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Human-Driven Degradation Impacts on Mangroves in Southern Sierra Leone

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Master of Science - International Environmental Studies

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Declaration

I, (Lea Christin Huber), declare that this thesis is a result of my own work. All sources of information other than my own have been acknowledged and referenced.

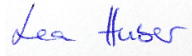
I have not collected the data myself but have used data collected under the responsibility of Moses Nsanyi Sainge and Richard Abdoulaye Kamara.

This thesis has not been previously submitted to any other university for award of any type of academic degree.

This work is written in the format of a scientific article and has additionally been submitted to the journal *Forest Ecology and Management*.

Date: 13/05/2023

Signature:

A handwritten signature in blue ink that reads "Lea Huber". The signature is written in a cursive style and is placed on a light blue rectangular background.

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Abstract

Mangroves are important stores and sinks of blue carbon, foster biodiversity, and provide ecosystem services supporting local livelihoods and enhancing coastal protection. However, mangroves are declining in extent and experiencing degradation due to human activities. This study investigated how different levels of human-driven degradation affect forest structure, above- and belowground carbon stocks, tree species composition, and the population structures of dominant tree species. We investigated 19 transects of different degradation levels (pristine, moderately degraded, heavily degraded) in various locations (oceanic, riverine, interior) in the Sherbro river estuary in Sierra Leone.

Results showed that pristine mangroves stored 707 Mg ha⁻¹ of carbon including soils to 1 m depth, which is higher than values reported from many other regions in Africa. Degradation resulted in declining basal area, decreasing density of large trees (≥ 30 cm diameter) but increasing stem density of small trees (≥ 5 cm diameter). All carbon pools declined with increasing degradation across all locations. While above- and belowground carbon decreased by 96 %, soil carbon (up to 1 m depth) decreased by 44 %. Heavy degradation resulted in a change in the dominating species from *Rhizophora racemosa* to *Rhizophora mangle* across all locations. Overall, we found major effects of degradation on mangrove characteristics and minor effects of location. We urge conservation of pristine forests, restoration of degraded mangroves, and the establishment of management systems that are ecologically informed and based on a comprehensive understanding of human-driven degradation impacts.

Keywords

blue carbon, mangrove conservation, species composition, restoration potential, West Africa

Research Highlights

- Pristine mangroves stored 707 Mg ha⁻¹ of carbon including soils to 1 m depth.
- Aboveground carbon in pristine mangroves exceeded estimates of most studies in Africa.
- Degradation decreased above- and belowground carbon by 96 %, soil carbon by 44 %.
- Heavy degradation changed dominating species from *Rhizophora racemosa* to *R. mangle*.

Human-Driven Degradation Impacts on Mangroves in Southern Sierra Leone

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1 Introduction

Mangroves are salt-tolerant forest ecosystems growing in the intertidal zone in tropical and subtropical regions (Hamilton, 2020). Mangroves are one of the most carbon-rich ecosystems per unit area (Twilley et al., 2018) as they store carbon in trees but also in sediments that are trapped by roots and pneumatophores (Donato et al., 2011; Kathiresan, 2021). Therefore, mangroves are receiving increased attention for their potential contribution to climate change mitigation as part of the 'Blue Carbon ecosystems' (Leal & Spalding, 2022), which also include seagrass beds and salt marshes (Alongi, 2020; Rao et al., 2021).

Apart from their role in climate change mitigation, mangroves protect coastal areas from erosion, tides, and tropical cyclones (Carugati et al., 2018; Kathiresan, 2021), they maintain water quality by filtering and trapping pollutants and sediments (Nunoo & Agyekumhene, 2022), and they serve as habitat, breeding, feeding and nursery sites for terrestrial and marine organisms including important commercial species, such as different fish species, oysters or shrimps (Carugati et al., 2018; Dahdouh-Guebas et al., 2020; Nunoo & Agyekumhene, 2022). Furthermore, mangrove ecosystems support local communities' livelihoods by providing them with provisioning services (such as food, fuelwood, medicine, or construction materials), and cultural services (identity, tourism etc.) (Ajonina et al., 2018; Jones et al., 2014; Kathiresan, 2021). In brief, mangroves are of key importance from local to global scales.

Despite their importance, more than 1 million hectares of mangroves were lost globally between 1990 and 2020 (FAO, 2020). Even though rates of decline have decreased in the past few years (Arumugam et al., 2020; Hamilton & Casey, 2016; Lee et al., 2021), mangroves diminish 3–5 times faster than other forest ecosystems (Dayal et al., 2022; UNEP, 2014). Distributed on 113 countries, there are 14.8 million hectares of mangroves left, of which about 21,200 hectares are lost annually (FAO, 2020). According to an estimate by Liu et al. (2021) 2,007,411 hectares of mangroves are found in West Africa, of which 170,341 hectares are located in Sierra Leone, which positions the country among the top ten countries in Africa when it comes to both mangrove extent and change in extent (UNEP, 2023). Compared to available research globally, there are few studies on mangroves in Africa, especially in West Africa. Mangroves in East and West Africa differ in species composition and usage pressures (Machava-António et al., 2022), so research needs to focus on all different regions.

Natural causes (e.g. tropical cyclones) can lead to mangrove loss in some sites. However, anthropogenic causes such as infrastructure development, conversion to agriculture, aquaculture, over-extraction of forest products, salt production, mining, and pollution are driving most mangrove loss across the tropics (Dayal et al., 2022; Goldberg et al., 2020; Goussard & Ducrocq, 2014; Islam et al., 2018; Jones et al., 2014; Yando et al., 2021).

Restoring areas from which mangroves have disappeared entirely, often requires planting nursery-raised mangrove seedlings, while passive restoration can be a viable

approach for areas that are not yet fully degraded (Moudingo et al., 2018). Many restoration projects failed in the past, both due to financial constraints, institutional challenges in implementation and insufficiently addressed root causes of degradation, but also due to insufficient understanding of mangrove ecology and lack of locally adapted techniques (Leal & Spalding, 2022). In order to get a comprehensive picture of the effects of human-driven degradation on mangroves, to establish a baseline for restoration efforts, and to design effective restoration and conservation measures, impacts of degradation on forest structure, species composition, ecosystem functions, and related loss of ecosystem services should be investigated (Yando et al., 2021).

Some studies in Africa have already addressed certain effects of human-driven mangrove degradation. For instance, Ajonina et al. (2014a) studied mangroves in Cameroon, Gabon, the Republic of Congo (RoC), and the Democratic Republic of Congo (DRC). These authors showed that all carbon pools (aboveground (AGC), belowground (BGC), soil organic carbon (SOC)) decreased with increasing human-driven degradation pressures. While AGC decreased by 74–88%, SOC decreased by 20–23% (Ajonina et al., 2014a). However, Ajonina et al. (2014a) did not investigate effects on species composition. Indeed, very few studies considered the effect of degradation on species composition. A study in Gazi Bay in Kenya by Kihia (2014) reported changes in dominance and abundance of species when comparing disturbed and undisturbed sites. Additionally, population structures of species, the proportion of stems of different age or diameter classes, can provide knowledge about harvesting impacts and regeneration capacity as was demonstrated for different tropical forests including mangroves (Ajonina et al., 2014a; Batumike et al., 2022; Obiri et al., 2002). Other studies addressing mangrove degradation, either generally or specifically in Africa, often focused on loss of areal extent (e.g. (Lee et al. 2021), impacts on flora and fauna (Carugati et al., 2018; Nunoo & Agyekumhene, 2022) or investigated drivers or narratives of degradation as well as impacts on local livelihoods (Andrieu et al., 2020; Dayal et al., 2022). However, there is a lack of studies that comprehensively compare different degradation levels within the same region, rather than comparing differently degraded regions, as in the latter case degradation impacts can be confounded by location.

Depending on their location, mangroves are affected by rivers and tides to varying degrees (see e.g. (Kathiresan, 2021; Kjerfve, 1990)). Therefore, it is important to also consider potentially context-dependent effects of degradation on mangroves at different locations (e.g. oceanic, riverine, interior). When these impacts combined are considered to get a more complete picture of human-driven degradation, conservation and restoration plans can be improved, either preserving existing ecosystems or assisting the recovery of degraded ecosystems. If ecological requirements and ecological knowledge are neglected, success of e.g. restoration projects is limited (Primavera & Esteban, 2008).

To fill these knowledge gaps this study focused on oceanic, riverine, and interior mangroves in the Sherbro river estuary in Sierra Leone and addressed the following research questions:

- 1) How does human-driven degradation affect mangrove forest structure and different carbon pools (aboveground, belowground, soil)?
- 2) How does human-driven degradation affect tree species composition?
- 3) How do different drivers of degradation affect population structures of dominant tree species?

2 Methods

2.1 Study Area

Our study focused on mangroves in the Sherbro river estuary (thereafter SRE) in southern Sierra Leone, on the Atlantic Coast of West Africa (see Fig. 1). The SRE constitutes one of four marine protected areas in Sierra Leone and encompasses about 50% of the country's mangroves, about 90,000 ha (estimate for 2016) (Feka & Sainge, 2022; Mondal et al., 2017). The climate is tropical and humid with annual rainfall at the coast ranging from 3,000 to 5,000 mm and average temperatures of 26° C (SLMet, 2023). The rainy season extends from May to November (SLMet, 2023).

Compared to other estuaries in Sierra Leone, the SRE is characterised by low species diversity, higher number of large trees (defined as ≥ 30 cm diameter), and less disturbance and degradation over the past decades (USAID, 2020). The estimated human population of the SRE region comprises 223,230 people distributed among 105 villages and belonging mainly to Mende and Sherbro ethnic groups (Feka & Sainge, 2022). Pressures driving mangrove loss in the SRE are wood harvesting for fish smoking, cooking, and house construction as well as land conversion for agriculture (rice farming) and infrastructure development. Main species used are *Rhizophora mangle* (for cooking and construction of houses), *Rhizophora racemosa* (for fish smoking), *Rhizophora harrisonii* (for construction of houses and canoes) and *Avicennia* spp. (for construction of houses and fish smoking) (Sainge, 2022).

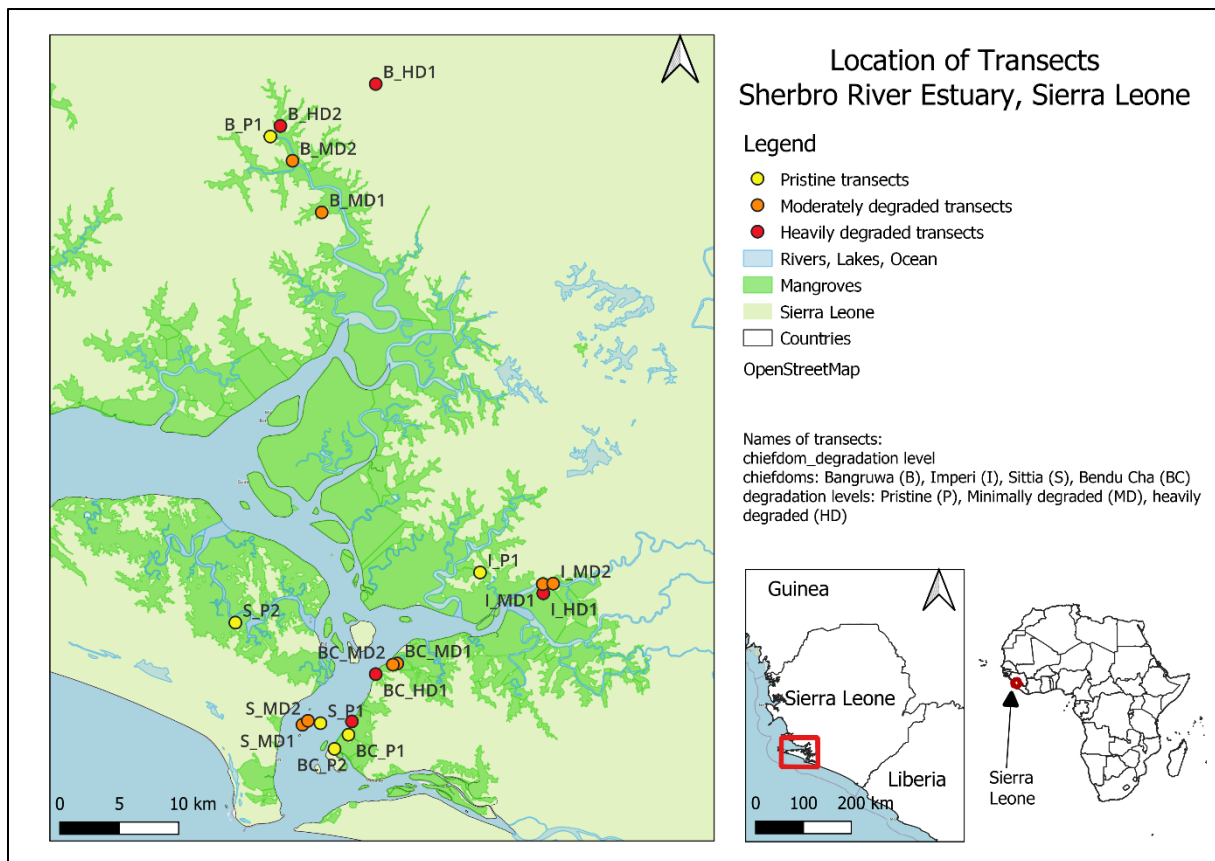


Figure 1: Map of the Sherbro river estuary in Sierra Leone indicating transect locations and respective degradation level (data sources: OpenStreetMap, ICPAC Geoportal)

2.2 Data Collection and Analysis of Soil Samples

Data collection took place in November 2021 and during low tides. A total of 19 transects of 20x100 m were laid out across different parts of the SRE. The location of transects depended upon accessibility from existing villages located across four chiefdoms (Imperi, Bangruwa, Bendu Cha and Sittia), so they are not randomly distributed across the whole SRE (see Fig 1.). Based on calculations of distance to shoreline and closest rivers, transects were classified as riverine (all transects in Imperi, n = 4), interior (all transects in Bangruwa and transect S_P2, located in the interior of the Sherbro island in Sittia, n = 6) and oceanic (all remaining transects in Sittia and all in Bendu Cha, n = 9). At each location, we distinguished sites of three different forest use regimes based on a priori descriptions of use by local communities: Pristine (thereafter P, intact or near intact, old-growth mangroves, without disturbance, n = 6 transects), moderately degraded (thereafter MD, in which wood extraction for fish smoking, building materials or firewood is common, n = 8 transects) and heavily degraded (thereafter HD, in which rice farming or heavy cutting of mangroves is or has been taking place, n = 5 transects). Note that due to time constraints, one pristine transect in Bangruwa was shorter (20x 60m).

In each transect, for each tree with ≥ 5 cm diameter at breast height (Dbh) or at 30–50 cm above the last stilt root, diameter (Dbh), height, and species were recorded adapting

methods from the CIFOR protocol (Kauffman & Donato, 2012). The status of all observed species was checked on the IUCN Red List (IUCN, 2023). The results given in the following only refer to living trees.

In each of the four chiefdoms, one soil sample was taken with a cylindrical auger of known volume (10 cm diameter, 20 cm height) for each of the three degradation levels (except for Bangruwa, where there is no soil sample from pristine mangroves due to time constraints). Thereby, soil samples covered P (n=3 transects), MD (n=4 transects) and HD mangroves (n=4 transects). Adhering to the recommendation given in de Jong Cleyndert et al. (2020), each location was sampled until 1 m depth, subdivided as 0–15 cm, 16–30 cm, 31–60 cm and 61–100 cm. Soil samples were air-dried and sieved through a 2 mm sieve. Soil carbon was determined using the Walkley-Black method (Walkley & Black, 1934) at the Njala University Quality Control Laboratory in Sierra Leone.

2.3. Data Analysis

To investigate differences in forest structure between degradation levels, we calculated stem density of all trees (≥ 5 cm diameter), stem density of large trees (≥ 30 cm diameter) and basal area ($\text{m}^2 \text{ha}^{-1}$) for each transect, and then computed the average per degradation level. For each transect, we further visualised species' abundances in terms of percentage of stems, using QGIS.

To estimate aboveground biomass (AGB), we used species-specific AGB equations where possible (for *Avicennia ssp.*, *Rhizophora mangle*, and *Rhizophora ssp.*) (Fromard et al., 1998; Imbert & Rollet, 1989). For the other species (0.7% of all stems), we used a general equation for mangroves (Komiyama et al., 2005). To estimate belowground biomass (BGB), the equation of Komiyama et al. (2005) was used (see Table A1 in Appendix for a list of all equations used). Following Kauffman & Donato (2012), aboveground carbon (AGC) was calculated as $0.5 \cdot \text{AGB}$, and belowground carbon (BGC) as $0.39 \cdot \text{BGB}$. As outlined in Kauffman & Donato (2012), soil carbon concentration per depth interval was calculated as: soil carbon (SOC) [Mg C ha^{-1}] = bulk density [g cm^{-3}] * soil depth interval [cm] * % C. Subsequently, total soil carbon was attained by summing the masses of sampled depth for each location.

Size class distributions (SCDs) at 5 cm diameter increments were created for the most abundant tree species at different degradation levels. Moreover, a simple linear regression was conducted using the natural logarithm of size classes and observations falling into the respective size class. Slope, coefficient of determination R^2 , and significance level were calculated and formed the bases for assessing whether a species showed continuous recruitment (see Young et al. (2017)). A resulting negative slope points towards good

recruitment, whereas a positive slope (more individuals in upper size classes) indicates impaired recruitment (Obiri et al., 2002).

Statistical analyses were carried out using R studio (version 4.2.1). The Kruskal-Wallis test was used to determine significant differences ($p < 0.05$) between degradation levels both regarding forest structure and carbon stocks. Post-hoc pairwise comparisons of two degradation levels at a time were performed using Dunn's test.

3 Results

3.1 Forest Structure

Basal area ($\text{m}^2 \text{ha}^{-1}$) ranged from $26.8 \text{ m}^2 \text{ha}^{-1}$ (P) to $14.5 \text{ m}^2 \text{ha}^{-1}$ (MD) and $1.7 \text{ m}^2 \text{ha}^{-1}$ (HD). Stem density ($\geq 5 \text{ cm}$) ranged from $540 \text{ stems ha}^{-1}$ (P) to $732 \text{ stems ha}^{-1}$ (MD) and $141 \text{ stems ha}^{-1}$ (HD). Stem density of large stems ($\geq 30 \text{ cm}$) ranged from $129 \text{ stems ha}^{-1}$ (P) to 38 stems ha^{-1} (MD) and 3 stems ha^{-1} (HD). Overall, degradation level had a significant effect on most forest structure variables studied (see Table 1).

Table 1: Forest structure compared between degradation levels of mangroves in the Sherbro river estuary, Sierra Leone

Mean diameter at breast height (Dbh) [in cm], mean height [in m], basal area for ≥ 5 cm Dbh, 5– 10 cm Dbh, ≥ 30 cm Dbh, and ≥ 30 cm Dbh [in $\text{m}^2 \text{ha}^{-1}$], stem density for ≥ 5 cm Dbh, 5– 10 cm Dbh, ≥ 10 cm Dbh, and ≥ 30 cm Dbh [in stems ha^{-1}], compared between degradation levels (pristine (P), moderately degraded (MD), heavily degraded (HD))

Different letters within columns mark significant differences at $p < 0.05$ (displayed to the right of respective values to be compared for each row).

Degradation level	Pristine (P)	Moderately degraded (MD)	Heavily degraded (HD)
Number of transects	6	8	5
mean Dbh [cm]	22.5 \pm 4	a 13.7 \pm 3.1	ab 11.6 \pm 7.9
mean height [m]	23.6 \pm 4.4	a 14.2 \pm 2.9	b 9.3 \pm 7.1
basal area [m² ha⁻¹]	26.8 \pm 10.1	a 14.5 \pm 5.2	a 1.7 \pm 0.7
Basal area (5– 10 cm) [m² ha⁻¹]	0.4 \pm 0.2	a 2.1 \pm 1.4	b 0.4 \pm 0.3
Basal area (≥ 10 cm) [m² ha⁻¹]	26.4 \pm 10.2	a 12.7 \pm 5.4	ab 0.9 \pm 1.0
basal area (≥ 30 cm) [m² ha⁻¹]	16.5 \pm 9.7	a 4.5 \pm 4.0	ab 0.4 \pm 0.7
stem density (≥ 5cm) [stems ha⁻¹]	539.2 \pm 129.2	a 731.9 \pm 365.9	a 141 \pm 82.3
stem density (5– 10 cm) [stems ha⁻¹]	94.4 \pm 58.0	a 365 \pm 237.0	b 114 \pm 75.0
stem density (≥ 10 cm) [stems ha⁻¹]	418.1 \pm 149.2	a 386.9 \pm 161.5	a 27 \pm 20.4
stem density (≥ 30 cm) [stems ha⁻¹]	128.6 \pm 71.4	a 38.1 \pm 29.3	ab 3 \pm 6.0

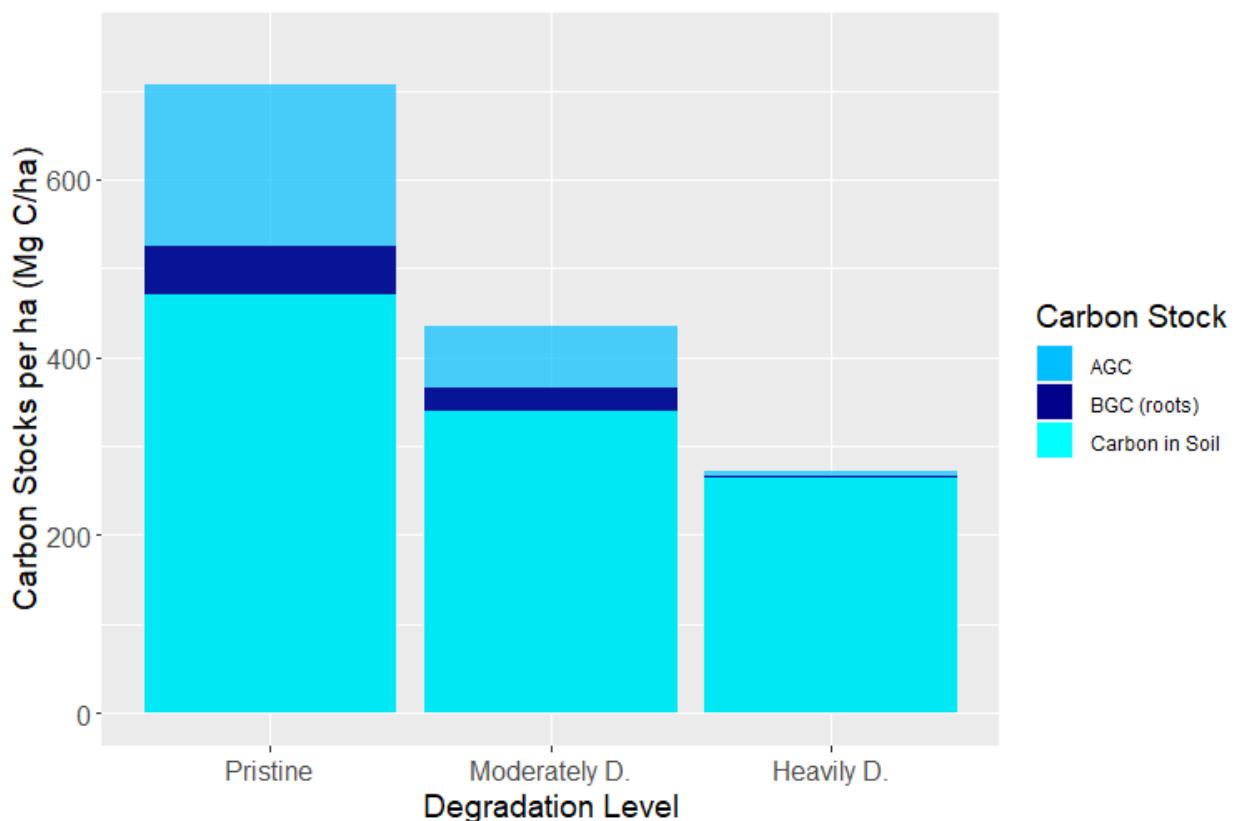
3.2 Carbon Stocks

Total carbon stocks ranged from 707.1 Mg C ha⁻¹ (P) to 435.8 Mg C ha⁻¹ (MD) and 271.9 Mg C ha⁻¹ (HD). AGC ranged from 182.0 Mg C ha⁻¹ in P transects to 70.0 Mg C ha⁻¹ in MD transects and 5.8 Mg C ha⁻¹ in HD transects. Stems \geq 10 cm Dbh accounted for most of AGC (Table 2). Belowground carbon (BGC) decreased from 55.1 Mg C ha⁻¹ (P) to 25.7 Mg C ha⁻¹ (MD) and 2.2 Mg C ha⁻¹ (HD). SOC decreased from 470.0 Mg C ha⁻¹ (P) to 340.1 Mg C ha⁻¹ (MD) and 263.9 Mg C ha⁻¹ (HD). Degradation had a significant effect ($p < 0.05$) on AGC and BGC but did not have a significant effect for SOC (Table 2).

Averaged across all transects, carbon stored in soils accounted for the largest share of total carbon stocks (75.2%) followed by AGC (18.7%) and BGC (6.1%). The share of carbon stored in soils increased significantly with increasing degradation, while shares of AGC and BGC decreased (Fig. 2a, Table 2).

When comparing tree carbon stocks of pristine mangroves at different locations (Fig. 2b), an interior transect had the highest (Bangruwa, >15km from ocean, AGC+BGC: 415.2 Mg C ha⁻¹), followed by an oceanic transect (321.6 Mg C ha⁻¹ BC_P2, Bendu Cha) and an interior one (206.8 Mg C ha⁻¹) (S_P2, Sittia). Riverine transects in Imperi exhibited lowest carbon stocks (Fig. 2b).

a)



b)

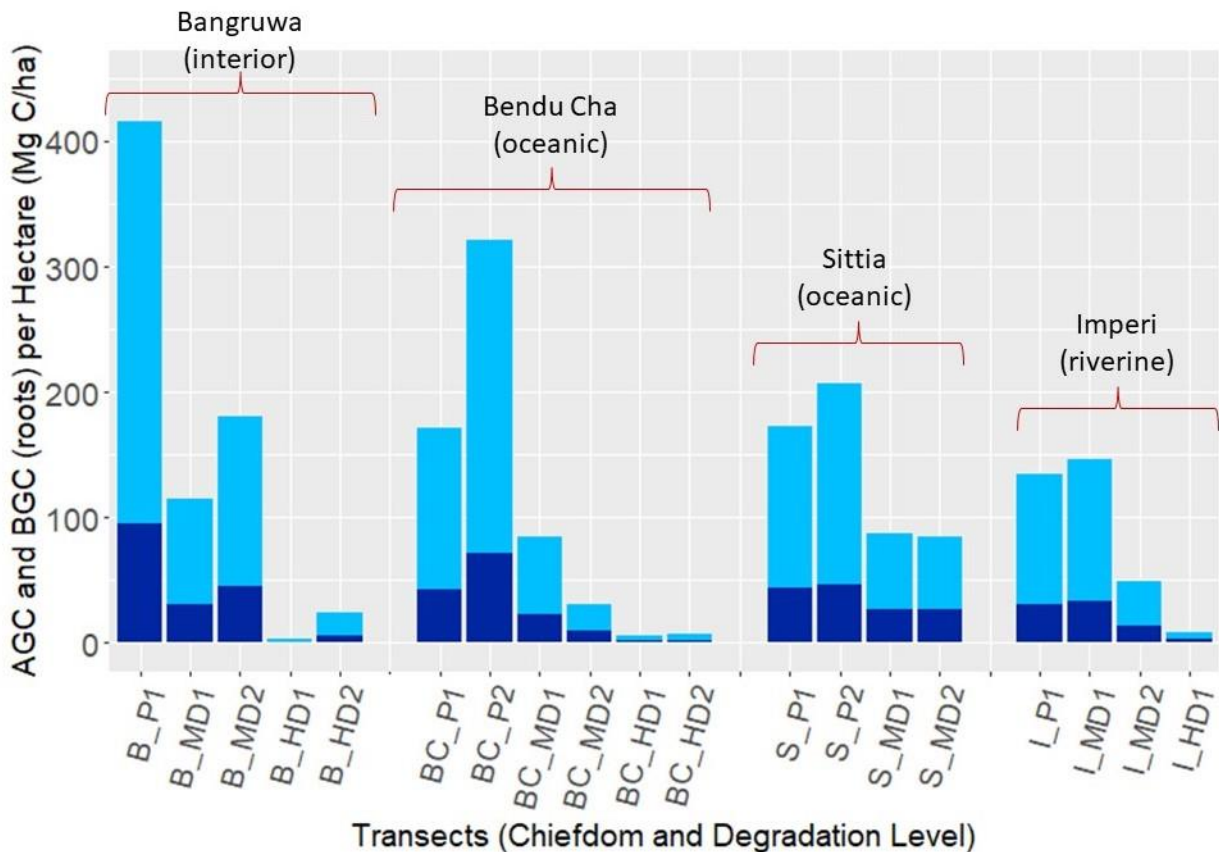


Figure 2a (top): Carbon stocks [in Mg C ha⁻¹] for different degradation levels of mangroves in the Sherbro river estuary, Sierra Leone

Colours indicate different carbon pools (from top: aboveground carbon (AGC), belowground carbon of roots (BGC), and carbon in soil (SOC))

Figure 2b (bottom): Comparison of aboveground carbon (AGC) (blue, top) and belowground carbon (BGC) (dark blue, bottom) between mangroves at different degradation levels and in different chiefdoms in the Sherbro river estuary, Sierra Leone [carbon pools in Mg C ha⁻¹]

Abbreviations of transects as combination of chiefdom (Bangruwa (B), Bendu Cha (BC), Imperi (I) and Sittia (S) and degradation levels (pristine (P), moderately degraded (MD) and heavily degraded (HD))

Table 2: Carbon stocks compared between degradation levels of mangroves in the Sherbro river estuary, Sierra Leone

Aboveground carbon (AGC) for ≥ 5 cm Dbh, 5–10 cm Dbh, and ≥ 10 cm Dbh [in Mg C ha⁻¹], belowground carbon (BGC) [in Mg C ha⁻¹], soil organic carbon (SOC) [in Mg C ha⁻¹], total carbon [in Mg C ha⁻¹], and proportion of soil organic carbon from total carbon [in %] compared between degradation levels (pristine (P), moderately degraded (MD), heavily degraded (HD)) Different letters within columns mark significant differences at $p < 0.05$ (displayed to the right of respective values, to be compared for each row).

Degradation level	Pristine (P)		Moderately degraded (MD)		Heavily degraded (HD)	
Number of transects	6		8		5	
AGC (≥ 5 cm) [Mg C ha⁻¹]	182.0 \pm 77.3	a	70.0 \pm 37.4	b	5.8 \pm 6.6	b
AGC (5–10 cm) [Mg C ha⁻¹]	1.1 \pm 0.6	a	4.2 \pm 2.8	b	1.2 \pm 0.9	a
AGC (≥ 10 cm) [Mg C ha⁻¹]	180.9 \pm 77.5	a	65.7 \pm 36.2	a	4.7 \pm 7.0	b
BGC [Mg C ha⁻¹]	55.1 \pm 21.7	a	25.7 \pm 11.2	ab	2.2 \pm 1.8	b
SOC [Mg C ha⁻¹]	470.0 \pm 105.7	a	340.1 \pm 95.3	a	263.9 \pm 119.3	a
Total Carbon [Mg C ha⁻¹]	707.1 \pm 87.8	a	435.8 \pm 99.4	ab	271.9 \pm 121.3	b
SOC/Total Carbon [%]	65.6 \pm 8.0	a	77.1 \pm 8.3	ab	97.5 \pm 1.6	b

3.3 Species Composition

All stems measured belonged to three families: *Rhizophoraceae* (86.5%), *Acanthaceae* (12.8%), and *Combretaceae* (0.7%), and to six species: *Rhizophora racemosa* (77.0%), *Rhizophora mangle* (8.4%), *Rhizophora harrisonii* (1.1%), *Avicennia africana* (11.8%), *Avicennia germinans* (1.0%), and *Conocarpus erectus* (0.7%). All species were listed as least concern according to the IUCN Red List. Species diversity ranged from 1 to 4 tree species per transect.

Overall, we observed differences in species composition related to both location and degradation level (see Fig. 3). P transects across all locations were dominated by *Rhizophora racemosa*. MD transects were also dominated by *Rhizophora racemosa* in Imperi (riverine) and Bangruwa (interior), while *Avicennia africana* dominated in Sittia (oceanic on islands), and some variation was observed in MD transects in Bendu Cha (oceanic) (Fig. 3). *Rhizophora racemosa* was not found in HD transect in any location. Instead, *Rhizophora mangle* dominated in HD together with *Rhizophora harrisonii*, *Avicennia germinans* or *Conocarpus erectus* (Fig. 3).

Some variation in species composition could be explained by location, e.g. *Avicennia* being most abundant in Sittia (oceanic on small islands). However, the changes in the dominant species from *Rhizophora racemosa* to *Rhizophora mangle* could not be explained by location as in Imperi, Bangruwa and Bendu Cha transects dominated by *Rhizophora racemosa* and *Rhizophora mangle* were located in immediate proximity to each other (Fig. 3).

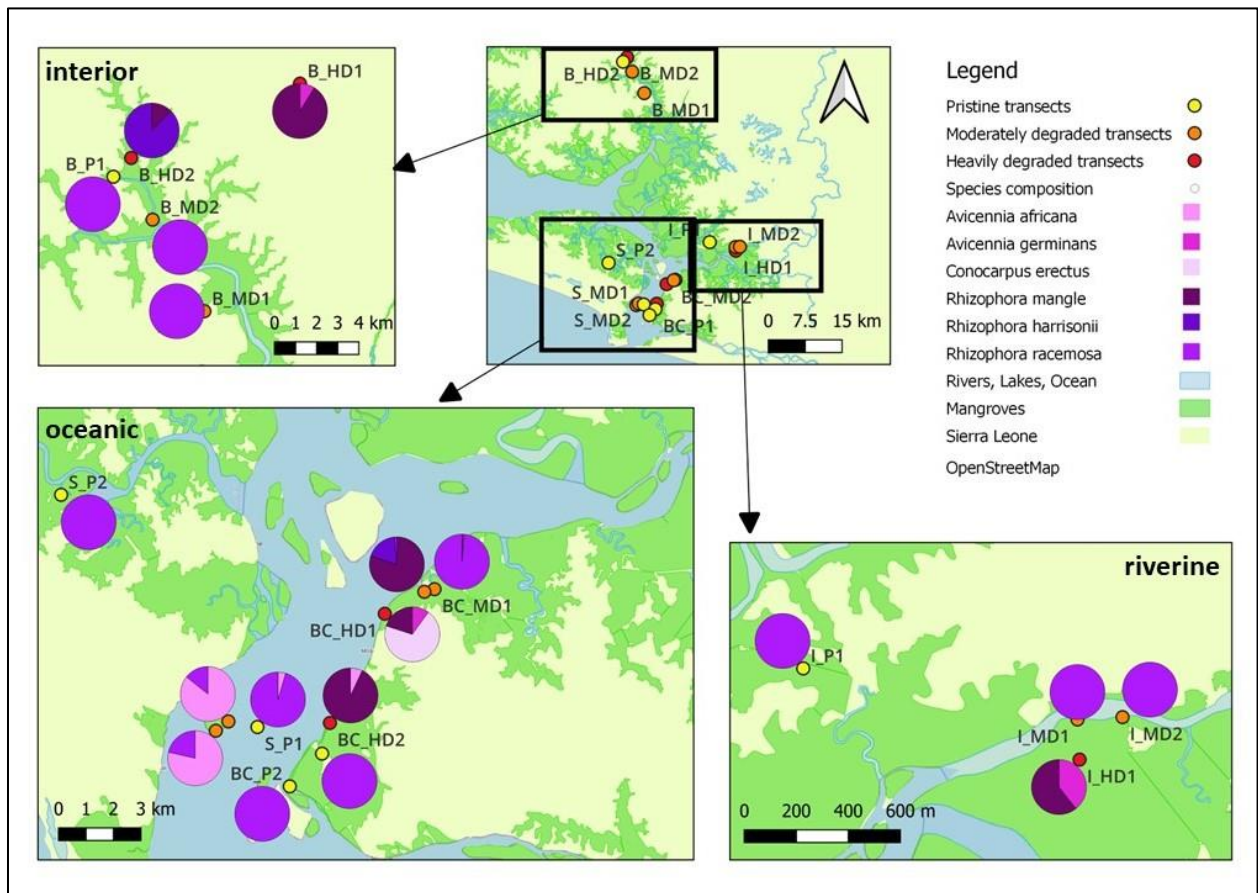
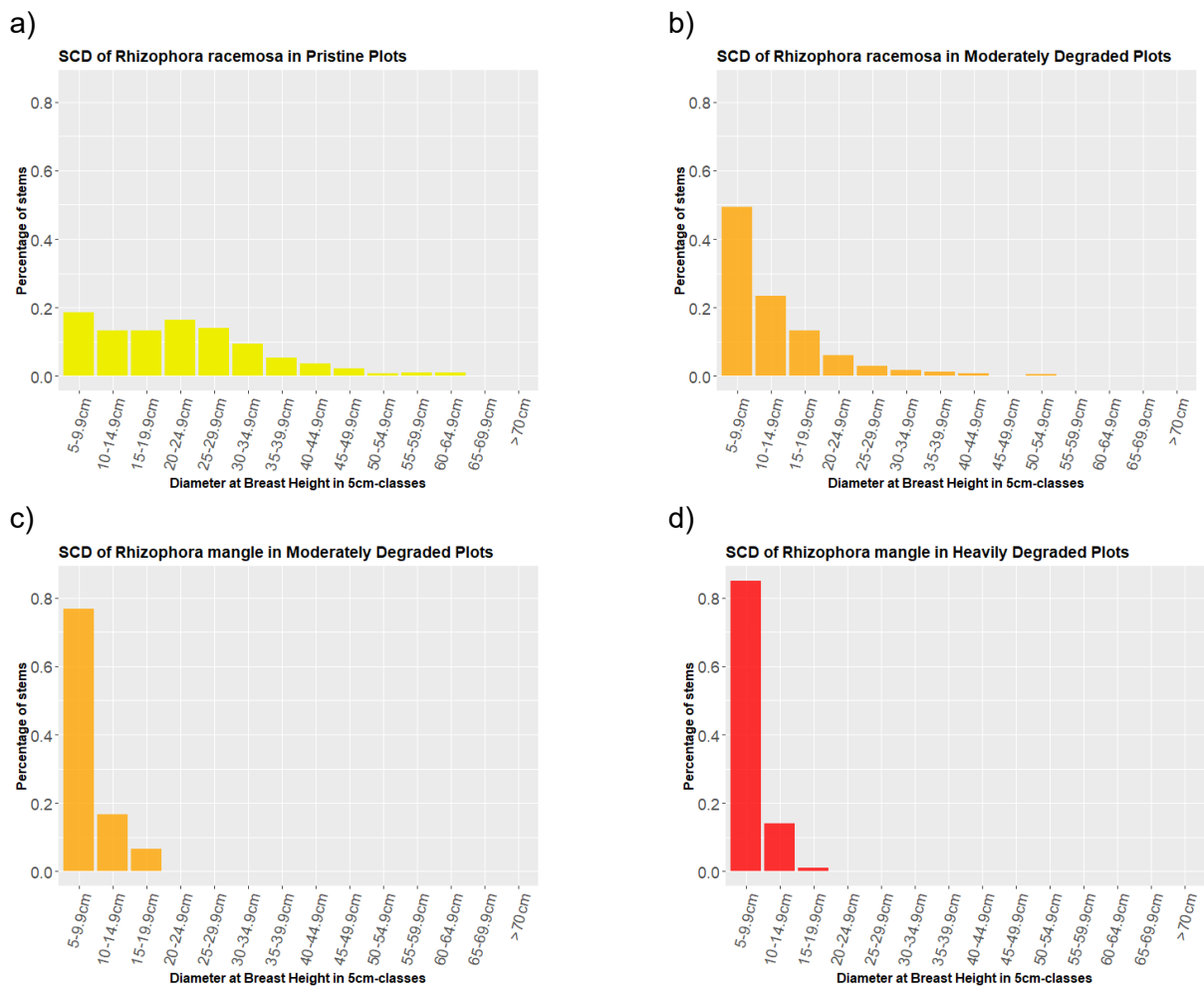


Figure 3: Species composition of transects in the Sherbro river estuary in Sierra Leone (species abundances shown as proportions in pie chart for each transect) Abbreviations of transects as combination of chiefdom (Bangruwa (B), Bendu Cha (BC), Imperi (I) and Sittia (S) and degradation levels (pristine (P), moderately degraded (MD), and heavily degraded (HD)) (map data sources: OpenStreetMap, species: own investigations)

3.4 Population Structures

For *Rhizophora racemosa*, the size class distributions (SCDs) showed negative slopes in both P and MD transects (Fig 4a, b), but fewer small stems were observed in P than in MD transects. This suggests that regeneration may be affected by light requirements and that prevalent uses for this species (fish smoking) target predominantly larger size classes.

For *Rhizophora mangle*, the SCDs showed negative slopes in both MD and HD transects with very similar distributions in both degradation categories (Fig 4c, d). For *Avicennia africana* in MD transects the SCD also showed a negative slope (Fig 4e). In all cases, SCDs could be classified as reverse J-shapes and continuous recruitment (according to (Young et al., 2017)) (Appendix Table B1).



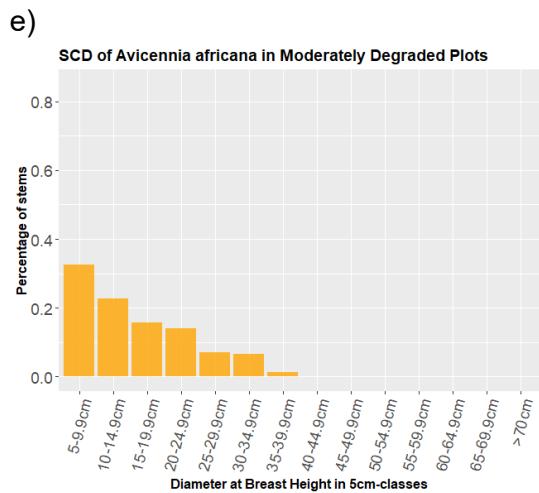


Figure 4: Size class distributions (SCD) of most abundant species at different degradation levels of mangroves in the Sherbro river estuary, Sierra Leone

- a) SCD of *Rhizophora racemosa* in pristine transects
- b) SCD of *Rhizophora racemosa* in moderately degraded transects
- c) SCD of *Rhizophora mangle* in moderately degraded transects
- d) SCD of *Rhizophora mangle* in heavily degraded transects
- e) SCD of *Avicennia africana* in moderately degraded transects

4 Discussion

4.1 Effects of Human-Driven Degradation on Forest Structure

First, it should be noted that the pristine transects sampled in the SRE have many large trees with some of the highest basal areas reported on the African continent (higher than e.g. in Cameroon, Gabon, RoC, DRC (Ajonina et al., 2014a), Senegal, Liberia (Kauffman & Bhomia, 2017) or Tanzania (Njana et al., 2018)).

In our study, human-driven degradation significantly reduced basal area and density of large stems (≥ 30 cm). When stem density of all stems (≥ 5 cm) was considered, degradation increased stem density in MD, and decreased it in HD. The trend of decreasing number of large trees (≥ 30 cm) with increasing degradation levels aligns with findings from other studies, e.g. in Central Africa (Cameroon, Gabon, RoC, DRC) (Ajonina et al., 2014a) or Kenya (Kihia, 2014).

Comparisons of stem density to the literature is challenging due to ambiguities in thresholds used in previous studies. Therefore, we highlight the importance of stating the diameter threshold used and to preferably report estimates for different diameter thresholds separately (e.g. ≥ 5 cm, ≥ 10 cm, ≥ 30 cm Dbh), as suggested by de Jong Cleyndert et al. (2020). The issue of diverging diameter thresholds for reporting stem density is particularly challenging for study comparison across locations. For instance, a study by Ajonina et al. (2014a) from Cameroon, Gabon, RoC, and DRC included all stems > 1 cm Dbh and reported values ranging

from 450 stems ha⁻¹ to 3,256 stems ha⁻¹ for mangroves of different degradation intensities (from heavily degraded to undisturbed). A study from Tanzania by de Jong Cleyndert et al. (2020) reported stem density of 242.5 stems ha⁻¹ for 5–10 cm, 517.8 stems ha⁻¹ for stems ≥10 cm and 1.3 stems ha⁻¹ for ≥30 cm Dbh. Mangroves in that study had a history of exploitation and continued to be used e.g. for construction and firewood. While we report more large trees and fewer small trees in the SRE, their results most resemble the values found in our study for MD transects for 5–10 cm and ≥10 cm Dbh, which matches the similar usage pressures.

4.2 Effects of Human-Driven Degradation on Carbon Stocks

Total carbon stocks ranged from 707.1 Mg C ha⁻¹ (P) to 435.8 Mg C ha⁻¹ (MD) and 271.9 Mg C ha⁻¹ (HD) on average. However, in some locations, e.g. pristine interior transects in Bangruwa, AGC+BGC amounted to more than twice the average of all pristine transects.

Carbon stocks in pristine transects in the SRE (with soils sampled up to 1m depth) are lower than the global mean of 865 Mg C ha⁻¹ estimated by Kauffman et al. (2020). However, their estimate was derived from 190 sites across five continents with a greater average soil depths (216 cm) and 35% of sites > 300 cm. Other authors reported similar values to ours: Alongi (2020) estimated global carbon stocks in mangroves to be around 738.9 Mg C ha⁻¹ drawing on measurements from 52 countries worldwide that also included soils > 1 m depth. In this latter study, estimates for SOC are higher than in our study, whereas AGC is lower (109.3 Mg C ha⁻¹ in the study by Alongi (2020) and 182 Mg C ha⁻¹ (P) in ours). As both global estimates considered SOC up to greater depths than our study, our results could underestimate total carbon stocks. In case greater soil depth was considered, carbon stocks of pristine mangroves in the SRE would most likely exceed global averages.

Aboveground carbon stocks found in pristine transects in our study exceed those reported by many studies in East African countries such as Cohen et al. (2013) or Gress et al. (2017) in Kenya, Fatoyinbo & Simard (2013), Siteo et al. (2014) or Stringer et al. (2015) in Mozambique and Njana et al. (2018) or de Jong Cleyndert et al. (2020) in Tanzania. When compared to carbon stocks found in West Africa, findings from our study exceed estimates by Ajonina et al. (2014a) for Gabon, Carreiras et al. (2012) for Guinea Bissau, Egnankou et al. (2022) for Côte d'Ivoire, Alongi (2020) for Senegal, Togo, Ghana, Guinea, and Sierra Leone, Kauffman & Bhomia (2017) for Liberia and Senegal but are lower than estimates by Ajonina et al. (2014a) for Cameroon and Ghana.

Ratios between carbon pools were found to be on average 75% SOC, 19% AGC and 6% BGC in our study. This aligns with typical ratios for mangrove forests like reported by Alongi (2020). Reasons for large SOC pools in mangrove ecosystems relate to high sedimentation

rates, importance of soils as nutrient reservoirs and for stabilisation, anoxic soils, and therefore slow decomposition (Alongi, 2014; Kathiresan, 2021).

Regarding location, high total carbon stocks of interior mangroves were observed in a study in Vietnam (Dung et al., 2016). The trend of increasing AGC with increasing seaward distance could be partly confirmed in this study, at least regarding high carbon stocks in Bangruwa (interior). In some studies, riverine sites were also reported as exhibiting higher aboveground carbon than oceanic sites (Donato et al., 2011; Kauffman et al., 2020). This could not be confirmed here. However, as transects in Imperi (riverine) were most affected by degradation due to vicinity to roads and villages, results may be distorted by that. A study of mangroves in Tanzania found a significant correlation of decreasing SOC and associated total carbon stocks with increasing seaward distance (de Jong Cleyndert et al., 2020). This was explained by areas closer to the shore experiencing greater tidal inundation, higher salinity levels, slower decomposition, and therefore higher SOC (de Jong Cleyndert et al., 2020). This trend could not be confirmed here. However, in the study by de Jong Cleyndert et al. (2020), transects were located at least 4 km away from the shoreline, so soil erosion was not an issue but likely plays a role closer to the shoreline as in transects measured in our study.

Concerning effects of human-driven degradation on different carbon pools, we found declines in AGC and BGC to exceed those of SOC. AGC decreased on average by 61.5 % from P to MD transects and by 96.8 % to HD ones. BGC decreased slightly less by 53.4 % from P to MD and 96.0 % from P to HD. SOC decreased by 27.6 % from P to MD and 43.8 % from P to HD. This aligns with findings from other studies like for instance Ajonina et al. (2014a), who also reported declines in AGC and BGC to exceed those of SOC though generally the declines reported in their study were lower than ours, especially for SOC. AGC in their study decreased by 87.6% in HD compared to P transects, BGC by 79.5% and SOC by 24.6% (Ajonina et al., 2014a). However, the study design of Ajonina et al. (2014a) differed from ours as a total of 15 regions distributed among four countries was considered rather than comparing different degradation levels within one region. Transforming mangroves to rice farms through burning can also exacerbate carbon loss from soils as was done in HD transects in the SRE. Release of carbon through degradation of mangroves depends mainly on how deeply the soils are affected (Donato et al., 2011). Slower degradation of SOC can be explained by alluvial deposition by rivers, soils receiving carbon input from outside the system (flood water, tides, alluvial deposits) and decomposition over time when formerly anoxic soils are subjected to aerobic decomposition (Ajonina et al., 2014a; Kauffman et al., 2020).

4.3 Effects of Human-Driven Degradation on Species Composition

While location mattered for explaining some variety in species composition, degradation level seemed to be the main explanatory factor driving changes in species compositions, with heavy degradation driving changes from mono specific *Rhizophora racemosa* stands to *Rhizophora mangle* as dominant species in all locations.

Varying species composition in mangroves has mostly been studied in relation to zonation patterns, which are commonly linked to salinity gradients, tidal flushing, and soil type (Ukpong, 1995). *Rhizophora racemosa* commonly dominates mangroves in West Africa (Feka & Ajonina, 2011). While being tolerant of saline conditions (Numbere, 2018), this species is preferably found on sites of intermediate or low salinity (Duke et al., 1998) along rivers and streams especially in tidal estuaries, where it forms pure stands (Ajonina et al., 2018). Species distributions observed in our study align with findings from Ghana, where *Avicennia* tended to dominate in areas of high salinity, like lagoons, whereas *Rhizophora racemosa* was absent from these sites, but was found at less saline locations (Nunoo & Agyekumhene, 2022). In a study on the Aberdeen creek mangroves in Sierra Leone (located about 130 km north from SRE), *Rhizophora racemosa* was only found on sites without human influence (Gledhill, 1963). *Rhizophora mangle* is very salinity tolerant, a typical pioneer species in succession (Ellison, 2021), adapted to sandy soils and drier conditions (Numbere, 2018). On more human influenced sites in the Aberdeen creek mangrove swamp in Sierra Leone mainly *Rhizophora mangle* and *Rhizophora harrisonii* were found (Gledhill, 1963), which aligns with our findings for HD transects.

Consequently, possible explanations for observed changes in species composition are changed abiotic conditions that favour *Rhizophora mangle* as disturbed areas are characterised by more extreme conditions like higher temperatures and irradiation (Carugati et al., 2018). Further reasons relate to natural succession starting in HD transects after they had been transformed to rice fields with *Rhizophora mangle* as pioneer species. Moreover, it plays a role for *Rhizophora racemosa*'s decreased importance and competitive ability that it is the most preferably used species for fish smoking.

There are few studies that focused on how mangrove degradation drives changes in species composition. Despite some studies looking at mangroves affected by different degradation intensities, they did not report any changes in species composition, e.g. (Ajonina et al., 2014a). In a study on mangroves in Gazi Bay, Kenya by Kihia (2014), tree cutting, trampling by man, vehicles, and digging for fish bait by fishermen were the main disturbances. The study in Kenya reported changes in dominance and abundance of species comparing disturbed and undisturbed sites. However, differently to our study, all six recorded species were present in both disturbed and undisturbed sites (Kihia, 2014).

Yudha et al. (2021) also found logging to impact species composition in mangrove forests in West Papua, Indonesia. However, in this case, rather than changing species compositions or replacement of dominating species, the occurring species remained the same in primary and secondary forests and the dominant species (*Rhizophora apiculata*) increased in dominance after logging (from 46% to 69%), while the shares of co-dominant species decreased in secondary forests despite *Rhizophora apiculata* being one of the targeted commercial species.

To put it briefly, degradation seems to be the main factor behind observed changes from *Rhizophora racemosa* to *Rhizophora mangle* as dominating species. Our findings stand out by documenting fundamental changes in species compositions from one mono-dominant species to entirely different dominating species rather than merely changed dominances and abundances within the same community of species.

4.4 Effects of Human-Driven Degradation on Population Structures

Size class distributions (SCDs) of the three most abundant species showed negative slopes and were classified as having continuous recruitment for all degradation levels. Fewer large stems and more small stems were observed in degraded transects. Different drivers of degradation (wood extraction in MD transects, and conversion to rice farms in HD transects) had similar effects on population structures when comparing the same species. However, species compositions differed between degradation levels (see 3.3 and 4.3).

Other studies reported that various uses that target particular stem sizes, affect SCDs differently as e.g. species used for firewood exhibit fewer stems in smaller size classes, whereas species used for construction are characterised by a lack of large stems (Batumike et al., 2022; Obiri et al., 2002). Our findings correspond to studies by Ajonina et al. (2014a), who looked at mangroves of different degradation levels (as increasing intensity of wood extraction) in Cameroon, Gabon, RoC, and DRC. They also report reverse J-shaped curves for P, MD, and HD transects with HD transects exhibiting a particularly large number of small stems and no large stems (≥ 30 cm Dbh). Another study from the Greater Amanzule mangroves in Ghana showed similar findings to ours and additionally measured seedlings (0–1 cm Dbh) and saplings (1–5 cm Dbh) (Ajonina et al., 2014). They reported reverse-J shaped distributions for both degraded and intact mangroves. While degraded ones exhibited many seedlings of 0–1 cm Dbh and some stems up to 10 cm Dbh, pristine ones were characterised by fewer seedlings but more saplings and trees up to 50 cm Dbh.

The continuous recruitment visible in the SCDs of species analysed in the SRE generally points towards good regeneration capacities and potential for passive restoration if degradation pressures are removed. However, changes in species composition as observed

from MD to HD transects may impede passive restoration of HD transects back to pristine-like states with *Rhizophora racemosa* as dominant species.

4.5 Study Limitations

Our study has some limitations. First, only four out of ten chiefdoms in the SRE area were sampled due to accessibility and time constraints. However, they covered a range of locations within the SRE. Although chiefdoms closest to roads and villages are likely to be those in which degradation is most prevalent, future work should consider other chiefdoms. Second, the classification of transects into only three use or degradation levels could be improved by differentiating a larger number of degradation levels. Third, there are limitations of inferring population structures from size class distributions since static information about size class distributions may not provide a good picture of future population trends (Condit et al., 1998). Still, the relative proportion of small vs. large stems in a population can give an idea of the direction of future population change if it is applied to intraspecific differences and at a local level (Feeley et al., 2007) as it was done in this study. Additionally, measuring all stems ≥ 5 cm, instead of ≥ 10 cm as in many other studies, also provides a more accurate assessment of population dynamics as potential impairments of recruitment already become visible for stems < 10 cm. Additionally, counting seedlings in smaller sub-plots as done or recommended e.g. by Ajonina et al. (2014a) could have provided an even more accurate picture.

4.6 Management Implications

Our findings highlight that mangroves in the SRE store important quantities of carbon, in both living trees and soils. Total carbon stocks were higher than in many other mangrove forests in Africa and could even exceed global averages if SOC at greater depth (> 1 m) was considered. As demonstrated in our study, human activities have significant impacts on these carbon stocks. As there is considerable international focus on avoiding degradation of carbon-rich ecosystems, mangroves can play an important role if they are not degraded. In addition, size and spatial arrangement of remaining mangrove patches matter for regeneration capacity (Bryan-Brown et al., 2020), especially when considering changing species compositions and succession dynamics. The negative effects of current human usage pressures are likely to be exacerbated by climate change impacts, especially sea level rise (Gilman et al., 2008; Goussard & Ducrocq, 2014). Considering these pressures and the dependence of surrounding communities on the multiple ecosystem services provided by mangroves in the SRE, more efforts should be made to increase both the conservation of remaining pristine mangroves, and the restoration of degraded ones, particularly MD ones.

5 Conclusion

This study assessed impacts of human-driven degradation on mangrove forest structure, carbon stocks, and species composition. Degradation had a significant effect on forest structure with declining basal area, fewer large trees, and higher stem density of small trees, with increasing degradation level. Notably, AGC of pristine mangroves was higher than in many other mangrove forests in Africa. All carbon pools (AGC, BGC, and SOC) decreased with increasing degradation, limiting the potential for climate change mitigation. Results also showed that heavy degradation resulted in a change in the dominating species from *Rhizophora racemosa* to *Rhizophora mangle*. Intact, structurally complex, old mangrove stands of *Rhizophora racemosa* provide valuable habitat for birds, marine and terrestrial organisms (UNEP, 2023) as well as coastal protection – and have higher carbon stocks as our study demonstrated. Overall, this study showed how human-driven degradation affects forest structure, carbon stocks, species composition, and the population structures of dominant species. This can be built on by further studies in the region and can provide a basis for developing management, restoration, and conservation projects, which are ecologically informed and effective.

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Appendix

Appendix A: Equations Used for Calculating Aboveground and Belowground Biomass

Mangrove biomass is more species-dependent than site-dependent (Komiyama et al., 2008). Therefore, species-specific equations were used to calculate aboveground biomass (AGB). To attain AGB per hectare for each transect, results were summed according to species composition.

Table A1: Equations used for calculating aboveground (AGB) and belowground biomass (BGB)

Equation, species this equation was used for, source, origin of equation (either region/country in which it was developed or review of existing studies on mangrove biomass)

	Equation	Species	Source	Origin
AGB	$AGB = 0.140 * Dbh^{2.4}$	<i>Avicennia germinans, Avicennia africana</i>	Fromard et al. (1998)	French Guiana Guadeloupe,
	$AGB = 0.178 * Dbh^{2.47}$	<i>Rhizophora mangle</i>	Imbert & Rollet (1989)	Antilles francaises
	$AGB = 0.128 * Dbh^{2.6}$	<i>Rhizophora racemosa, Rhizophora harrisonii</i>	Fromard et al. (1998)	French Guiana
	$AGB = 0.251 * WSG * Dbh^{2.46}$	<i>Conocarpus erectus</i>	Komiyama et al. (2005)	review
BGB	$BGB = 0.199 * WSG^{0.899} * Dbh^{2.22}$	all	Komiyama et al. (2005)	review

Appendix B: Results of Calculations for Size Class Distributions of Most Abundant Species at Different Degradation Levels

Table B1: Analysis of size class distributions of most abundant species at different degradation levels

Species analysed, degradation level, chiefdom, number of observations falling into the respective analysis unit, maximum diameter at breast height (max Dbh) [in cm] resulting slopes, R² and p-values

Species	Degradation level	Chiefdom	Observations	max Dbh (cm)	Slope	R ²	p-value
<i>Rhizophora racemosa</i>	Pristine	all	550	69.6	-0.43	0.72	< 0.001
<i>Rhizophora racemosa</i>	Moderately degraded	all	912	72.4	-0.36	0.95	< 0.001
<i>Rhizophora mangle</i>	Moderately degraded	all	60	19.0	-0.37	0.99	0.033
<i>Avicennia africana</i>	Moderately degraded	Sittia	215	39.0	-0.53	0.79	0.007
<i>Rhizophora mangle</i>	Heavily degraded	all	99	18.5	-0.22	0.96	0.103



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