North Brazil Shelf Mangrove Project

Blue Carbon Feasibility Assessment

CONSERVATION INTERNATIONAL

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TABLE OF CONTENTS

TABLES

FIGURES

ACRONYMS / ABBREVIATIONS

Acronym	Signification
AGB	Aboveground Biomass
ALOS PALSAR	Advanced Land Observing Satellite Phased Array type 1-Band Synthetic Aperture Radar
ARR	Afforestation, Reforestation, and Revegetation
BGB	Belowground Biomass
C	Carbon
CH_4	Methane
CLME+ SAP	Caribbean and North Brazil Shelf Large Marine Ecosystems Strategic Action Program
cm	Centimeter
CO_2e	Carbon Dioxide equivalent
DBH	Diameter at Breast Height
GFC	Guyana Forestry Commission
GHG	Greenhouse Gas(es)
GRIF	Guyana REDD+ Investment Fund
GYD	Guyana Dollar
ha	Hectare
HFLD	High Forest Low Deforestation
IPCC	Intergovernmental Panel on Climate Change
LME	Large Marine Ecosystem
m	Meter
Mg	Megagrams
MRVS	Monitoring, Reporting and Verification System
MUMA	Multiple Use Management Areas
N_2O	Nitrous Oxide
NAO	North Atlantic Oscillation
NBS	North Brazil Shelf
NIMOS	National Institute for Environment and Development in Suriname
PSU	Practical Salinity Unit
REDD	Reducing Emissions from Deforestation and Forest Degradation
REDD+	Reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries
REDD+MF	REDD+ Methodology Framework
R-P P	Readiness Preparation Proposal

SRD	Suriname Dollar
Tg	Teragrams
UNDP	United Nations Development Program
USD	United States Dollar
VCS	Verified Carbon Standard
WRC	Wetland Restoration and Conservation

1 Executive Summary

Blue carbon ecosystems, which are coastal vegetated ecosystems including mangroves, tidal wetlands, and seagrass beds, sequester significant amounts of carbon within the soil for tens to thousands of years. In addition to carbon storage, blue carbon ecosystems offer many ecosystem services, including habitat and nurseries for fish, birds, invertebrates, and mammals, shoreline protection, and land-building capacity, among many others. Once disturbed and drained, the substantial soil carbon stores can be released at significant rates. These ecosystems also will be significantly impacted by predicted accelerated sea level rise, especially if abutting hardened structures. This Blue Carbon Feasibility Assessment presents current data specifically on mangrove distributions, carbon stocks, and deforestation in Guyana and Suriname along the North Brazil Shelf Large Marine Ecosystem (NBS-LME), and outlines the methodologies and feasibility of developing carbon finance projects.

Mangrove ecosystems along the NBS-LME are dominated by *Avicennia germinans*, with two others, *Laguncularia racemosa* and *Rhizophora mangle*, also present. The development of mangrove stands is largely dependent on the presence of large mud banks originating from the Amazon River that migrate slowly westward and dampen wind wave effects. Mangroves grow quickly once established given the correct inundation and salinity regimes. Stands typically last for 60 to 70 years until the mud bank moves and largescale erosion occurs. While deforestation historically has occurred to a larger extent in Guyana than Suriname, current deforestation rates are low; however, mangrove regeneration in these areas is not occurring naturally and restoration efforts have been implemented, with varying success.

Most mangrove research along the NBS-LME has occurred in French Guiana, and the rich dataset of tree and soil carbon stocks was utilized in estimating carbon stocks in Suriname and Guyana. Additionally, many global-scale estimates of mangrove areas over time were incorporated to provide a range of country-level area estimates and associated carbon stocks. In Guyana, country-level mangrove C stock estimates range widely from 3.11 to 8.15 Tg C and from 11.6 to 19.8 Tg C in Suriname.

Key components needed to assess the feasibility of a carbon finance project are a carbon project methodology, a carbon market (currently in the NBS-LME, this would be a voluntary market), landscape feasibility, assessment of additionality, greenhouse gas accounting, financial feasibility, legal feasibility, organizational feasibility, and a permanence assessment. The latter is integral for assessing project longevity and feasibility. There is great potential for carbon financing projects in Suriname and Guyana for both restoration and conservation. Project implementation, validation, and monitoring costs are estimated at $350,000 USD, and, considering this, most scenarios presented for the NBS-LME result in revenue. A clear plan including organizational feasibility and structure combined with community engagement and involvement are integral in creating a successful carbon project.

2 Introduction

2.1 Project Background

The project entitled "Setting the foundations for zero net loss of the mangroves that underpin human wellbeing in the North Brazil Shelf LME (NBS-LME)" (from here on the "NBS Mangrove Project"), is a one-year primer project to help establish a shared and multi-national process for Integrated Coastal Management (ICM). The project recognizes the prevalence, socio-ecological importance and connectivity of mangroves in the retention and generation of key ecosystem services (fisheries, coastal protection and defense, water quality, blue carbon, etc.) from which communities in the NBS countries are beneficiaries. This project builds on, and supports, the antecedents and key elements of the regional agreement established within the CLME+ SAP for the NBS region.

The objectives of the NBS Mangrove Project are:

1. To generate the necessary baseline knowledge and technical assessments as inputs towards a collaborative vision and a coordinated well-informed management of NBS mangrove systems, with emphasis on the information needs of Guyana and Suriname.
2. To support development of transboundary coordination mechanism(s) between the countries of Guyana, Suriname, French Guiana, and Brazil (state of Amapá) towards the improved integrated coastal management of the extensive, ecologically-connected yet vulnerable mangrove habitat of the NBS region.

2.2 Report Objectives

The objective of this report is to evaluate the potential of Guyana and Suriname's mangrove ecosystems to contribute to climate change mitigation by exploiting their ability to sequester carbon and their role as important national carbon sinks. Specific objectives are to provide:

1. A review of NBS mangrove ecological structure, function, and key environmental factors regarding carbon sequestration and storage potential;
2. dimensioning NBS mangrove potential as carbon sinks; and
3. dimensioning NBS mangrove carbon value.

3 Mangrove Ecological Structure

3.1 Mangrove Species Assemblages

The coastlines of Suriname and Guyana extend 386 and 459 km, respectively, and were historically lined in fringing mangroves upwards of 4 km wide. Compared to the Indo-Pacific, the mangroves of the NBS-

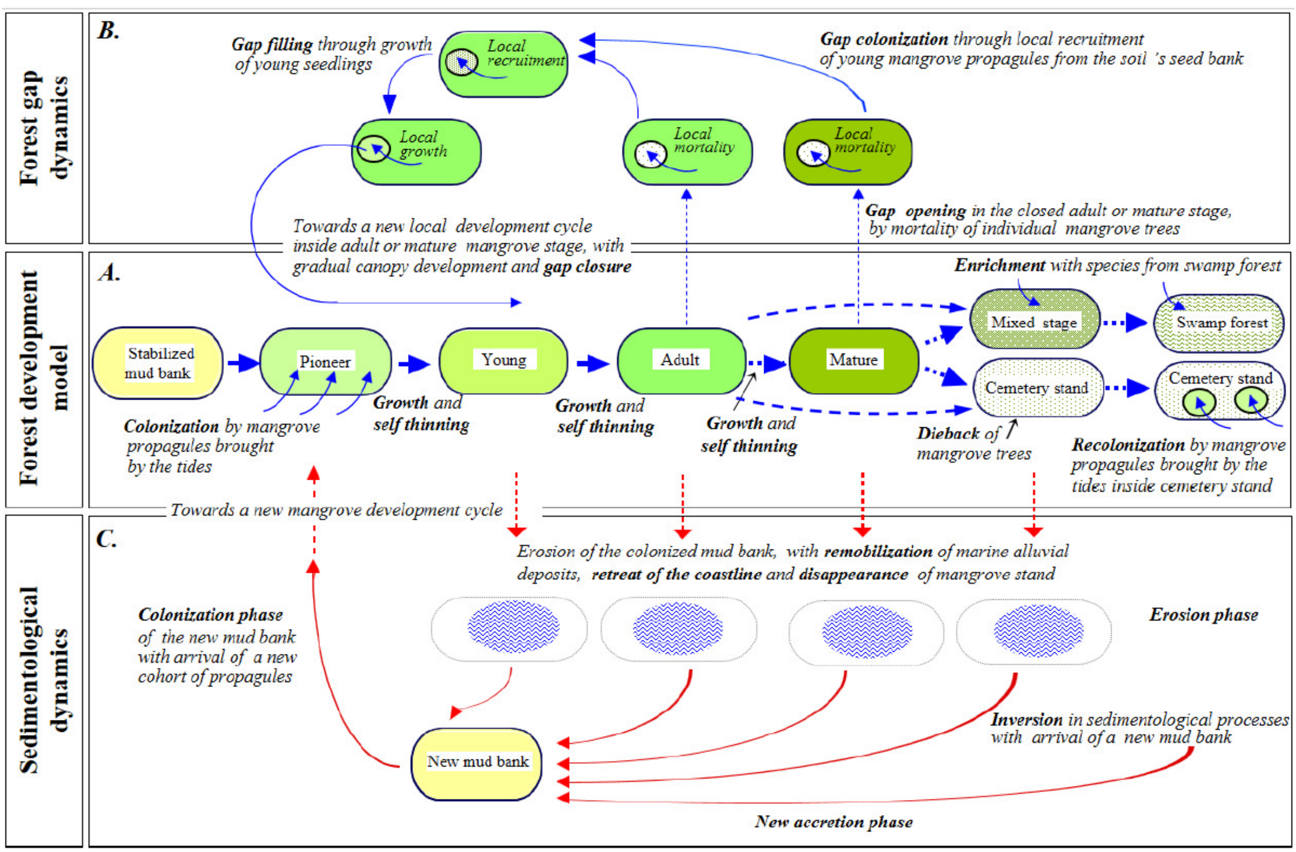
LME, and the Caribbean in general, are low in species diversity yet they still serve the same important ecosystem functions. Mangrove systems along the NBS-LME are comprised of three genera: *Avicennia germinans* (also accounts of *A. schaueriana* in Suriname; black mangrove, parwa, courida), *Rhizophora mangle* (also accounts of *R. racemosa* and *R. harrisonii* (hybrid); red mangrove, mango, red mango), and *Laguncularia racemosa* (white mangrove, akira). Black mangroves are the dominant species along the North Atlantic Ocean and have the potential to withstand high soil salinities, upwards of 60 PSU (Marchand et al. 2004). Growth is very rapid once seedlings establish and trees can reach 30 m in height once mature. Red mangroves are more abundant inshore of coasts in riverine zones and along the edges of swamp forests, where fluctuations in water salinity are common. These species can also reach heights upwards of 30 m. White mangroves, reaching heights of 6 m, are not as common as black mangroves but can colonize along with them on the ocean-front and are most often found at the landward edge of mangrove stands in areas that are inundated mainly by spring tides. At the landward side of the mangrove band, marsh forests occur, composed of *Symphonia globulifera*, *Ficus* spp., *Virola surinamensis*, *and Euterpe oleracea*, which can be mixed with old *A. germinans* stands (Lee et al. 2014).

3.1.1 Stand Characterization and Propagation

There are four main mangrove stand characterizations along the NBS coast: (1) pioneer and young mangroves, (2) mature coastal/pure mangroves, (3) mature riverine/mixed mangroves, and (4) declining or dead mangroves (Figure 1; Fromard et al. 1998). Prolific black mangrove propagule production occurs January through April (Proisy et al. 2009). Through elevation surveys in French Guiana, the minimal ideal elevation for mangrove propagule colonization is approximately 2.45 m above the local datum; the sediment is sufficiently exposed and dewatered during low tides such that cracks are formed (Gensac et al. 2011). The propagules colonize bare mud, mostly in these cracks (Fromard et al. 1998, Proisy et al. 2009). Black mangrove seedlings establish roots that grow horizontally, avoiding the more anoxic deeper soils, and growth is rapid once established (Toorman et al. 2018). Propagules can grow to 15 cm in height within 20 to 30 days of establishing and high densities of small trees can be reached (up to 30,000 ha^{-1}; Table 1; Figure 2; Fromard et al. 2004). White and black mangroves are equally likely to colonize mudflats; however, over time, black mangroves outcompete white mangroves due to their higher growth rate and overall taller stature (Fromard et al. 2004). Young black mangrove stands have an average height of five to six meters and densities below 10,000 tree ha^{-1} (Fromard et al. 2004). Black mangroves also dominate mature stands with roughly 1,000 trees ha^{-1}. The width of a mangrove stand from the landward edge can vary from less than 1 km to nearly 4 km wide, which is likely influenced by river mouth proximity (Plaziat and Augustinus 2004). As stands age, senescent ('cemetery') mangrove stands begin to form with patches of dead mangroves and the formation of gaps with younger trees growing within them (Fromard et al. 1998). It is not uncommon to see a mixture of young, dying and dead mangroves in the same area, a unique feature of NBS-LME mangroves. As off-shore mudbanks migrate west and exposure to wind/wave energy increases, soil below the mature mangroves begins to erode and trees become uprooted. The amount of erosion can be small or can encompass the entire stand. Pneumatophores can also be buried in mud, essentially suffocating the trees (Fromard et al. 1998, Fromard et al. 2004).

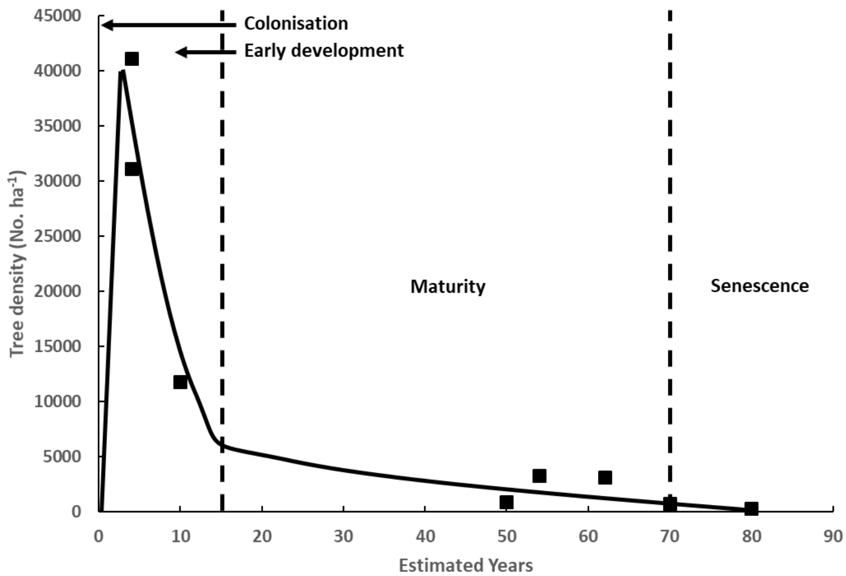
3.1.2 Degradation

This highly dynamic system formed by the ever-changing location of mudbanks creates a coastal mosaic with the presence and absence of mangroves. This complex mangrove-mudbank dynamic has functioned in this manner for 5-6,000 years, and the mangrove distributions are inextricably tied to the location and movement of mudbanks along the coast (Anthony 2015). However, human removal and degradation of mangroves has already affected this dynamic system, and a general lack of understanding on how this system works could significantly affect its protection (Anthony 2015). In French Guiana, the conversion of mangroves to rice fields has resulted in shoreline retreat of up to 180 m per year in some areas (Anthony 2015). Continued conversion and degradation could disrupt the unique mangrove-mudbank relationship, resulting in far fewer mangrove systems along the coastline over time (Anthony 2015).



Source: Fromard et al. 2004

Figure 1. Mangrove dynamics along the North Brazil Shelf.



Source: modified from Fromard et al. 1998

Figure 2. Changes in tree density during mangrove stand development.

Table 1. Mangrove structure from stands of different ages in French Guiana.

Stage	Density (trees ha^{-1})	Mean height (m)	Surface area (m^2 ha^{-1})	Mean diameter (cm)	Aerial biomass (Mg ha^{-1})
Pioneer	8,400	2.5 – 5	4 – 21	2.3 – 2.7	11.4 – 56.6
Young	31,100	5 – 6.1	4 – 21.4	4.3 – 4.8	14.6 – 73.1
Adult	2,400 – 9,200	18.2 – 22	22.5 – 26.8	23.6 – 44.9	180 – 228.8
Mature	450 – 917	24.8	51.4	67.1	431.9
Mixed	3,047	19	17.8	21.7	122.2
Cemetery	267 – 825	15 – 17	13.8 – 18.5	28.5 – 31.1	77.6 – 110

Source: Fromard et al. 1998

3.2 Mangrove Ecological Function and Services

Mangrove ecosystem services can take many forms, including supporting (nutrient cycling, carbon storage, primary production), provisioning (wood, fuel, food), regulating (climate and flood regulation, water purification), and cultural (recreation, spiritual, aesthetic, education) services (Vo et al. 2012, Lee et al. 2014). Lee et al. (2014) further elucidate these ecosystem services, discussing current knowledge, data gaps and future directions for (1) the dynamics of carbon fixation, storage and mineralization, (2) their nursery function, (3) shoreline protection, and (4) their land-building capacity.

3.2.1 Blue Carbon – Sequestration and Release

Over the past 15 years, the crucial role of coastal ecosystems, specifically mangroves, tidal marshes, and seagrass beds, in sequestering significant amounts of carbon, termed 'blue carbon' due to its coastal influence, has been clearly demonstrated (Donato et al. 2011, McLeod et al. 2011, Sifleet et al. 2011, Fourqurean et al. 2012, Pendleton et al. 2012, Windham-Myers et al. 2018). This is largely due to the extremely slow decomposition and mineralization rates of organic matter produced by wetland plants that occur under conditions created with inundated, anoxic soils. As soon as these soils are exposed to oxygen through diking and draining of wetlands, mineralization occurs quickly, and the stored carbon is released rapidly to the atmosphere. This impact is particularly notable in mangroves that have been converted to shrimp ponds or cattle pastures. Kauffman et al. (2017) reported that 54% of soil carbon pools can be lost with conversion to shrimp ponds and that total ecosystem C stocks can decline by 554 ± 230 Mg C ha^{-1}, a staggering number considering the short duration of land use from these activities.

3.2.2 Nursery and Habitat Function

When inundated, mangroves provide refuge and nurseries for many fish and invertebrate species as part of a mosaic of adjoining habitats. Predation is lower due to more turbid conditions and reduced visibility, food is more abundant, and habitat complexity from pneumatophores and prop roots is beneficial (Beck et al. 2001). Bird surveys conducted in Guyana show a rich assemblage that varies between sites; diversity and bird counts are greater within dense mature mangrove stands, with a third of species being migratory (Da Silva 2015b). Species can use mudflats and mangrove habitat for forage; the later also providing breeding habitat for many local and migratory species.

3.2.3 Shoreline Protection and Accretion

Shoreline protection by mangroves is largely attributed to persistent low level wave energy dissipation, which can be variable depending on the location of settlements in relation to mangroves (next to versus behind) and forest structure (width and degree of human degradation; type of mangrove species present) (Lee et al. 2014). More fragmented and degraded mangroves are less likely to provide shoreline protection. The ability for sediment to be trapped by mangroves and for accretion to occur relative to sea level rise is significantly related to local hydrology and erosion regimes, the health of the stand, and the species composition (settling rates can be different for trees with prop roots versus pneumatophores).

Mangroves play a significant role in the stabilization of sediment during periods of progradation at the leading edge of a mud bank. This helps to provide a continuous mangrove band with older mangroves serving as a local source of propagules for colonization of consolidated mud banks (Anthony 2015). Individual mudbanks are separated by wide interbank zones, with the spacing of the individual mudbanks controlled by a ~30-year cyclicity in trade wind intensities. These create a discontinuous sediment supply (Allison et al. 2000, Gratiot et al. 2008). When the coastline is protected from wave attack by a mudbank, mangroves rapidly colonize and stabilize the inboard mudflats. However, when the mudbank migrates, the erosive interbank stage can result in erosion of the sub-mature mangrove stands at rates of up to 40 m per year (Allison and Lee 2004, Anthony 2015). This rarely results in the total eradication of established mangroves; however, total eradication can occur in periods of very high wave energy such as in El Niño years or phasing of the NAO (Anthony 2015). Eradication also can happen in mangrove areas disturbed or deforested by humans, such as in an area formerly occupied by rice fields in French Guiana. In this case, the shoreline retreated at rates up to 180 m a year and threatens to disrupt the positive feedback loop between mangroves and mud banks (Anthony 2015). Actual mangrove removal has occurred in Suriname, in particular near Paramaribo, where discussion of dyke building has surmounted mangrove preservation in regard to protecting the coast. From experience in Guyana, this action of removing mangroves in favor of hardened walls has resulted in significant loss of coastal protection, providing important lessons to be heeded (Anthony 2015).

There are multiple personal and small-scale commercial uses of mangroves: honey and crab harvesting, production of poles for fishing and fencing, fuel wood, and bark used for tanning (Allan et al. 2002, Da Silva 2015a). Mangroves are also commonly used for grazing livestock, which can have negative consequences particularly in pioneer stands (Da Silva 2015a).

3.3 Environmental Factors Structuring Mangroves

Coastal environments can be some of the harshest globally due to regular tidal inundation and high salinities, which challenge the growth and survival of most plant species. Mangroves have specific morphological and physiological characteristics that enable them to persist in this inundated environment; however, there are limitations that can result in reduced growth and even mortality.

3.3.1 Salinity

Mangroves are facultative halophytes, meaning that they can tolerate high salinity but do not require salt for growth. They can prevent salt from entering the plant at the root interface (*Rhizophora* and *Laguncularia*) and can excrete salt from the leaves or store the salt within the leaves, which senesce and fall from the tree (*Avicennia* and *Laguncularia*).

Marchand et al. (2004) conducted a comprehensive study on mangrove soil pore-water chemistry in French Guiana, under conditions comparable to Suriname and Guyana. Surface salinities were largely affected by seasonal changes and proximity to freshwater sources. Soil salinity can range from 5-20 PSU at the soil surface, increasing to upwards of 60 PSU below a depth of 20 cm (Marchand et al. 2004). They also found that soil salinity did not differ across mangrove species; therefore, salinity does not appear to be a driving factor in species zonation. *Rhizophora* species are more commonly found in areas exposed to more freshwater, thus likely have a higher tolerance to variable salinity (Marchand et al. 2004).

There are limits to mangrove growth under saline conditions, despite the high tolerance by *Avicennia*. Saline waters and soil conditions are often improved by freshwater flows from the lowland coastal swamps and rivers. If this flow is interrupted, hypersaline conditions can be created that result in widespread mangrove death, as seen in Suriname near Coronie (Toorman et al. 2018) and in Region 5 in Guyana (Evans 1998).

3.3.2 Inundation

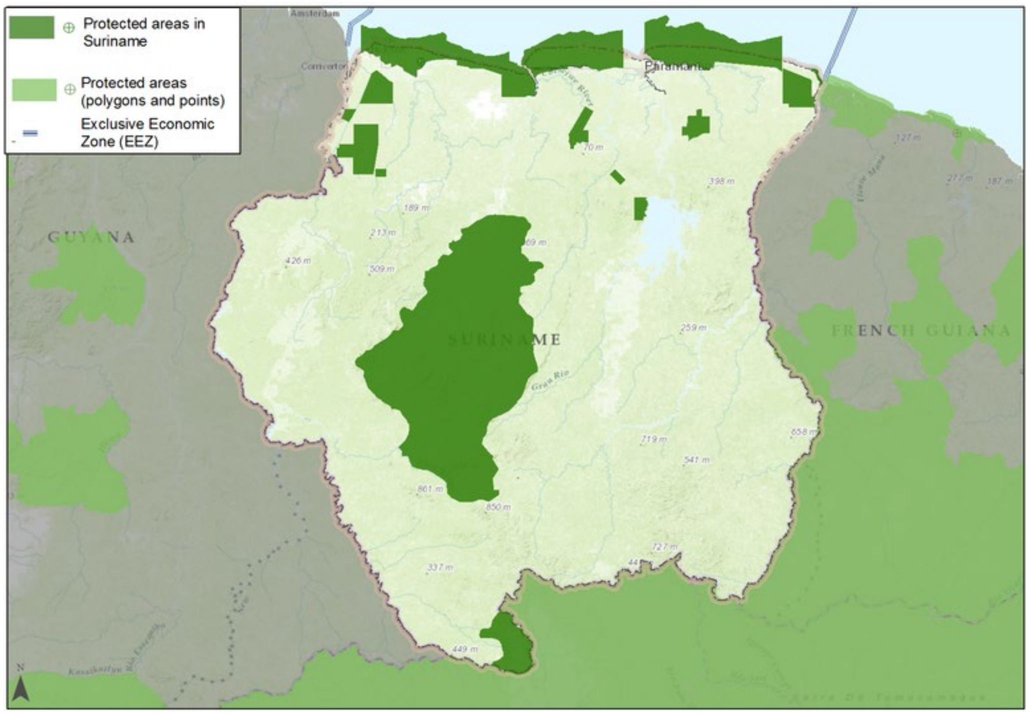
Tides along the NBS are semidiurnal with a mean tidal range of 1.8 m and a maximum tidal range of 2.5 m (Rine 1980). Along the Guianas, black mangroves grow between mean tide level and mean high water level (Proisy et al. 2008). This is comparable for Guyana and Suriname. Tidal waters provide abundant sediment that settles during inundation. Along the coast of Suriname, net accretion has occurred since 1966, likely due to increased sediment transport from the Amazon River (Martinez et al. 2009, Toorman et al. 2018). The interplay between sea level rise and increases in suspended sediment from the Amazon river could even result in neutral sea level rise impacts (Toorman et al. 2018) as long as sea level rise does not outpace sediment accretion.

Mangroves have adapted to growth under inundated and anoxic soil conditions. Black mangroves produce abundant pneumatophores, which are areal roots that extend 20 to 30 cm from the soil surface and are covered in lenticels (small pores) that serve to increase oxygenation of the root system. White mangroves also produce pneumatophores. Red mangroves are notable for their highly branching prop roots that elevate the main stem above tidal waters and also contain numerous lenticels. Another mangrove adaptation to alleviate inundation stress is the production of viviparous and crypto-viviparous seedlings, meaning that seeds germinate while still on the tree and can float in tidal waters and establish quickly once settled.

Mangroves in the NBS-LME are reliant upon sufficient sediment supply to raise mudbank elevations to a point where propagules can colonize, and mature trees can maintain elevation relative to sea levels. This is opposed to many island and carbonate systems, where mangroves are dependent solely on autochthonous soil production to keep pace with sea level rise (McKee et al. 2007). Currently, there appears to be no shortage of sediment along the NBS, so a combination of mineral and organic accretion will likely maintain adequate mangrove surface elevations for now. Additionally, tidal wetland and mangrove systems can migrate upland as sea levels rise, permitting that there is area available for this transition. In many areas of Guyana where there are hardened shoreline structures, there will be no room for mangroves to migrate and many stands will likely drown due to increased wave action and an inability to keep pace with faster rates of sea level rise.

3.4 Mangrove Protected Areas

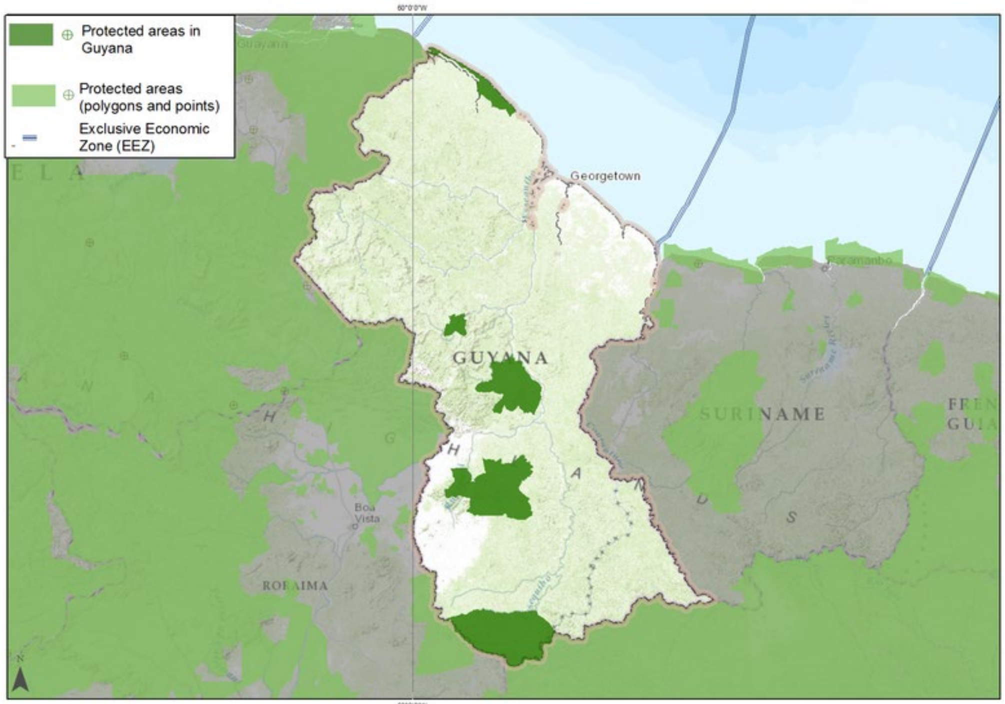
The majority of the coast of Suriname is protected through four Multiple Use Management Areas (MUMAs) and six Nature Reserves, totaling approximately 128,000 ha; only the area around Paramaribo and the eastern stretch of coast are not protected (Figure 3; UNDP 2016). There currently are no marine protected areas. While there is institutional knowledge about the importance and role of mangroves in coastal processes along the NBS-LME and capacity is in place amongst agencies, mangrove management has not received much attention from the government, which has complicated institutional arrangements, and no programs for monitoring and management are currently in place (UNDP 2016).



Source: www.biodiversitya-z.org

Figure 3. Map of protected areas and exclusive economic zones in Suriname.

There is currently only one coastal protected area along the coast of Guyana: the Shell Beach Protected Area in the Barima-Waini region (Figure 4); the largest and most intact mangrove areas occur here. In 2001, the Guyana National Mangrove Management Action Plan was formed to address legislation regarding mangroves, highlighting the need within government agencies with administrative capacity to create a legal framework for mangrove management and promote sustainable management, to support and manage research in mangroves, to effectively create and implement mangrove restoration, rehabilitation and protection, and to continue public outreach and education about mangroves (Evans 1998). There has been mixed public perception regarding the roles of mangroves as sea defense, although outreach campaigns have proven effective in increasing community knowledge about mangroves (Allan et al. 2002, Da Silva 2015a). This is due in large part to the Guyana Mangrove Restoration Project, which was initiated in 2010. The Project's objectives were to create administrative capacity and legal frameworks for mangrove management, foster sustainable mangrove management with community involvement, conduct public outreach campaigns, and support and develop mangrove research, restoration, and protection (Da Silva 2015a).



Source: www.biodiversitya-z.org

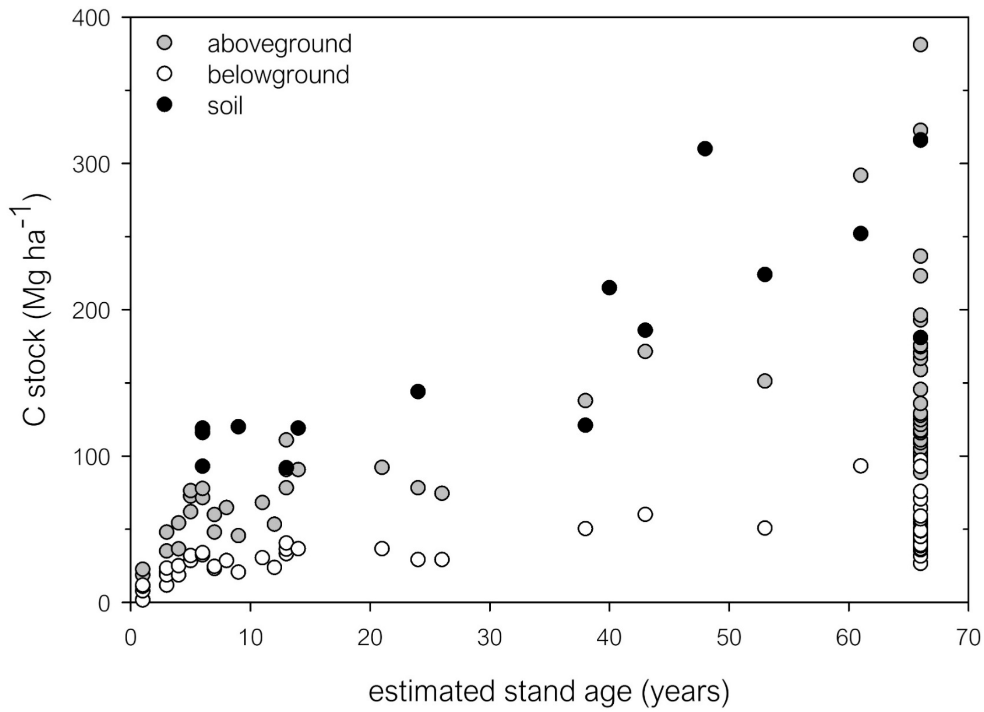
Figure 4. Map of protected areas and Exclusive Economic Zones in Guyana.

4 Mangrove Stock Quantification

Mangrove ecosystem carbon stocks are comprised of above- and belowground tree, dead tree, downed wood, and soil components. If a carbon project is being considered, methane (CH_4) and nitrous oxide (N_2O) emissions also must be factored in. Each stock component is addressed below, including stock values and sources.

4.1 Tree Stocks

Mangrove biomass and carbon stock data are derived from measurements in French Guiana, which is the nearest country to Guyana and Suriname in the NBS-LME where a rich history of mangrove research has occurred (Figure 5; Table 2; Marchand 2017, Walcker et al. 2018). Mangroves of various stand ages were surveyed by Walcker et al. (2018) and estimates of above- and belowground biomass were made using region-specific allometric equations. No data was collected regarding standing dead and downed wood carbon stocks. Tree biomass was converted to tree carbon using conversion factors of 0.48 and 0.39 for above- and belowground biomass, respectively (Kauffman and Donato 2012). Based on area estimates for mangrove stands of different age classes in French Guiana, it was assumed that 25% of mangrove forests are pioneer (<10 years old), 26% of forests are young (10 to 20 years old), and 48% of forests are mature / senescent (>20 years old). The stand distribution could be different from French Guiana but incorporating variability in stand age in the dynamic NBS-LME system will allow for a more accurate assessment than assuming a uniform stand age.



Source: Walcker et al. 2018

Note: values at year 66 represent a minimum age of at least 66 years and are likely older.

Figure 5. Above- and belowground mangrove tree and soil carbon stocks in stands of varying ages in French Guiana.

Table 2. Mangrove biomass and stocks and soil carbon stocks from mangrove stands of different ages in French Guiana, with estimated percentage of area for each.

Stand Type	AGB ($Mg\ ha^{-1}$)	BGB ($Mg\ ha^{-1}$)	AGB C ($Mg\ C\ ha^{-1}$)	BGB C ($Mg\ C\ ha^{-1}$)	Soil C ($Mg\ C\ ha^{-1}$)	% area
mature forest (> 20 yrs)	332.00	129.94	159.36	50.68	204.13	48
young forest (10 – 20 yrs)	170.83	85.83	82.00	33.48	105.50	26
pioneer forest (< 10 yrs)	95.78	54.83	45.97	21.39	112.00	25

Source: Walcker et al. 2018

4.2 Soil Carbon Stocks

Soil carbon to 1 m in depth was quantified in mangroves of various stand ages in French Guiana by Walcker et al. (2018) in a portion of stands where tree measurements were made, and additional soil data were incorporated from Marchand (2017; Table 2). Marchand (2017) also examined the soils under mangrove stands of differing ages in French Guiana and determined that the depth of mangrove-influenced soil (the pedogenetic layer) increased with stand age and that the oldest senescent stands had mangrove-influenced soil that was nearly 50 cm deep (Table 3). Below that, organic carbon content in the mud bank sediment was never greater than 1% and was comparable to that from shoreface sediment (Marchand 2017). Marchand (2017) also quantified soil carbon accumulation rates based on stands of different ages (Table 4). Soil carbon data specific to Guyana and Suriname were not located; therefore, the data from French Guiana were used in this analysis.

Table 3. Mangrove biomass and stocks and soil carbon stocks from mangrove stands of different ages in French Guiana, with estimated percentage of area for each.

Mangrove stage	Age (yr)	Pedogenic layer thickness (cm)	C stock in pedogenic layer ($Mg\ C\ ha^{-1}$)
Pioneer	3 ± 1	6 ± 2	4.8 ± 1.3
Young	6 ± 2	15 ± 2	12.5 ± 2.3
Young mature	9 ± 1	25 ± 2	20.4 ± 8.4
Mixed mature	40 ± 7	40 ± 2	68.6 ± 31.3
Senescent	48 ± 2	45 ± 5	107.5 ± 23.2

Source: Table 1 from Marchand 2017

Table 4. Soil accretion and carbon burial rates for mangrove stands of varying ages in French Guiana.

Mangrove stage	Accretion rate ($cm\ yr^{-1}$)	Carbon burial rate ($Mg\ C\ ha^{-1}\ yr^{-1}$)
Pioneer	2	0.72
Young	2.5	2.54
Young mature	2.8	4.8
Mixed mature	1.0	1.35
Senescent	0.9	4.86

Source: Table 2 from Marchand 2017

4.3 Methane Emissions

Methane (CH_4) emissions within coastal mangroves are naturally occurring and vary based on salinity. Mangroves that experience salinity greater than 20 PSU are very likely to have insignificant CH_4 emissions (Poffenbarger et al. 2011); however, emissions cannot be considered negligible according to the Verified Carbon Standard (VCS) unless field data is collected that demonstrate otherwise (Needelman et al. 2018). Without local data, the default value for tidal systems with salinities greater than 20 PSU of 0.0056 Mg CH_4 ha^{-1} yr^{-1} or 0.14 Mg CO_2e ha^{-1} yr^{-1} can be used (Needelman et al. 2018). A global warming potential of 25 was used to convert CH_4 to CO_2e, which is the conversion used in the IPCC Fourth Assessment that is currently adopted within the VCS. There are a variety of methods for calculating global warming potential.

4.4 Nitrous Oxide Emissions

Nitrous oxide (N_2O) is a naturally occurring trace greenhouse gas that has a significantly greater contribution to atmospheric warming than CO_2, on the order of 298 times greater (IPCC 2007). As with CH_4, N_2O can vary seasonally but is also influenced regionally by human activity via runoff from agriculture and sewage. To date, no N_2O measurements have been made in mangroves along the NBS; however, emissions from Ciénaga Grande de Santa Marta, Colombia, the closest estuary to the NBS-LME where data is available, ranged from 0.013 to 0.014 Mg N_2O ha^{-1} yr^{-1} (3.87 to 4.17 Mg CO_2e ha^{-1} yr^{-1}) in the dry and rainy seasons, respectively (Konnerup et al. 2014).

4.5 Mangrove Area Estimates

Mangrove area estimates came from several sources, all of which were determined using remote sensing techniques on a global basis. One of the most widely-used datasets of global mangrove coverage is from Giri et al. (2011) who classified mangrove distributions for the year 2000 using Landsat imagery (resolution of 30 m). The World Atlas of Mangroves data also represent mangrove distributions from 2000 (Spalding et al. 2010). The Global Mangrove Watch distributions were estimated with an accuracy of 94% from classification of ALOS PALSAR and Landsat sensor data, and were informed by the Giri et al. (2011) and Spalding et al. (2010) distributions (Bunting et al. 2018). They calculated global mangrove distributions for 1996, 2007-2010, 2015, and 2016 (Global Mangrove Watch, unpublished data, used with permission). Hamilton and Casey (2016) used two initial datasets to estimate mangrove distributions from 2000 through 2012 that incorporated the area within a Landsat pixel that is covered by mangroves instead of just presence/absence. They also projected mangrove areas for 2013 and 2014. In this report, only the area estimates based on the Giri et al. (2011) dataset were used since the other dataset using the Terrestrial Forest of the World appears to overestimate mangrove coverage.

For each of the estimated mangrove areas and years described above, country-level carbon stocks were calculated for mangrove trees, including above- and belowground stocks, and soil to one meter (Table 2 above) to assess variability over time and methodology. Using the percentage of stand age areas listed in Table 2, tree carbon stocks were scaled accordingly for Guyana and Suriname.

Additionally, total mangrove carbon stock data from Hamilton and Friess (2018) was incorporated. Hamilton and Friess (2018) calculated total mangrove and soil carbon stocks globally by using five different models and the 2012 mangrove coverage estimated in Hamilton and Casey (2016). The average total carbon stock as well as minimum and maximum values were presented for Guyana and Suriname; however, the pieces of data were not presented by individual carbon pools (e.g. above- and belowground tree and soil C stocks) or by each of the five models.

4.6 Mangrove Deforestation Calculations

Because of the progradational and erosional mangrove dynamics in the NBS-LME, teasing apart natural from man-made loss can be difficult from the available global mangrove datasets. Country or region-specific data and insight are needed. In Guyana, mangrove degradation data points were assembled by the Guyana Forest Commission during seven time periods between 1990 and 2014 (Guyana Forest Commission, data used by permission). Deforestation spatial data for Suriname was provided by the National Land Monitoring System of Suriname between 2000 and 2017 over six time periods (GONINI 2019). Mangrove deforestation was calculated by taking the deforestation spatial data for each time period and intersecting that with the most relevant mangrove coverage data for that time using ArcGIS

Pro 2.3.2 (Spalding et al. 2010, Giri et al. 2011, Bunting et al. 2018). The total area of mangroves deforested across the country was then collated.

Much of the mangrove deforestation that has occurred in Guyana and Suriname happened in conjunction with the development and growth of Georgetown and Paramaribo, respectively, largely before coverage estimates were known. As stated earlier, most of the coast of Suriname is protected in some form; therefore, mangrove deforestation in theory should be minimal if protection measures are enforced. Future development plans, and subsequent deforestation, in Suriname suggest that the majority will occur in freshwater marsh forests: 1,500 ha was converted to sugar cane as part of a bio-fuels initiative and rice paddy formation is expected to increase another 30,000 ha (FAO 2015).

4.7 Results

Like other mangrove systems globally, those in the NBS-LME store significant amounts of carbon within the trees and soils. Approximately the same amount of carbon is stored above- and belowground portions of mature mangrove stands as in the soil (Tables 5 – 10). Due to the very rapid growth of black mangroves in the NBS-LME, trees are the dominant carbon stock during pioneer and young mangrove stages until a more mature forest develops and enough time has passed for mangroves to build organic soils (Tables 3 & 4). Since mangrove and soil stock data from French Guiana were used to estimate stocks in Suriname and Guyana, instead of country-specific data, it is currently unknown if stocks on a per area basis differ between countries.

Compared to mangrove systems within the North Atlantic Ocean and Caribbean, carbon stocks along the NBS are comparable. Kauffman et al. (2018a,b) measured mangrove carbon stocks near the mouth of the Amazon river and in the state of Ceará and documented total tree carbon stocks between 72 and 156 Mg C ha^{-1} and soil stocks between 111 and 162 Mg C ha^{-1} (note that soil cores were taken to a maximum of 3 m). Numerous soil cores were taken in the Morrocoy National Park in Venezuela; however, they were shallow (20 cm) and thus not comparable (Barreto et al. 2016). Tree stocks were smaller in the Dominican Republic (76.3 Mg C ha^{-1}), Panama (55.3 Mg C ha^{-1}) and Honduras (103.5 Mg C ha^{-1}); however, soil stocks were significantly greater: 753.1, 711.0 and 925.1 Mg C ha^{-1}, respectively (soil cores were all greater than 1 m; Kauffman et al. 2014, Schile-Beers et al. unpublished data, Bhomia et al. 2016).

Each of the methods used to quantify mangrove area in Guyana and Suriname produced different estimates, even when comparing values within the same year. Overall, however, more mangrove habitat is found in Suriname than Guyana, approximately three times as much. Within Guyana, mangrove area estimates from Global Mangrove Watch (2018) and Hamilton and Casey (2016) show slight decreases in area over time (Tables 6 & 7). Data collected by the Guyana Forestry Commission also supports a small but consistent loss of mangroves since 1990 (Figure 6). Changes in mangrove area over time for

Suriname are more variable across data sources. Global Mangrove Watch estimates show alternating gains and losses across years with a significant decrease in area between 2015 and 2016, especially notable between Totness and just west of Paramaribo (Table 9). The loss estimates from Hamilton and Casey (2016) are more consistent within Guyana in that small decreases in mangrove coverage have occurred between 2000 and 2014, all less than 0.5% between years (Table 10). Local deforestation data in Suriname shows that mangrove loss has decreased since 2000 but the area is greater than in Guyana (Figure 7). More detailed examination of aerial imagery needs to occur to assess and verify the extent and location of mangrove loss in Suriname and compare against mud bank locations to determine if natural or human-based drivers are at play.

Table 5. Carbon stock data for Guyana using mangrove area estimates from Global Mangrove Watch (2018).

Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)	C stock change from prior year (%)
1996	27,983	6,428,715	2,778,484	3,085,783	1,083,609	4,334,503	8,503,895	
2007	27,424	6,300,276	2,722,973	3,024,132	1,061,959	4,247,904	8,333,996	-2.00
2008	27,284	6,268,122	2,709,076	3,008,698	1,056,539	4,226,224	8,291,462	-0.51
2009	27,202	6,249,324	2,700,951	2,999,675	1,053,371	4,213,550	8,266,596	-0.30
2010	27,393	6,293,226	2,719,926	3,020,749	1,060,771	4,243,151	8,324,671	0.70
2015	26,739	6,142,856	2,654,936	2,948,571	1,035,425	4,141,765	8,125,761	-2.39
2016	26,836	6,165,209	2,664,597	2,959,300	1,039,193	4,156,837	8,155,330	0.36

Table 6. Carbon stock data for Guyana in 2000 using mangrove area estimates from the World Atlas of Mangroves (Spalding et al. 2010) and Mangrove Forests of the World (Giri et al. 2010).

Source	Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)
Spalding et al.	2000	37,556	8,627,983	3,729,005	4,141,432	1,454,312	5,817,339	11,413,083
Giri et al.	2000	20,072	4,611,270	1,992,986	2,213,410	777,265	3,109,107	6,099,782

Table 7. Carbon stock data for Guyana using mangrove area estimates from Hamilton and Casey (2016).

Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)	C stock change from prior year (%)
2000	18,828	4,325,478	1,869,467	2,076,229	729,092	2,916,415	5,721,736	
2001	18,826	4,325,019	1,869,269	2,076,009	729,015	2,916,105	5,721,128	-0.01
2002	18,824	4,324,559	1,869,070	2,075,788	723,937	2,915,795	5,720,521	-0.01
2003	18,819	4,323,410	1,868,574	2,075,237	728,744	2,915,020	5,719,001	-0.03
2004	18,816	4,322,721	1,868,276	2,074,906	728,627	2,914,556	5,718,089	-0.02
2005	18,812	4,321,802	1,867,878	2,074,465	728,473	2,913,936	5,716,874	-0.02
2006	18,810	4,321,343	1,867,680	2,074,245	728,395	2,913,626	5,716,266	-0.01
2007	18,807	4,320,654	1,867,382	2,073,914	728,279	2,913,162	5,715,354	-0.02
2008	18,800	4,319,045	1,866,687	2,073,142	728,008	2,913,077	5,713,227	-0.04
2009	18,797	4,318,356	1,866,389	2,072,811	727,892	2,911,613	5,712,315	-0.02
2010	18,790	4,316,748	1,865,694	2,072,039	727,621	2,910,528	5,710,188	-0.04
2011	18,780	4,314,451	1,864,701	2,070,936	727,233	2,908,979	5,707,149	-0.05
2012	18,777	4,313,762	1,864,403	2,070,606	727,117	2,908,515	5,706,238	-0.02
2013	18,777	4,313,717	1,864,384	2,070,584	727,110	2,908,485	5,706,179	-0.001
2014	18,773	4,312,739	1,863,961	2,070,115	726,945	2,907,825	5,704,885	-0.02

Table 8. Carbon stock data for Suriname using mangrove area estimates from Global Mangrove Watch (2018).

Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)	C stock change from prior year (%)
1996	73,331	16,846,762	7,281,153	8,086,446	2,839,650	11,358,777	22,284,872	
2007	76,103	17,483,558	7,556,376	8,392,108	2,946,987	11,788,131	23,127,226	3.78
2008	76,258	17,519,223	7,571,790	8,409,227	2,952,998	11,812,178	23,174,403	0.20
2009	76,113	17,485,885	7,557,381	8,393,225	2,947,379	11,789,700	23,130,304	-0.19
2010	77,689	17,847,998	7,713,886	8,567,039	3,008,416	12,033,851	23,609,305	2.07
2015	76,242	17,515,470	7,570,168	8,407,426	2,952,365	11,809,647	23,169,438	-1.86
2016	65,092	14,954,118	6,463,154	7,177,976	2,520,630	10,082,679	19,781,285	-14.62

Table 9. Carbon stock data for Suriname in 2000 using mangrove area estimates from the World Atlas of Mangroves (Spalding et al. 2010) and Mangrove Forests of the World (Giri et al. 2010).

Source	Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)
Spalding et al.	2000	50,727	11,653,747	5,036,737	5,593,799	1,964,328	7,857,434	15,415,560
Giri et al.	2000	74,781	17,179,922	7,425,145	8,246,363	2,895,806	11,583,407	22,725,576

Table 10. Carbon stock data for Suriname using mangrove area estimates from Hamilton and Casey (2016).

Year	Area (ha)	AGB (Mg)	BGB (Mg)	AGB C (Mg C)	BGB C (Mg C)	Soil C (Mg C)	Total C (Mg C)	C stock change from prior year (%)
2000	52,356	5,773,479	2,027,425	8,109,826	5,773,479	2,027,425	15,910,730	
2001	52,323	5,769,840	2,026,147	8,104,714	5,769,840	2,026,147	15,900,701	-0.06
2002	52,271	5,764,106	2,024,133	8,096,659	5,764,106	2,024,133	15,884,899	-0.10
2003	52,211	5,757,490	2,021,810	8,087,365	5,757,490	2,021,810	15,866,665	-0.11
2004	51,973	5,731,245	2,012,593	8,050,500	5,731,245	2,102,593	15,794,338	-0.46
2005	51,873	5,720,217	2,008,721	8,035,010	5,720,217	2,008,721	15,763,948	-0.19
2006	51,812	5,713,491	2,006,359	8,025,561	5,713,491	2,006,359	15,745,411	-0.12
2007	51,684	5,699,376	2,001,402	8,005,734	5,699,376	2,001,402	15,706,512	-0.25
2008	51,544	5,683,937	1,955,981	7,984,049	5,683,937	1,995,981	15,663,967	-0.27
2009	51,518	5,681,070	1,994,974	7,980,021	5,681,070	1,994,974	15,656,066	-0.05
2010	51,490	5,667,983	1,993,890	7,975,684	5,677,983	1,993,890	15,647,557	-0.05
2011	51,401	5,668,168	1,990,443	7,961,898	5,668,168	1,990,443	15,620,510	-0.17
2012	51,201	5,646,114	1,982,699	7,930,919	5,646,114	1,982,699	15,559,731	-0.39
2013	51,136	5,638,895	1,980,164	7,920,779	5,638,895	1,980,164	15,539,837	-0.13
2014	51,038	5,628,116	1,976,379	7,905,638	5,628,116	1,976,379	15,510,133	-0.19

Table 11. Total mangrove carbon stock estimates for Guyana and Suriname from Hamilton and Friess (2018).

Country	Year	Area (ha)	Total C (Mg C)	Min. Total C (Mg C)	Max. Total C (Mg C)
Guyana	2012	18,770	9,905,985	9,773,820	10,038,149
Suriname	2012	51,201	26,827,971	26,385,436	27,270,505

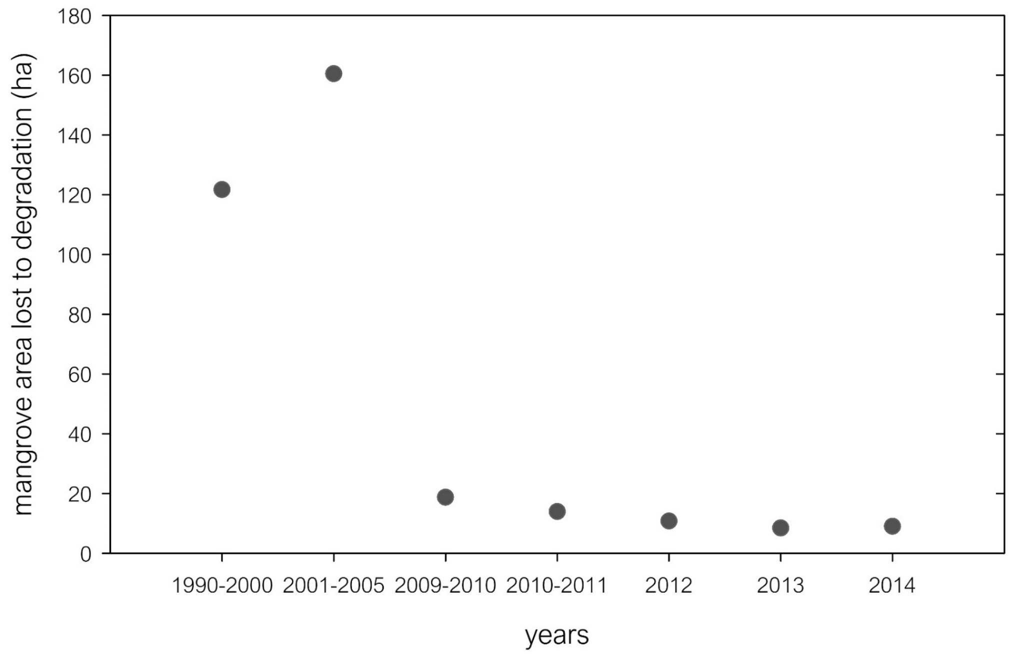
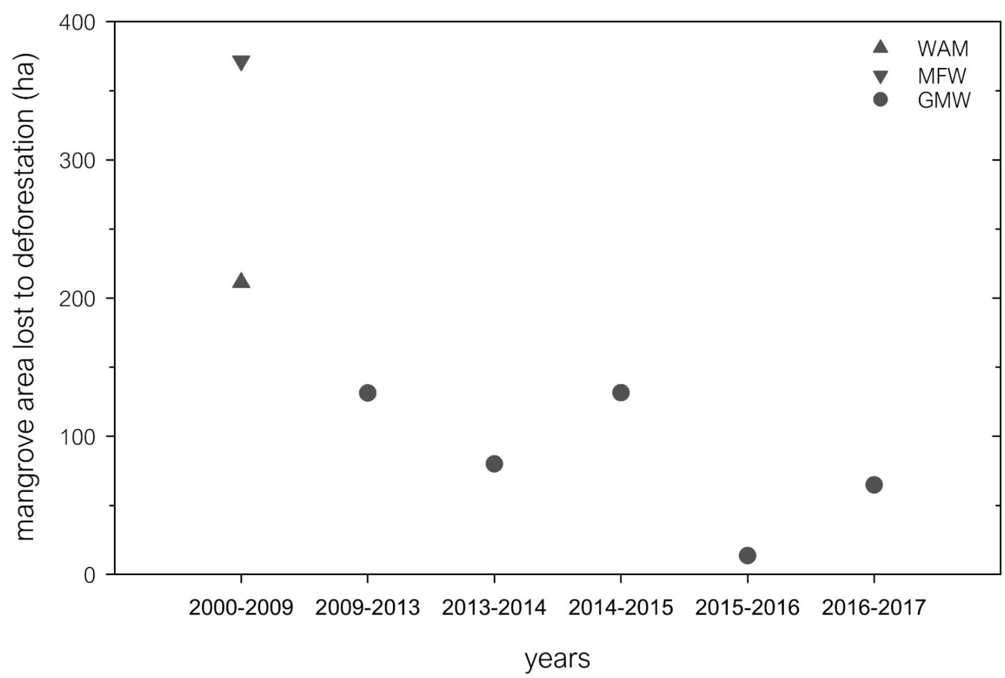


Figure 6. Mangrove deforestation in Guyana.



Note: Years represent the time period that deforestation occurred, and symbols represent the mangrove dataset used (WAM = World Atlas of Mangroves; MFW = Mangrove Forests of the World; GMW = Global Mangrove Watch).

Figure 7. Mangrove deforestation in Suriname.

4.8 Considerations

The unique mangrove system created by the mud banks along the NBS results in soil and tree carbon that is generally only stabilized for 30 to 50 years (Fromard et al. 2004). Within this time, however, significant amounts of carbon can be sequestered within the trees and soil. As described earlier, the data used for creating the stock estimates in Suriname and Guyana were collected in French Guiana. For the most accurate assessment of carbon stocks, local data should be collected, specifically tree characteristics (density, height, diameter at breast height (DBH), canopy width, etc.) in stands of various ages, soil salinity with depth, soil cores to quantify carbon stocks, and ideally measurements of CH_4 and N_2O, both from the soil and from the trees themselves. Protocols have been developed recently that outline the theory, sampling design and methodology, and sample and data analysis for total ecosystem blue carbon stocks (Kauffman and Donato 2012; Howard et al. 2014). In addition to clearly and concisely describing methods, these protocols also allow for uniform sampling across multiple ecosystems and climates, and further fosters the development and integration of the blue carbon scientific community.

In most mangrove systems, coring to 1 m would provide a conservative estimate of soil carbon, since mangrove-influence soils can reach more than 14 m in depth (McKee et al. 2007); however, calculating mangrove soil carbon stocks to 1 m seems appropriate along the NBS due to the mostly ephemeral nature of mangrove systems. In most studies of mangrove soil carbon stock quantification, the methodology by Kauffman and Donato (2012) is used due to its simplicity and rapid nature; soil cores are taken to a maximum of 3 m in depth and 5 cm subsamples are collected at midpoints in pre-defined depth increments (0-15, 15-30, 30-50, 50-100, 100-200, 200-300 cm). This method would not be satisfactory for NBS systems since the mangrove-influenced soil depth is relatively shallow (Table 3; Marchand 2017). Cores would need to be subsampled in smaller increments (e.g. 2 cm) and continuously for the first 30 cm, if not 50 cm, to ensure that the mangrove-influenced soil is properly documented. Although not assessed in this report, soil cores in coastal swamp forests likely would need to be collected to deeper depths to fully understand and quantify this stock, since soil carbon data is minimal if not lacking in Suriname and Guyana. In this case, the methodology from Kauffman and Donato (2012) and Howard et al. (2014) would be appropriate.

5 Carbon Financing Feasibility

5.1 Carbon Project Methodologies

Here, the approaches and requirements for a carbon project are described, although no specific areas or potential projects within Guyana or Suriname are included. In both this report and the NBS Biophysical Assessment Report (Crooks et al. 2019), the potential for a carbon project through landscape feasibility is explored and other elements of a carbon project that will need to be addressed in further carbon project planning are outlined. To proceed with a carbon project, and to assess its value, there is a need to know current and future conditions of the mangroves, especially regarding policy and climate change risks. This report focuses on the VCS methodology; however, another leading approach, Plan Vivo, is also discussed. At the end of this section, key considerations and assumptions are discussed as they relate to the NBS.

5.1.1 Verified Carbon Standard

Under the VCS, the approved approaches for measuring the GHG benefits of a mangrove restoration project are contained in the VCS Tidal Wetlands and Seagrass Restoration Methodology (hereafter referred to as VM0033) and for conservation and restoration in REDD+ Methodology (hereafter referred to as VM0007). The GHG benefits for a potential project are calculated as the sum of the GHG benefits in the biomass carbon pool, the soil organic carbon pool, and deductions for CH_4 and N_2O emissions, as applicable. The GHG benefits are calculated as the difference between emissions in the "baseline" (assuming no restoration occurs) and "with-project" (after restoration occurs) scenarios. Hypothetical baseline scenarios that illustrate net positive impacts of a project are shown in Figure 8.

Applying these methodologies, the baseline scenario biomass stocks, soil stocks, and GHG emissions are measured at the beginning of the project, based on a field inventory. The carbon content is then modelled over the project crediting period using appropriate growth, yield and avoided emissions assumptions. Biomass stocks, soil stocks, and GHG emissions in the with-project scenario are then measured periodically after restoration, also by way of a field inventory. At each monitoring event, changes in biomass stocks in the baseline (assumed to be zero) and in the with-project scenario (based on actual growth) since the last monitoring event are compared to determine the net GHG emission reductions from this pool. Changes in soil organic carbon stocks can be measured using field data, suitable literature values, suitable peer-reviewed models, or default values. Additionally, if deemed appropriate, measurements of CH_4 and/or N_2O would need to be quantified or a proxy used from a neighboring or other applicable region using the methodologies.

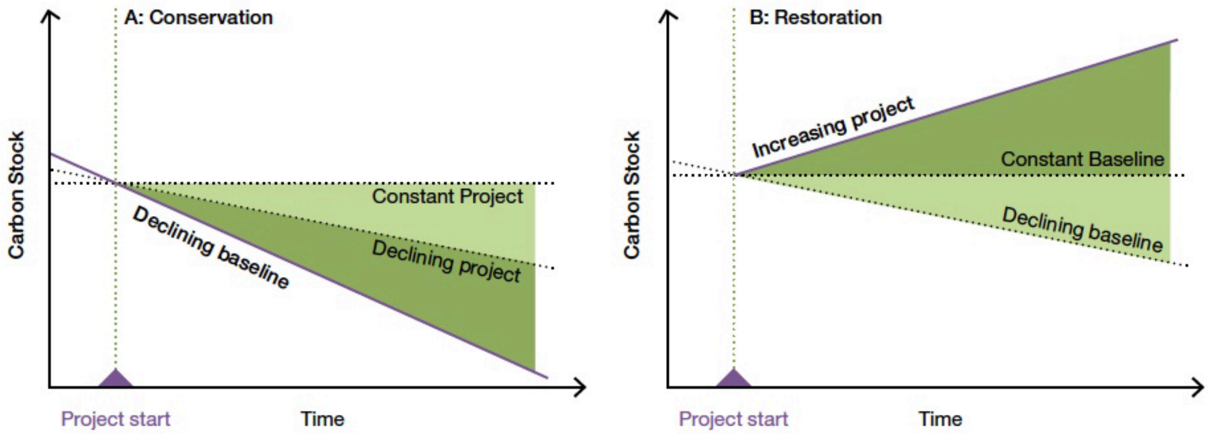


Figure 8. Hypothetical scenarios for net project benefits (green shades) of carbon conservation and restoration projects, based on the differences between the baseline and the project scenario.

5.1.2 Plan Vivo

Plan Vivo is a certification body that administers the Plan Vivo Standard, an international framework for sustainable community land use and forestry projects, that is highly adaptable with broad project scopes and diversity. The Plan Vivo Standard was developed exclusively for use in community-based projects using a "payments for ecosystem services" approach. Projects are certified to be sustainable long-term and to benefit livelihoods and maintain healthy ecosystems. Focus is also placed on fairly-traded and ethical climate services, with more than 60% of funding going towards local communities. By quantifying ecosystem services, demonstrating good governance and measuring performance every year, Plan Vivo fosters results-based payments made on a yearly basis. Plan Vivo Certificates are generated by projects for every 1 metric ton of CO_2e for avoided and/or sequestered CO_2e in addition to other benefits such as water provisioning and climate adaptation, and credits can be sold on the voluntary carbon market. Proceeds are either in cash or in community development such as micro-enterprise development. Projects can comprise one area or many areas, which can expand over time. The Standard is practical and flexible, adapting to specific region's geographic and societal needs and legal frameworks. Examples include 'Mikoko Pamoja', a community-led mangrove conservation and restoration project in Kenya[1], and 'the Nakau Programme', a landscape-level program that uses Payments for Ecosystem Services (PES) for community-based mangrove and rainforest protection in the Pacific Islands[2].

[1] http://www.planvivo.org/project-network/mikoko-pamoja-kenya/

[2] http://www.planvivo.org/project-network/the-nakau-programme/

5.2 Voluntary Markets

As there is currently no compliance market for carbon offset transactions for mangroves and other tidal wetland systems, all interactions are voluntary. In voluntary carbon markets, buyers are consciously and wilfully involved and participating with a desired outcome to reduce GHG emissions. The dominant voluntary market is the VCS although there is also the American Carbon Registry (ACR), the Climate Action Reserve (CAR) in California, United States, and the Markit Environmental Registry, where Plan Vivo Certificates are managed. The voluntary market has been active since the early 2000s and nearly one billion metric tonnes of CO_2e in cumulative volume have been sold globally as of 2016, totalling approximately $4.6 billion USD (about $10 trillion GYD / $35.8 billion SRD[3]; Hamrick and Gallant 2017). The price per metric ton has fluctuated over recent years, ranging from $5 USD ($1,045 GYD / $37 SRD)[4] per metric ton in 2013 to just $3 USD ($627 GYD / $22 SRD) in 2016 due to more available carbon offsets than demands (Goldstein 2015). More than 25% of all transactions occur in the forestry sector and prices are higher than the average price, ranging from $4.20 USD ($878 GYD / $34 SRD) to $9.50 USD ($1,985 GYD / $71 SRD) for avoided deforestation and improved forest management, respectively (Goldstein 2015). Mangroves and other tidal wetlands are likely to be key components of the voluntary market since they are viewed as having many co-benefits outside of carbon mitigation, such as wave and storm protection. For example, the Mikoko Pamoja project in Kenya has negotiated $12 USD ($2,508 GYD / $90 SRD) per metric ton for buyers. The payments are used to restore and protect mangroves and fund education and clean drinking water initiatives in the community. Another prospective community-based mangrove restoration and protection project through Plan Vivo led by Blue Ventures in Madagascar is estimating carbon prices closer to $20 USD ($4,162 GYD / $149 SRD) per metric ton. Both examples are small volume projects focused on the social and environmental benefits and are looking for buyers interested in such a pedigree in a project.

5.3 Market Considerations

Due to the current high supply of voluntary offsets available and the long lead-time of setting up certifiable projects amenable for voluntary buyers, immediate engagement with potential buyers of offsets is recommended. Sharing summary information about the project, requesting indications of interest from buyers and negotiating future sales agreements are forms of engagement. These agreements can be structured in a way that allows the buyer to purchase an amount of credits in the future at a pre-agreed price, and to pay for these credits in the future when the credits are delivered. This type of "payment on delivery" agreement would provide the project with the certainty that credits

[3] Applying May 2019 USD/GYD/SRD exchange rates.

[4] About $4.96 USD in May 2019.

generated can be quickly monetized, while also protecting the buyer from loss of funds if credit generation is smaller or slower than expected.

5.4 Landscape Feasibility

The objective of the landscape feasibility analysis is to identify additional disturbed, deforested, or otherwise degraded mangrove areas in the NBS-LME and quantify an area that could be restored and included in a grouped carbon project. Grouped carbon projects allow multiple restoration activities, over a determined time period, by one or more landowners, to be combined into a single carbon project. With this option, smaller projects that are not cost-effective on their own, can join other projects and spread mostly fixed carbon costs over a larger land base.

Additionally, sea level rise is an important factor in the analysis, especially as it relates to project longevity, due to the potentially resulting mangrove migration, inundation, or erosion effects. The project boundary, which defines the extent of the project, needs to incorporate room for potential sea level rise effects. Sea level rise scenarios based on the IPCC or peer-reviewed literature are useful; however, the project boundary calculations need to be based on regional forecasts and need to be presented for both baseline and project conditions; global estimates of sea level rise are not sufficient.

5.5 Methodology Applicability

5.5.1 Methodology for Tidal Wetland and Seagrass Restoration (VM0033)

The VCS approved the first globally applicable methodology for tidal wetland restoration - VM0033[5] - in late 2015. The following 'applicability conditions' need to be met to use the methodology for a potential project in the Guianas:

1. Project activities restore tidal wetlands (condition #1);
2. Project activities include removing barriers and restoring tidal flows to tidally restricted wetlands (condition #2);
3. No productive activities are occurring in the project area that could be displaced from restoration and result in off-site emissions (condition #3);
4. While allowed, harvesting and prescribed burnings are not planned, and the project does not intend to request credits for fire risk reduction (conditions #4-7 are not applicable);
5. Afforestation/reforestation/revegetation activities (natural regeneration) will be combined with restoring hydrological conditions (condition #8);

[5] https://verra.org/methodology/vm0033-methodology-for-tidal-wetland-and-seagrass-restoration-v1-0/ accessed May 2019

6. Project activities (and related area) that avoid emissions from die-off/deforestation can be accounted for under this methodology (condition #9), (also see VM0007 discussion below);
7. Baseline conditions do not include commercial forestry (condition #10);
8. Project activities may lower the water table if improving hydrological connection (condition #11);
9. Hydrological connectivity of the project area to other areas will not result in increased offsite GHG emissions (condition #12); and
10. Project activities do not involve burning of soil (condition #13) or application of nitrogen fertilizers (condition #14).

5.5.2 REDD+ Methodology (VM0007)

If the potential for risk of loss can be demonstrated, there is potential for a conservation project. The VM0007[6] methodology provides a series of modules and tools that form the basic framework for a complete REDD baseline and monitoring methodology. These modules quantify GHG emission reductions and removals from REDD project scenarios, including avoiding unplanned and planned deforestation, and for activities to reduce emissions from forest degradation. Modules are now included for afforestation, reforestation and revegetation activities (ARR), and for activities that occur on peatlands and are combined with peatland rewetting or conservation (WRC). Identification of the most plausible VCS eligible activity is guided by a decision tree located in the REDD+ Methodology Framework (MF) module that provides the overarching structure for implementation of the VM0007 Methodology.

In 2019, VM0007 was revised to include modular procedures to recognize REDD+ projects on all peatlands. At the time of drafting this report, VM0007 is still under validation for additional models to recognize REDD+ projects in tidal wetlands and seagrass ecosystems. However, with the 2015 revision to VM0007, conservation projects on the coastal swamps may already be eligible for application.

The methodology includes the following project level 'applicability conditions' for REDD+MF:

1. **General**: All land areas already registered under a carbon trading scheme (both voluntary and compliance-oriented) must be transparently reported and excluded from the project area.
2. **All REDD activity types**

   REDD activity types are applicable under the following conditions:

   a. Land in the project area has qualified as forest for at least the 10 years before prior to the project start date. Mangrove forests are excluded from any tree height requirement

[6] https://verra.org/methodology/vm0007-redd-methodology-framework-redd-mf-v1-5/ accessed May 2019

in a forest definition, as they consist of nearly 100 mangrove species, which often do not reach the same height as other tree species, they occupy contiguous areas and their functioning as a forest is independent of tree height. If land within the project area is peatland or tidal wetlands and emissions from the soil carbon pool are deemed significant, the relevant WRC modules must be applied alongside other relevant modules.

b. Where the project activity involves the avoidance of future deforestation under deforestation or conversion concessions, which are without legal authorization and documentation at the project start date.

c. Baseline deforestation and forest degradation in the project area fall within one or more of the following categories:

   i. **Unplanned deforestation**: all project activities where the baseline agents of deforestation: (i) clear the land for tree harvesting, settlements, crop production (agriculturalist), ranching or aquaculture, where such clearing for crop production, ranching or aquaculture does not amount to large-scale industrial agriculture or aquaculture activities[7]; (ii) have no documented and uncontested legal right to deforest the land for these purposes; and (iii) are either residents in the reference region for deforestation or immigrants.

   ii. **Planned deforestation/degradation**: conversion of forest lands to a deforested condition must be legally permitted.

   iii. **Pre-project unsustainable fuelwood collection**: where, pre-project, unsustainable fuelwood collection is occurring within the project boundaries, modules BL-DFW and LK-DFW shall be used to determine potential leakage.

   iv. **Degradation (fuelwood, charcoal)**: Fuelwood collection and charcoal production must be "non-renewable" (as defined in Module BLDFW) in the baseline period. If degradation is caused by either illegal or legal tree extraction for timber, this framework cannot be used.

3. **ARR**

a. Procedures for estimating carbon stock changes in ARR project activities are provided in BL-ARR and M-ARR. Where exclusion of project activities on wetlands exist in the applicability conditions of methodologies and tools, these can be neglected for the purpose of their use within this Methodology Framework, as accounting procedures for the peat soil are provided in BL-PEAT and M-PEAT.

b. The with-project scenario does not involve the harvesting of trees. Therefore, procedures for the estimation of long-term average carbon stocks are not provided.

[7] Small-scale / large-scale agriculture and aquaculture is to be defined and justified by the project.

c. The with-project scenario does not involve application of nitrogen fertilizer.

4. **WRC**

a. Fire reduction projects on peatland that exclude rewetting as part of the project activity are not eligible.

b. Rewetting of drained peatland and conservation of undrained or partially drained peatland may be implemented in combination with REDD project activities. REDD project activities on peatland shall not increase drainage.

c. Rewetting of drained peatland may be implemented as a separate activity or in combination with ARR project activities. ARR activities shall not enhance peat oxidation and therefore this activity requires at least some degree of rewetting.

5.6 Additionality Requirements

In addition to meeting the methodology's 'applicability conditions', a project must also satisfy an additionality requirement to ensure that the project would not have occurred in the absence of carbon market incentives or as part of "business-as-usual" activities. In the case of the VM0033, tidal wetland restoration projects located in the United States are deemed to meet the additionality requirement (due to the low penetration or occurrence of these activities) so long as they are not required by any law, statute, or other regulatory framework. Since most restoration activities globally occur within the United States and these activities are deemed additional, it can be argued that any restoration within Guyana and Suriname is additional (Needelman et al. 2018).

5.7 GHG Accounting Approach

The approved approaches for measuring the GHG benefits of a potential mangrove restoration project are contained in VM0033, and for conservation in VM0007. As discussed above, the GHG benefits for a potential project would be calculated as the sum of the GHG benefits in the biomass carbon pool and the soil organic carbon pool minus any CH_4 and/or N_2O emissions. In both cases, GHG benefits are calculated as the difference between emissions in the "baseline" (assuming no restoration occurs) and "with-project" (after restoration occurs) scenarios.

Non-permanence deductions represent a 10% contribution of net emission reductions to the VCS non-permanence buffer pool. All land-use projects must contribute to this pool that protects against future reversals or loss of carbon previously credited (e.g. due to erosion) that may occur. Contributions are determined using the VCS non-permanence risk tool. If project development is pursued, further analysis and support will be needed to complete the non-permanence risk assessment and to determine the buffer contribution.

5.8 Financial Feasibility

There are several factors that need to be considered regarding the financial feasibility of a carbon project. A few key financial assumptions that could be used in a carbon project analysis are described below.

1. Net emission reductions represent the difference between the baseline and with-project emissions as discussed above.
2. Tradeable carbon offsets are the net emission reductions less non-permanence deduction and represent the amount of carbon credits that can be sold.
3. A range of carbon prices were estimated to account for low, base case, and high scenarios. For example, in a project proposed in mangroves of Florida, USA, the carbon prices examined were $5, $7.50, and $15 USD ($1,045, $1,567, and $3,135 GYD / $37, $56, and $112 SRD) per metric ton. As previously mentioned, the mangrove carbon project in Kenya garnered $12 USD ($2,508 GYD / $90 SRD) per metric ton.
4. Carbon development and validation costs are assumed to be $150,000 USD ($31.3 million GYD / $ 1.1 million SRD) and relate to the third-party fees and travel expenses of preparing ($100,000 USD / $20.9 million GYD / $0.7 million SRD) and validating ($50,000 USD / $10.4 million GYD/ $372,900 SRD) the Project Description to be registered with the VCS. These are one-time expenses that are incurred at the inception of the project.
5. Carbon monitoring and verification costs are assumed to be $30,000-60,000 USD ($6.3-12.5 million GYD / $223,740-447,480 SRD) per monitoring event assuming 5-year monitoring intervals (maximum elapsed time between verifications before VCS buffer credits are put on hold). The estimated costs include the costs of collecting field biomass and soil organic carbon data ($20,000 USD / $4.2 million GYD / $149,160 SRD per event) and preparing and verifying the VCS monitoring report ($40,000 USD / $8.4 million GYD / $298,320 SRD per event).

5.9 Legal Feasibility

Project planning must include analysis of: (1) land tenure, (2) carbon rights, (3) taxation issues, (4) need for licenses and permits, (5) state of relevant legislation and regulation; and (6) potential transactional structures for a project. The VCS requires the project proponent (owner) to demonstrate carbon ownership by a right of use. For land use projects, a right of use can arise by virtue of property rights in land of the project area, or by an enforceable and irrevocable agreement with the landowner who transfers such rights to the project proponents.

5.10 Organization Feasibility

To create and implement a project, an entity with overall management responsibility needs to be established. The key participants in land-based carbon projects consist of the project developer, landowners, implementation partners, technical partners, and funding partners. The project developer plays a central role in the project, coordinating the efforts with each of the other participants and the registry. The project needs to be registered with an account at the registry (generally with a project design document and validation report) and registry requirements need to be met (captured in monitoring and verification reports) to receive credits. The project developer directs this process and the transfer or retirement of credits. The role of the landowner generally gets established in a landowner agreement. The role of the implementation partners is to first obtain engineering designs and regulatory permits for restoration measures and, secondly, implement the prescribed restoration measures, including initial restoration and ongoing management. The technical partners provide research, technical consulting, and third-party validation and verification. Funding partners can include offset funding partners and sometimes grant funding partners. An offset funding partner would provide funding to the project through advance payments and/or payments-on-delivery in return for emission reductions rights (credits). If a grant funding partner can be secured in advance, this funding partner would provide initial grant funding to cover some or all of the initial costs that are incurred prior to the generation of carbon revenues (e.g. project design and validation costs as well as the monitoring and verification costs for the first monitoring event). As the project moves forward, agreements between participants are developed, detailing roles and responsibilities for the specific project. Figure 9 highlights the connections across partners in a carbon project.

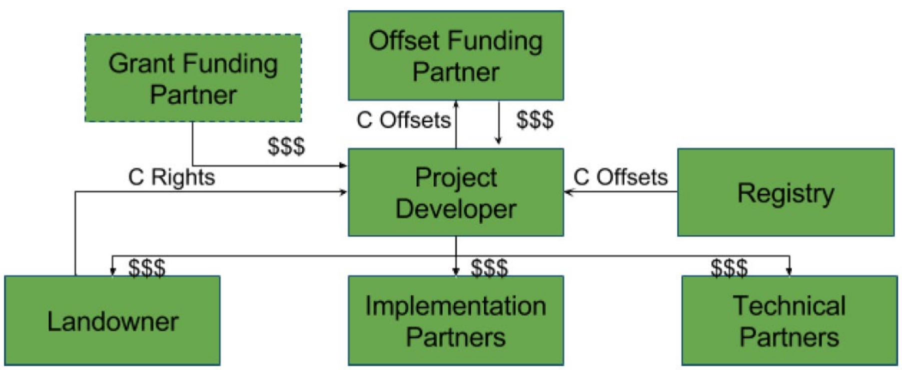


Figure 9. Generalized organizational structure of a carbon project.

5.11 Relevance to the NBS-LME

There is significant potential for carbon projects in Suriname and Guyana through mangrove restoration and conservation under VCS or Plan Vivo methodologies. Preventing emissions by protecting intact wetlands from erosion is a recognized potential project activity under the VCS. If such measures are possible, which has yet to be determined, a first order estimate of the emissions reductions are as follows. Considering a 1 km long mangrove stand, avoided retreat of 243 m (total area 24.3 ha) would maintain a total carbon stock of 7,316 Mg C (26,828 Mg CO_2e; using values in Table 2), which equates to $402,415 USD ($8.4 million GYD / $3.0 million SRD) assuming carbon credits at $15 USD Mg CO_2e^{-1}

($3,139 GYD / $112 SRD) and negligible CH_4 and N_2O emissions. If the scale of the prevented erosion were greater, for example avoiding mangrove retreat by 609 m (total area 60.9 ha), the potential value of carbon credits would increase to $1,008,522 USD ($210.6 million GYD / $7.5 million SRD). This assumes 1 meter of soil would be impacted. Total revenue for each scenario is also calculated for lower value carbon credits and for larger project areas (Table 12). A mangrove restoration project would result in slightly greater carbon credits. IPCC tier 1 values for aboveground growth and root to shoot ratio were used to obtain estimates of tree C accumulation of 199.3 Mg C ha^{-1} (730.8 Mg CO_2e ha^{-1})[8] over a 30-year project period. Soil C accumulation is assumed to be the average value for mature mangroves >20 years old from Table 2: 204.1 Mg C ha^{-1} (748.4 Mg CO_2e ha^{-1})[9]. Cumulatively, this results in a total of 403.4 Mg C ha^{-1} (1,479.1 Mg CO_2e ha^{-1}) over a 30-year project duration, assuming negligible CH_4 and N_2O emissions. Using this value and a range of potential project areas and carbon credit values, total C project revenues are presented in Table 12. Project implementation and monitoring costs are estimated to be $350,000 over 30 years (see section 5.8). Given the conservation and restoration examples presented here, a project, or group of projects, at a minimum would need to be at least 63 and 47 ha, respectively, at a carbon value of $5 Mg CO_2e^{-1} to break even with project implementation costs. Further analysis is required to determine the magnitude of erosion risk, options for reducing erosion and depth of soil that would be protected. An agency, whether a non-profit or government entity, would also need to be named as the project developer before proceeding with a carbon financing feasibility report. A project that is supported and ideally organized by the local community, such as in Kenya (see footnote 2), with agency and government support, would likely have the most traction and could maximize all potential co-benefits.

[8] This value includes the IPCC Wetlands Supplement tier 1 estimate for wet tropical mangrove aboveground biomass accumulation rate of 9.9 Mg ha^{-1} yr^{-1} and the tier 1 root to shoot ratio for wet tropical mangroves of 0.49 (Hiraishi et al. 2014); conversion of dry weight to C is 0.48 and 0.39 for above- and belowground biomass, respectively (Howard et al. 2014).

[9] Since mangroves along the NBS-LME start from bare mudflat conditions with a uniform stand age like in a restoration project, it is assumed that the mature mangrove soil C stock of stands >20 years from Walcker et al. (2018) is representative of restoration conditions as well.

Table 12. Estimated carbon revenue for conserving and restoring mangroves along the NBS-LME assuming a project duration of 30 years. Total revenue does not include project implementation, planting or monitoring costs.

Project Type	Project size (ha)	C revenue, \$5 Mg CO_2e^{-1} (\$ USD)	C revenue, \$7.50 Mg CO_2e^{-1} (\$ USD)	C revenue, \$15 Mg CO_2e^{-1} (\$ USD)
Conservation	24.3	\$134,138	\$201,208	\$402,415
Conservation	60.9	\$336,174	\$504,261	\$1,008,522
Conservation	1,000	\$1,505,482	\$2,258,223	\$4,516,446
Conservation	10,000	\$55,201,007	\$82,801,510	\$165,603,020
Restoration	50	\$369,826	\$554,739	\$1,109,478
Restoration	100	\$739.652	\$1,109,478	\$2,218,957
Restoration	1,000	\$7,396,523	\$11,094,784	\$22,189,569
Restoration	10,000	\$73,965,228	\$110,947,843	\$221,895,685

Both Guyana and Suriname are considered High Forest Low Deforestation (HFLD) countries, meaning that they have high forest cover[10] and historically low deforestation rates, but that future population, economic and agriculture pressures could result in increased deforestation rates. As stated earlier, the creation of REDD+ project enables these countries to generate revenue through forest conservation, enhancement of degraded forests, and sustainable management, and both countries are at different stages of the REDD+ implementation process. In 2009, Guyana and Norway signed a Memorandum of Understanding where up to \$250 million USD over five years were committed by Norway to support Guyana's goal to reduce forest degradation and deforestation, which was one of the first nationwide REDD+ plans implemented. The funding allowed the Guyana Forestry Commission and the Office of the President to determine forest carbon stocks, historical rates of degradation and deforestation, develop a Monitoring, Reporting and Verification System (MRVS), and establish a Guyana REDD+ Investment Fund (GRIF). The GFC manages Guyana's State forests, which includes mangroves, and is tasked with the technical implementation of REDD+ as well as intensive and thorough stakeholder and public outreach. Guyana, in accordance with the Norwegian government, developed a REDD+ Governance Development Plan that outlines the main activities and their time frame and lead responsible agencies. A mid-term progress report released in 2019 concluded that three quarters of the 34 progress indicators

[10] These countries, along with nine other developing countries, hold approximately 18% of the world's forest carbon.

had either been achieved or are in the process of or had significant progress in achieving[11]. In 2013, Suriname began the process of creating a REDD+ proposal and received $3.8 million USD, implemented through the United Nations Development Program (UNDP), to create a Readiness Preparation Proposal (R-P P), which describes the country's plan for a REDD+ strategy, policy development, stakeholder involvement, MRVS, management structure, and monitoring and evaluation framework. The R-P P is implemented by the Climate Compatible Development Unit (in the Cabinet of the President of Suriname), the National Institute for Environment and Development in Suriname (NIMOS), and the country's 17 ministries (CCDA 2013). Suriname also received a Widening Informed Stakeholder Engagement for REDD+ (WIDE REDD+) grant in 2013 to broaden stakeholder engagement in the preparation of the R-P P.

The dynamic interplay between mud bank locations and mangrove presence along the NBS-LME poses an intriguing quandary in that the system is not fixed in space and time compared to other fringing mangrove systems worldwide. A particular mangrove stand is relatively short-lived (~60-70 years); however, the larger-scale cumulative progradation and erosion of mangroves results in a relatively consistent country-level ecosystem carbon stock, given that deforestation or other degradation would not disrupt the natural processes. Under most carbon standards, an increased carbon stock or avoided loss of carbon stock as a result of a project activity must be maintained for a long period (usually at least for 100 years), and its reversal must be avoided. Permanence is important when emission reductions or removals are used as offsets – if the underlying carbon stock disappears, the offset will also be affected. Current project standards offset the risk of non-permanence by issuing only temporary credits, or by installing a fixed (e.g. Gold Standard) or variable (e.g. VCS) buffer withholding. For example, in VCS language, the "non-permanence risk analysis only needs to be applied to GHG removals or avoided emissions through carbon sinks. Project activities generating emissions reductions of N_2O, CH_4 or fossil-derived CO_2 are not subject to buffer withholding, since these GHG benefits cannot be reversed".[12] Non-permanence risk is seen to consist of three risk factors, internal, external, and natural risks, for which rating can be obtained. Under the VCS, the total risk rating shall not exceed a value of 60% or the project risk is deemed unacceptably high and thus the project not eligible. Note that each percent withholding means a deduction on the return on investment, although the standard has created opportunities to reduce the withholding over time. Especially for mangrove restoration projects conducted outside of seawalls, which are more prone to erosion, it would need to be determined whether they would pass such a permanence test. The overall best placement for a project thus needs to be considered at the onset. For example, projects on set-back sites may have a greater chance on longevity than projects in front of sea walls, yet they may also be smaller in scale and thus provide less

[11] https://www.forestcarbonpartnership.org/country/guyana; accessed in June 2019

[12] See the VCS AFOLU Non-Permanence Risk Tool at https://verra.org/project/vcs-program/rules-and-requirements/; accessed June 2019

overall area for which credits can be secured. In addition, it will need to be determined whether the historic rates of loss of mangroves in the NBS-LME can give an indication of future rates, especially given that national land use plans highlight both risk of development on the coastal plains with sea level rise, and yet also mention the opportunity to increase agricultural and urban development in these areas.

There has been strong interest in restoring mangroves in Guyana, and multiple restoration efforts have been implemented under the Guyana Mangrove Restoration Project, with differing success. Some restoration sites successfully transitioned to young stands while others were washed away due to strong storm events. The cost of implementation could be quite high, ranging from $25k to $100k USD ha^{-1} ($5.2 to 20.8 million GYD / $186,450 to $745,800 SRD; Lewis III 2005). The best and most economical approach along the NBS, taking into consideration the size and dynamic nature of the muddy system, is to conserve mangrove patches and maintain the freshwater connections with coastal freshwater swamps so that the ability for natural regeneration can be maintained (Anthony and Gratiot 2012). Yet, regeneration of the mud bank – interbank system, which ultimately governs the stability and persistence of coastal mangrove stands, may take decades to re-establish in zones where mangroves have been eradicated (Gratiot et al. 2008). In sheltered locations with low wave energy and gentle foreshore slopes, "green" or nature-based coastal defense strategies are preferred because they provide a variety of co-benefits, including enhancing or increasing biodiversity and promoting human well-being. Green solutions can be coupled with habitat restoration to meet multiple species and community goals. In locations with high wave energy and steeper foreshore slopes, more traditional "gray" or engineered coastal defense strategies are more common. Gray strategies can provide a higher level of flood protection than green strategies, but gray strategies often have ecosystem impacts, including habitat loss and disconnecting communities from the shoreline. Green and gray strategies can be integrated to develop solutions that provide coastal hazard reduction (during high water and wave events), while also enhancing habitat health. These hybrid "green-gray" nature-based solutions can also help preserve the connection between upland and coastal ecosystems and maintain community access to the shoreline.

The assessment of carbon storage and emissions presented here is limited to mangroves, despite the knowledge that coastal freshwater swamps in Suriname and Guyana have the potential to sequester soil carbon for a longer period than mangroves, likely to a greater depth as well, and are also more at risk of loss due to growing pressures from urban growth, aquaculture and agriculture. Research in Indonesian forested peat swamps have identified rich carbon stores and, subsequently, large carbon emissions associated with conversion to aquaculture and oil palm plantations (Novita 2016, Arifanti et al. 2019). Few data pieces are available, however, regarding how much carbon coastal freshwater swamps produce and store and what the rates of loss and types of conversions are along the NBS-LME. Future research into these data needs would be beneficial to provide more insight into expanded carbon project capabilities along the NBS.

6 Next Steps

As noted previously, country-specific mangrove carbon stock assessments should occur in Guyana and Suriname. Appendix 1 highlights the key methods used for quantifying carbon stocks, which have been used globally and allow for ease of analysis across sites and countries (Howard et al. 2014). Additionally, yearly mangrove change analysis specific to Suriname and Guyana using remotely sensed data (e.g. Landsat) should occur to elucidate the underlying reasons behind mangrove loss over time. Country-based estimates from global mangrove area assessments do not necessarily capture the subtle nuances in the key drivers of change, which are unique in the NBS-LME due to both natural (shifting mud banks and the resulting natural patterns in progradation and erosion) and anthropogenic causes. Currently, it does not appear that there is active widescale mangrove removal but there could be antecedent effects from historic removal that are preventing natural colonization dynamics. This is also needed to assess potential leakages in the development of a carbon financing project.

The location of mudbanks along the NBS coast should be considered when identifying targeted areas for restoration. There is likely a higher chance for restoration success in areas that are on the leading edge of a mudbank, since the wave reducing and sediment building properties of mudbanks create a protective boundary behind which mangrove colonization and growth can occur. Depending on the size of the mudbank, this increased protection could last upwards of 60 years. Similarly, restoration success is likely to be lower in areas at the tail end of a mudbank or within an interbank zone, where erosive conditions are more likely to persist. Efforts also could be implemented to reduce erosion of existing mangroves and enhance sediment accretion.

Finally, a carbon stock assessment combined with research into the extent and drivers of deforestation are needed in coastal freshwater swamps along the NBS-LME. Currently, little to no data exists. Coastal swamps are integrally tied into the coastal plains of Suriname and Guyana, providing freshwater to mangroves and many local communities, habitat for wildlife, and potentially large soil carbon stores. The extent of coastal swamps is larger, with estimates of 2.7 million ha in Guyana and 361,000 ha in Suriname (Alder & van Kuijk 2009, GONINI 2019). The anthropogenic impact, both in urban and agricultural development, currently appears to be greater here than in mangroves due to the rich organic soils, and has a greater chance of affecting freshwater flow and coastal plain dynamics.

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Appendix 1: Mangrove Carbon Stock Sampling Protocol

Below is a summary of key components for sampling carbon pools in blue carbon ecosystems, specifically mangroves. More detailed sampling protocols and laboratory methods, from which this synopsis is drawn from, are found in Howard et al. (2014).

Pre-Sampling Logistics

Prior to conducting a field campaign, key sites should be chosen that represent the range of mangrove stand ages (mudflat, pioneer, young, mature, senescent, degraded), species assemblages (black, red, and white mangroves), and landscape position (coastal, riverine). This can be completed through analysis of aerial imagery (i.e. Google Earth) combined with land manager and local knowledge. Site access and tide levels need to be considered; sites ideally should be sampled at low tide. Multiple replicates of each strata should be sampled across the mangrove extent in both Guyana and Suriname.

Tree Sampling

The most common methodology for estimating tree carbon stocks involves a 100 m long transect with six plots spaced 20 m apart and oriented perpendicular to the shoreline/river edge (Figure 1). The spacing between plots and number of transects may need to be adjusted based on mangrove stand width. The coordinates for each plot center are recorded using a GPS and marked with a PVC pipe if plots will be revisited over time. Within each plot, all trees with a diameter at breast height (DBH) 5 cm or greater are measured within a 7 m radius circle. The stem diameter is measured 1.3 m from the ground, or directly above the highest prop root if branching still occurs at or above this point. Depending on the age of the forest or general stand characteristics, however, the DBH size requirement may need to be lowered (see Schile et al. 2017). Tree height, canopy radius, and DBH of each applicable tree are measured. Standing dead trees are also measured within the plot, using the decay status detailed in Howard et al. (2014). Within a 2 m radius circle, centered at the same point as the larger plot, all trees with DBH < 5 cm are measured as above, and all seedlings, defined as trees less than 1.3 m tall, are counted. Dead and downed wood stocks are quantified along two perpendicular 10 m long transects crossing at the plot's center. Species specific allometric equations are used to quantify above- and belowground biomass, which is converted to carbon content using default values of 0.48 and 0.39, respectively.

Soil Sampling

Near the center of each plot in undisturbed sediment, a soil core is collected. The core should be collected to a depth of 3 m, until refusal or until the depth of mangrove influence is reached, whatever comes first. A 1 m long open-faced gouge corer is the most effective at removing mangrove soil without compaction (Figure 2). The corer is inserted perpendicular to the ground, without pushing the top below the surface or inserting at an angle. The corer is rotated clockwise while it is removed slowly to maintain core integrity. Once removed, the corer is placed flat on the ground, the top of the exposed core is cut even with the corer surface, the length is measured, and the core is subdivided for processing in the laboratory. Due to the shallow nature of mangrove-influenced soils along the NBS, ideally the entire top 50 cm should be collected in 5 cm increments. If sample processing or funding is limited, however, the first 30 cm should be sampled in 5 cm increments, at the midpoint between 30 and 50 cm and at the midpoint of every subsequent 50 cm section. If unique soil profiles are found within the core including parent or bedrock sediment, sample them separately and note the depth at which they are found. Once the first meter has been sampled, use the same hole to collect the next meter of soil. Soil samples should be placed into metal tins, labeled, and transported back to the lab to be dried, weighed, and processed. See Howard et al. (2014) for detailed methods for analyzing soil carbon content. Once the core has been removed, collect pore-water from the hole before filling it in with remnant core soil. At a minimum, salinity and pH should be measured. Depending on the scope of work, more water samples can be collected for analysis of methane, sulfate and sulfide, nitrate, among other biogeochemical properties.

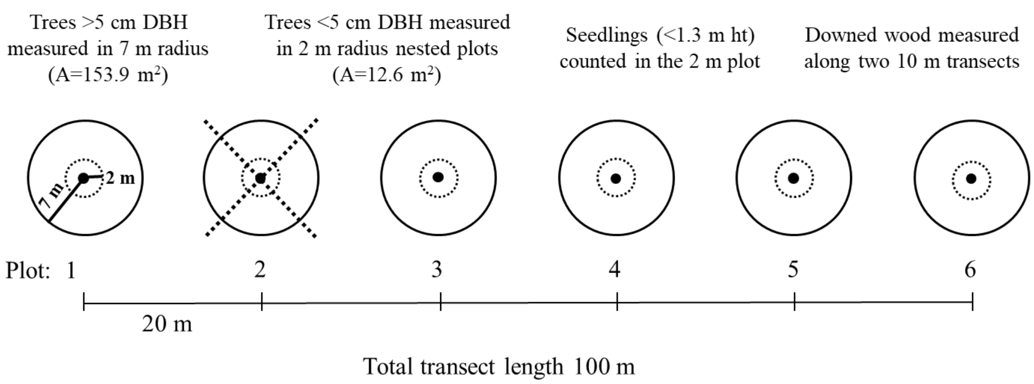

Figure 1. Schematic of mangrove sampling design (adapted from Schile et al. 2017).


Figure 2. Image of an open-faced gouge corer with a mangrove peat core. The top portion of the core is being removed before the subsamples are collected (Photo: C. Beers).

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