsequent analysis, and the same mean rate shall be used for each year of the baseline period.

4.3 Economic analysis of carbon conservation investments

Given baseline deforestation rates, the total deforested area after 25 years is 9,240 ha in Cacheu, 6,795 ha in Cantanhez, and 102,680 ha nationwide; all values are below the established cap (see section Economic analysis of avoided mangrove conversions) defined as the area of mangrove at risk of conversion into agriculture. At the price of USD 10.00/ $MgCO_2e$, all the four cash flows show positive returns assuming prevention of all baseline deforestation. Table 6 summarizes the results of the analysis.

A sensitivity analysis was performed to evaluate the C price to deliver a net present value (NPV) equal to zero (Table 6). This represents the minimum value that C transactions need to achieve to cover all costs related to mangrove conservation. The break-even C price ranged from USD 6.69 to USD 7.20/t.

5 Discussion

According to our study, mangrove extent in 2010 was estimated at 310,694 ha, which contrasts with the 199,000 ha reported in the Forest Resource Assessment 2010 (FRA2010) (FAO 2010). This difference may be a result of the different methods applied in the two studies. The

Table 4 National gross deforestation rates of mangrove forests, in each sub-period analyzed and in the entire period (1990–2010), the corresponding number of hectares deforested and the corresponding emissions

	Gross deforestation		Gross emissions
	%/year	ha/year	$MtCO_2e$/year
Guinea-Bissau			
1990–1994	1.2	2,857	408 (±39.60)
1994–2002	0.1	175	25 (±02.43)
2002–2007	0.6	1,436	205 (±19.90)
2007–2010	0.4	833	119 (±11.55)

Fig. 6 Future deforestation and emissions estimation based on a linear projection of the past deforestation trend. The *black line* with a negative slope corresponds to the linear trend of decreasing deforestation that would apply if the coefficient of determination was acceptable. The *gray lines* correspond to constant average rates established from the 1990–2010 and 2000–2010 historical periods respectively
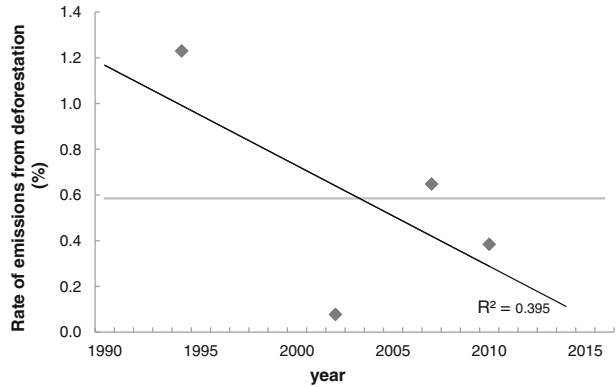
FRA2010 mangrove area was estimated by extrapolation from population data obtained in the 1990s (Diombera 1999), whereas in our study, mangrove area was obtained by a wall-to-wall cartography generated by processing medium-high resolution satellite imagery of the year 2010. A recent global assessment performed by Giri et al. (2011), also based on Landsat data, reports a total mangrove area of 338,652 ha for Guinea-Bissau for the year 2000. Our value, on the other hand, was 271,094 ha for 2002 and FRA2010 reported 221,000 ha for 2000. A close analysis of possible reasons for this discrepancy would require more information on the dates and processing approach of the imagery in the Giri et al. (2011) global study.

Even though it never reaches the extent reported by Altenburg and Spanje (1989) for 1978, mangrove coverage in Guinea-Bissau increased about 23.5 % during the 20-year period scrutinized in this study. In opposition to this trend, FAO (2007, 2010) found an average mangrove loss of about 20 % in the same period: 11 % (−1.1 %/year) between 1990 and 2000 and 5 % (−1 %/year) between 2000 and 2010. However, the latter results are not based on direct observations. Our results are consistent with findings by Conchedda et al. (2008), who studied mangrove dynamics by remote sensing in a neighbouring region of south-western Senegal (1986–2006), also reporting increases in mangrove extent.

Table 5 Gross deforestation rates of mangrove forests, in each subperiod analyzed and in the total historical reference period (2002–2010), the corresponding number of hectares deforested and the corresponding emissions for Guinea-Bissau and the two protected areas (Cacheu and Cantanhez)

	Gross deforestation		Gross emissions
	%	ha/year	$MgCO_2$/year (± SEM)
Guinea-Bissau			
2002–2007	1.7	4,310	616,265 (±59,750)
2007–2010	0.8	1,953	279,250 (±27,075)
Historical average	1.2	3,426	489,867 (±47,495)
Cacheu			
2002–2007	1.6	539	77,069 (±7,472)
2007–2010	0.4	119	17,015 (±1,650)
Historical average	1.0	382	54,549 (±5,289)
Cantanhez			
2002–2007	2.2	452	64,629 (±6,266)
2007–2010	0.4	79	11,296 (±1,095)
Historical average	1.3	312	44,629 (±4,327)

Table 6 Summary of the economic analysis results. The cash flows were prepared for each Protected Area individually and two alternative scenarios also evaluated, one where both protected areas are analyzed as one project and a national economic analysis

	Discount rate (%)	Period of analysis (years)	Net present value (USD)	Break even C price[a] USD/MgCO$_2$e
Cacheu Park	10	25	5,967,025.10	6.79
Cantanhez Park	10	25	3,330,583.48	7.20
Combined (Cacheu + Cantanhez)	10	25	10,098,721.64	6.69
Guinea Bissau	10	25	59,430,916.62	6.69

[a] Break even carbon price for all cash flows

The deforestation rates established for the two protected areas since 2002 show that the initially high rates (similar to the national value) become lower than the national figure in the later period. Even though more evidence is needed to confirm a sustained trend, this observation is consistent with the protected areas' establishment dates (2000 for Cacheu and mid-2000s with legalization in 2011 for Cantanhez) and associated subsequent management and conservation activities through community monitoring, stakeholder involvement, and induction of socially accepted new small businesses (World Bank 2004).

The costs of benefit distribution for a C-based financial implementation were included in our analysis, as were the costs of installing and operating a coherent and hierarchically integrated MRV and C accounting system for Guinea-Bissau. Nevertheless, the resources used for benefit distribution can be seen as an investment instead of a cost. In fact, the development of alternative, sustainable local businesses and improvement of community well-being actually correspond to further benefits and profits. This cost/investment in our study is set to be 10 % of the total C revenues and corresponds to USD 12,390,484 spent in local communities during the 25-year period considered (NPV USD 3,723,820). This investment is also crucial because local organization and participation induced by C revenues can constitute the foundation of a nested approach that allows reconciling the emission abatements reported locally with those observed at national level.

The living biomass results obtained in this study (82.9 Mg/ha) show that the C stock contained in the mangrove forests of Guinea-Bissau is not as high as that found in other regions of the globe or the African continent. For example, Albert et al. (2012) report a range of AGB values of 190–430 Mg/ha in a study performed in the Solomon Islands; Slim et al. (1996) obtained a value of 249 Mg/ha for a *R. mucronata* forest in Kenya, and Steinke et al. (1995) found values of 94.5 Mg/ha for *A. marina* mangroves in South Africa. Guinea-Bissau and Senegal lie in the upper latitude limits of the mangrove geographic range for the western coast of Africa and thus host less-developed mangrove stands than those present at lower latitudes. For instance, the mangroves of *R. racemosa* present in the estuary of the Kwanza River (Angola) show considerably higher and wider trees (20–25 m tall; L. Catarino, personal communication, 2011) and are similar to those present in Nigeria which are reported to have 150 Mg/ha (Abere and Ekeke 2011). Nevertheless, the largest C pool in mangrove stands is in the soil which is a poorly studied reservoir. In our calculations, we used a conservative value reported for the C stored in the top 1 m of mangrove soils (Murray et al. 2011). This may result in lower-than-real emissions predicted for destroyed mangroves and thus entail an underestimation of the true mitigation potential of Guinea-Bissau mangroves. Despite the uncertainty regarding international climate policy agreements and regulations, more work is needed to realize the true value obtained from including soil C in the C offset framework.

Even though C stocks in the mangroves of Guinea-Bissau are comparatively low, the low costs of establishing and developing C projects, together with a potentially large area at risk, make the effort for issuance of C credits competitive and attractive. The NPV of the C stored in the mangroves at risk is high enough to motivate conservation and protection projects in the context of the C market. However, C revenues are finite and thus ensuring future financial sustainability for mangrove protection is crucial. One solution might be the constitution of a fund to which C revenues could be directed and which would have an endowment structure for securing future revenues from these and other diverse sources. This fund, to be effective, must guarantee that the benefits obtained can reach the local organizations, communities, and families through credible monitoring and benefit-sharing systems. Given Guinea-Bissau's insufficient technical capacities and governance difficulties, this implies non-negligible up-front investments. Additionally, the unreliable evolving world of international finance for climate change and its inherent complexities, with real risks of a C market collapse (The Munden Project 2011), are a deterrent for investment.

Previous regional studies such as Siikamäki et al. (2012), suggested a high C supply potential for the mangroves of Guinea-Bissau. This suggestion is based on the fact that the areas of mangrove are large and the published deforestation estimates are high. However, our study does not confirm the high mangrove loss rate. Moreover, the lack of organizational and technical capacity in Guinea-Bissau, in addition to poor governance, can sharply increase risks, thus augmenting the monetary buffer required per ton of avoided C emission. Therefore, international financing such as that available through the Global Environment Facility, REDD+, or NAMAs, is crucial for improving the country's capacity and for promoting demonstration activities, while steering official institutions and individuals towards participatory procedures. Accessing these funds, which are nominally granted to developing countries and in particular to LDCs, is currently difficult for Guinea-Bissau. However, Guinea-Bissau has a group of emerging leaders, many returning from the diaspora, who understand these dynamics and are coming together to find adequate support and mobilize change (Glencorse and Yarwood 2013).

Guinea-Bissau's potential supply of avoided emissions from mangroves, with break-even C prices as low as USD 6.69, together with the recognized global importance of its mangroves for biodiversity conservation and for local livelihoods, should motivate further investment to ensure that future C offsets become tagged with CCB standards (CCBA 2013). This is especially relevant given that prior studies indicate that adding a biodiversity focus to C projects increases costs by only up to USD 1/t of CO_2 (Siikamäki et al. 2012) and the added value of objectively quantifying non-C benefits may substantially exceed this cost. This approach could increase revenues, given the higher market value of CCB-tagged C credits, improve conservation goals, and increase robustness to governance considerations.

Extending capacity-building efforts and strengthening institutional infrastructure from project-based funding can help alleviate the current limitations (Siikamäki et al. 2012), but is limited to short periods and disperses the structural long-term goals into atomized and overlapping initiatives. Even though the legal system, law enforcement, and land-use decisions are sovereign matters that can only be addressed by the country itself, organizing the manner in which climate change adaptation funds and biodiversity projects are delivered to Guinea-Bissau can be aided through coordinated external support. For this reason, the steps already taken with the creation of the BioGuinea Foundation, which aim at constituting a sustainable fund supporting the long-term financing of biodiversity conservation and local livelihood improvements, can constitute an important stepping stone in this process.

6 Conclusions

Mangroves are a significant source of emissions in Guinea-Bissau while also being a critical provider of broad ecosystem services. Given our results, and notwithstanding the comparatively low C stocks measured, reducing C emissions from Guinea-Bissau's mangroves is a truly significant development opportunity. Adequate and efficient use of this opportunity can contribute decisively to the long-term maintenance of the country's natural patrimony, also improving governance and the conservation of globally relevant biodiversity.

This study contributes to demonstrating that protection and enhancement of mangroves offers a cost-effective means to mitigating climate change that can have many other associated benefits, both locally and at the national level. That said, the potential revenues from mangrove C, even though significant, are finite. Thus it is essential that effective long-term financial mechanisms be conceived and put in place to guarantee stable financial flows into the conservation of mangrove forests in Guinea-Bissau. The international financing currently available for climate change mitigation and adaptation in developing countries can be mobilized to implement such mechanisms. As demonstrated by the financial analysis, a price of USD 10.00/t of blue carbon can provide revenues of 80 % over the cost of installing protection and managing the coastal area of Guinea-Bissau. These revenues could be directed to a facility with an endowment structure for securing further funds after the C benefits are fully monetized.

Appendix

Land use and the protected areas

Land cover conversion in Guinea-Bissau is mostly determined by dispersed subsistence needs, opportunistic market drivers, and individual or community initiatives. Almost 80 % of the population of Guinea-Bissau resides in the coastal zone where most economic activity is concentrated, resulting in an increased pressure on coastal and marine resources. Mangroves are cleared mostly for installing rice production paddies, which entail long waiting periods to let rainwater dissolve the salts and acids, and there are reports of long-term relevant mangrove

destruction (Temudo 2009), especially in the regions of Bissau and Tombali. More recently, with the expansion of cashew plantations and population growth in the coastal region, the traditional patterns of rice plantation have been changing.

Table 7 Date, path and row of the Landsat images used in this study

Satellite (Sensor)	Acquisition date DD-MM-YY	Scene or point identifier	
		Path	Row
Landsat TM	29-11-90	203	51
	05-11-90	203	52
	28-11-90	204	51
	28-11-90	204	52
	05-12-90	205	51
	22-04-94	203	51
	22-04-94	203	52
	13-04-94	204	51
	13-04-94	204	52
	04-04-94	205	51
	04-04-02	203	51
	04-04-02	203	52
	11-04-02	204	52
	18-03-02	205	51
	09-03-07	203	51
	09-03-07	203	52
	28-02-07	204	51
	28-02-07	204	52
	07-03-07	205	51
Landsat ETM+	11-04-02	204	51
	02-01-10	205	51
	18-01-10	205	51
	19-02-10	205	51
	07-03-10	205	51
	23-03-10	205	51
	27-01-10	204	51
	12-02-10	204	51
	28-02-10	204	51
	16-03-10	204	51
	01-04-10	204	51
	27-01-10	204	52
	12-02-10	204	52
	28-02-10	204	52
	16-03-10	204	52
	01-04-10	204	52
	20-01-10	203	52
	09-03-10	203	52

The Cacheu River Natural protected area was established in 2000 by Decree 12/2000. It is located in north-western Guinea-Bissau, and is composed of two discrete portions of land separated by the Cacheu River. The protected area occupies 74,700 ha, 68 % of which are covered by mangroves, including the largest continuous mangrove patch of West Africa (MNRE 2008). The other predominant vegetation cover type in this region is open forest, which includes palm groves.

The need for the Cantanhez National protected area was identified in 2001, but it was only made official in 2011 by Decree 14/2011. This protected area covers 106,500 ha located in south-eastern Guinea-Bissau. Mangroves cover a large proportion of the area, particularly to the south and west. Ecological conditions, with more than 2,000 mm of annual rainfall and deep soils, are favorable for the establishment of forest vegetation which develops there, forming the northernmost patches of sub-humid forest in West Africa. The most typical soils are Ferralsols in dry areas and Fluvisols in periodically flooded zones (Teixeira 1962).

Given that the territory of Guinea-Bissau includes globally relevant biodiversity (e.g. the Bijagós Archipelago) there has been international support for the establishment of protected areas since the early 2000s. However, due to the impending end of the main funding sources, managing the protected areas is increasingly dependent on the locally generated revenues, and this is the reason for the current interest in the UNFCCC financial mechanisms such as REDD+, which provide incentives for the conservation of standing forests.

Image processing, field data, collection and calculation of biomass

The pre-processing of the Landsat scenes was performed to produce clean image mosaics, one for each of the analyzed years. Distortions were removed based on ortho-rectified reference images downloaded from the United States Geological Survey (available at http://glovis.usgs.gov/) for UTM (Universal Transverse Mercator) Zone 28 North, WGS84 (World Geodetic
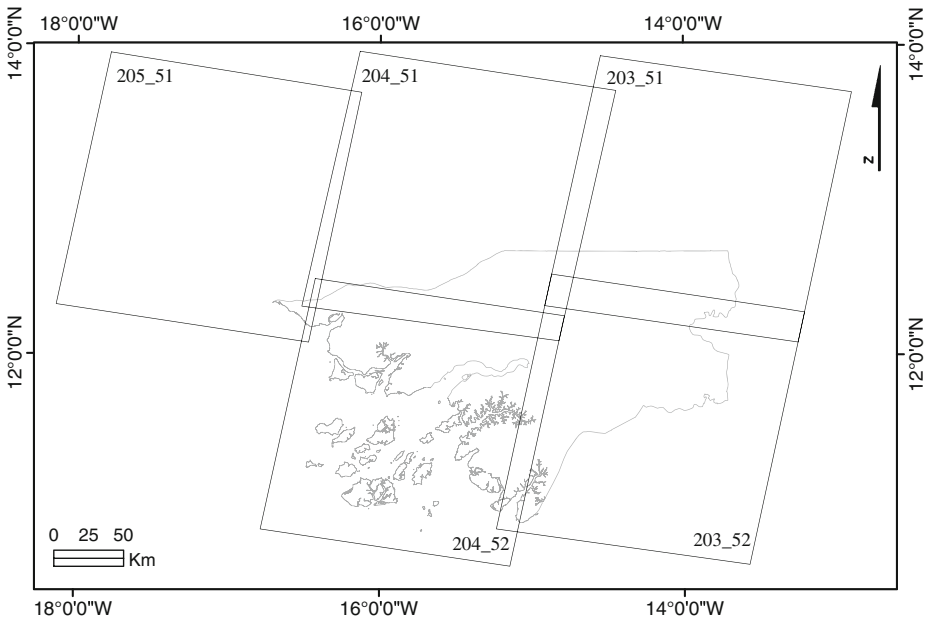

Fig. 7 Spatial arrangement of the Landsat scenes (Path_Row)

Table 8 Number of mangrove trees measured, DBH range/mean, and AGB range/mean, per species and total

Sampled species	Number of trees	DBH range (cm)			AGB (kg)		
		Min.	Max.	Mean	Min.	Max.	Mean
A. germinans	580	5.0	64.2	12.1	5.2	3,049	89.5
R. mangle	671	5.0	38.2	12.0	9.5	1,39	102.5
L. racemosa	4	5.1	8.6	7.1	6.0	22	15.0
Mangrove trees	1,255	5.0	51.8	12.0	5.2	3,049	96.3

System 84) datum, with a choice of 25-m spatial resolution for harmonizing calculations with the minimum mapping unit required for reporting forest inventory data (1 ha).

The gap fill methodology by Scaramuzza et al. (2004) was applied to the 2010 images affected by the malfunctioning of the Scan-line corrector mechanism and the relative radio-metric calibration procedure by Phua et al. (2008) was applied prior to building the mosaic layers for the years analyzed. The need for this procedure explains the high number of images used to compose a complete mosaic for the dry season of 2010. Table 7 lists the images used and Fig. 7 shows the spatial arrangement of the Landsat scenes (path, row).

To determine the number of plots needed for tree measurements and biomass estimation in each forest land-cover class, three factors were considered: (1) the variance of measured tree biomass within each class obtained from a trial sub-set using preliminary data from 2007; (2) the estimated area covered by each class; and (3) a targeted error level (95 % confidence interval within 10 % of the mean; Pearson et al. 2005a). After compilation and cleaning of the tree measurements data base, allometric models were applied to calculate biomass.

Allometric models are functions that relate AGB to measurable variables at tree level, such as Diameter at Breast Height (DBH), height (h) and wood density (ρ) (Brown et al. 2005; Vieira et al. 2008). The number of mangrove trees measured in the field campaigns, the corresponding measurable variable DBH and the corresponding AGB are shown in Table 8. In this study, equations published by Fromard et al. (1998) for the three species found in Guinea-Bissau, *A. germinans*, *Rhizophora* spp. and *L. racemosa*, are used. These authors use a logarithmic model that relates AGB (y), in kg, and DBH (x), in cm, in the form of $y = \beta x \alpha$. The equations used to estimate AGB of mangrove species are shown in Table 9.

For each measurement plot, total biomass of all individual trees is added to obtain the total tree biomass per plot. Data obtained at plot level is then extrapolated to the area of 1 ha (10,000 m^2) by calculating the proportion that is occupied by a given plot using a dimensional scaling factor (e), defined by the equation $e = (10,000/\pi * r^2)$, where r is the plot radius in meters. A standard C factor of 0.47 (IPCC 2006) was used to convert total mangrove biomass to mangrove C stock, while the total C stock in the mangrove forests of Guinea-Bissau was derived by expanding the values observed at plot level. This was then converted in tonnes per

Table 9 Allometric equations used to estimate AGB of mangroves species (Fromard et al. 1998)

[a]Specie	Equation (kg)	DBH range (cm)
A. germinans	$y = 0.1400 \, DBH^{2.4}$	4–42
Rhizophora spp.	$y = 0.1282 \, DBH^{2.6}$	1–32
L. racemosa	$y = 0.1023 \, DBH^{2.5}$	1–10

[a] Minimum diameter of trees (*DBH*) sampled in the Guinea-Bissau field campaigns was 5 cm

hectare, to the total area of mangrove obtained in the maps for several dates. This is a very simple method that does not take into account structural variations (Fatoyinbo and Armstrong 2010). By applying an emission factor (44/12) to C stocks, an estimate of potential CO_2 emissions by forest removal can be obtained (IPCC 2006).

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