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ORIGINAL RESEARCH ARTICLE

Evaluation of carbon stock in the sediment of two mangrove species, *Avicennia marina* and *Rhizophora mucronata*, growing in the Farasan Islands, Saudi Arabia

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Summary The aim of this study was to conduct the first comprehensive evaluation of carbon stock in the sediments of *Avicennia marina* (black mangrove) and *Rhizophora mucronata* (red mangrove) along the coastline of an arid region (Farasan Islands, Saudi Arabia). Such information is necessary for the development of any management plan for the mangrove ecosystems along the Saudi Red Sea islands and provide a rationale for the restoration of mangrove forests in Saudi Arabia. *A. marina* and *R. mucronata* locations showed significant ($P < 0.001$) differences in sediment bulk density (SBD) and sediment organic carbon (SOC) concentration with higher mean values for both in the sediments of *A. marina*. Considering the whole depth of

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sediment sampled (0—50 cm), the highest value of SOC stock (12.3 kg C m^{-2}) was recorded at *A. marina* locations and the lowest (10.8 kg C m^{-2}) at *R. mucronata* locations. Thus, the SOC stock of *A. marina* was greater than that of *R. mucronata* by 114.3%. Consequently, considering the rate of carbon sequestration and the area of mangrove forests (216.4 ha), the total carbon sequestration potential of mangroves in the Farasan Islands ranged between 10.3 Mg C yr^{-1} and 11.8 Mg C yr^{-1} for *R. mucronata* and *A. marina* locations, respectively. Thus, it is necessary to protect and restore these ecosystems for the sequestration of carbon and for their other valuable ecosystem services.

1. Introduction

Climate change is a major global issue (Eid et al., 2016), and CO_2, one of the greenhouse gases, has a major role in climate change and global warming (Shaltout et al., 2019). Land use changes (such as deforestation) and combustion of fossil fuels are thought to lead to high concentrations of CO_2 in the atmosphere (IPCC, 2007). The highest levels of carbon in the terrestrial carbon cycle are found in soil reservoirs (Sahu and Kathiresan, 2019). These levels are twice more than those in the atmosphere and three times more than those in vegetation (Sahu and Kathiresan, 2019). Thus, many scientists have indicated that CO_2 sequestration in soil organic carbon (SOC) may contribute substantially to reducing the effects of climate change (see Taillardat et al., 2018). Furthermore, increasing soil carbon stocks and protecting carbon rich soils are essential for reaching the climate targets of the Paris Climate Agreement (Rumpel et al., 2018).

Blue carbon has received international interest for its potential role in the mitigation of CO_2 emissions (Taillardat et al., 2018). Mangrove forests constitute one of the ecosystems with the greatest density of carbon in the world with most of the carbon stored in the soil, and organic-rich sediments of several metres depth have been observed in some mangrove ecosystems (Twilley et al., 1992). Therefore, mangroves are now being considered an important means of combatting climate change through approaches such as REDD+ and blue carbon (Alongi, 2012). Sanderman et al. (2018) described the need to record the distribution of carbon stock in mangrove forests so that they can be utilised in attempts to ease climate change. Therefore, accurate, location-specific quantification of carbon stocks is essential to understand the spatial range of coastal blue carbon and to estimate future carbon stocks as a result of habitat preservation, loss, degradation, or restoration (Radabaugh et al., 2018).

Mangrove forests are efficient ecosystems existing at the land—sea margin. Globally, these forests comprise approximately fifty-four species from twenty families of vascular plants (Tomlinson, 2016). Alongi (2012) described them as the sole woody halophytes existing in saltwater along tropical and subtropical coastal regions. The significance of mangrove forests in terms of social, economic, and ecological benefits are well documented worldwide as being crucial for both the wellbeing of humans and the environment (Shaltout et al., 2019). Mangroves aid in stabilising shorelines and decreasing the effects of natural disasters, including those of tsunamis and hurricanes. Mangrove forests are one of the coastal vegetation ecosystems that raise the seabed via soil accretion of autochthonous and allochthonous material, supporting a natural coastline protection system against sea level rise (Saderne et al., 2018). They serve as nursing grounds for marine biota, conserve biodiversity, and provide fuel, medicine, food, and material for construction to coastal communities. They also sequester and store large quantities of carbon, which help in the mitigation of climate change and enrich the ocean by providing organic inputs (Jennerjahn and Ittekot, 2002). The mean economic value assigned to mangroves has been approximated as second only to the value of estuaries and seagrass meadows and greater than the economic value of coral reefs, continental shelves, and the open sea (Costanza et al., 1997).

Mangroves cover approximately 137,600 km^2 of coastline in 118 countries with 38.7% found in Asia, 20.3% in Latin America and the Caribbean, 20% in Africa, 11.9% in Oceania, 8.4% in North America, and 0.7% in the European Overseas Territories (Bunting et al., 2018). The net primary production of carbon is formed partially (10%) from mangrove forests and these forests also result in a quarter of carbon burial in coastal zones worldwide — this, despite existing only along 0.7% of the coastal zones worldwide (Alongi, 2007). Donato et al. (2011) indicated that between 50% and 90% of the total carbon of the mangrove ecosystem is made up of SOC. Twilley et al. (2018) supported this by describing how mangroves comprise a greater amount of sedimentary carbon per m^2 than terrestrial regions, indicating that within the first metre, 36.1 kg m^{-2} of blue carbon can be found (Sanderman et al., 2018). This amount is 3.3-, 3.2-, 2.8-, 2.7-, 2.4-, 2.3-, 2.1-, 1.8-, 1.7-, 1.5-, and 1.4-fold greater than the soil carbon stocks estimated for closed shrublands, savannas, croplands, deciduous broadleaf forests, evergreen broadleaf forests, grasslands, open shrublands, mixed forests, evergreen needleleaf forests, permanent wetlands, and deciduous needleleaf forests, respectively (Sanderman et al., 2018).

Even though mangrove ecosystems are advantageous in so many ways, both changes in land use and growth of urban regions are threatening their survival. For instance, one study determined that the mass deforestation of 278,049 ha of a mangrove area (equalling 1.67% of the total area worldwide) occurred in 2000 and this led to between 30.4 and 122.0 Tg C (111.0—447.0 Tg CO_2) of committed emissions (Sanderman et al., 2018). Thus, the change in land use between 2000 and 2015 led to emissions of somewhere in this range from the soils of these mangrove forests. Considering the current rate of loss, Duke et al. (2007) estimated that all mangroves would disappear within a decade. Thus, understanding the distribution of SOC stock in mangrove forests is very important for prioritising protection and restoration efforts for climate change mitigation (Sanderman et al., 2018).

The location of the Red Sea mangroves, at the most northern point of their spread in the Indo-Pacific, confers them with great importance in terms of their ecology and biogeography (PERSGA, 2004). Saifullah (1994) demonstrated the significance of this mangrove owing to its growth in an area that should be considered uninhabitable in that it has a paucity of rain, the land is dry, rivers are absent, and salt levels are high. Hard-bottomed and euhaline—metahaline mangroves are the two forms of these forests present in the Red Sea (Price et al., 1987). The common species, *Avicennia marina,* has been the subject of various studies performed on mangrove forests in the Saudi Arabian Red Sea to determine their carbon sequestration potential (CSP). These studies were conducted at several locations with the majority being in the southern regions, two in the central areas, and one in a northern area. The efficacy of *A. marina* in carbon sequestration was examined by Eid et al. (2016) along the Saudi Arabian coast of the Red Sea. The central coast of the Red Sea in the Saudi Arabian region was similarly investigated for its carbon sequestration rate (CSR) and SOC stocks by Almahasheer et al. (2017). The southern Red Sea coast of Saudi Arabia was studied by Arshad et al. (2018), and the CSR in regions determined to be polluted and non-polluted was evaluated. *A. marina* was also studied in the southern area of the Red Sea in Saudi Arabia next to Jizan city by Eid et al. (2019) who sought to determine how differences in land use could affect the SOC stock in the sediments of this southern coastal region that was converted from areas flourishing with *A. marina* to shrimp farms. Another study examined the variation in the carbon sequestration capacity of *A. marina* with variation in the accessibility to nutrients and along a salinity gradient along the Saudi Arabian Red Sea (Shaltout et al., 2019).

To the best of our knowledge, this is the first study to compare *A. marina* and *Rhizophora mucronata* in Saudi Arabia with respect to sediment bulk density (SBD), SOC concentration, and SOC stock. Our null hypothesis is that both *A. marina* and *R. mucronata* locations have the same SBD, SOC concentration, and SOC stock. Through this study, we will be able to enhance our understanding of the part played by arid mangroves in sequestering carbon. Our specific research questions were what are the carbon stocks of the mangroves of the Farasan Islands? and how do they differ between *A. marina* and *R. mucronata* locations? The results from this study are expected to increase knowledge on the carbon stock in mangroves and to inform current discussions on blue carbon. This will be useful to government departments regarding the management of mangrove forests and will also improve the attitude and increase the concern of societies towards the conservation of mangroves.

2. Material and methods

2.1. Study area

The Farasan Islands were declared a nature reserve in 1989 owing to the presence of the last population of the Arabian Gazelle (Alfarhan et al., 2002). The region is protected and is currently seen as Saudi Arabia's most significant marine nature reserve. Alfarhan et al. (2002) also described other crucial needs of this region, including as a nesting area for migrating birds and as an area that is conducive to supporting various endemic snakes. Al Mutairi et al. (2012) described the high biodiversity of the region and the presence of various species that are either endemic or endangered and this was attributed to the exclusive location of these islands in the region between the west of Asia and the east of Africa in an arid setting. The fauna and flora of the region are also quite distinct as they are part of various phytogeographical areas including Africa, the Mediterranean, and Asia. The habitats within these islands are also diverse and comprise salt marshes (both wet and dry), sand plains, sand dunes, rocky regions, and mangroves (El-Demerdash, 1996).

The Farasan Islands can be found in the southern area of the Red Sea at a location ca. 50 km away from the axial trough of the Red Sea and approximately 40 km from Jizan City (Fig. 1). The Farasan Islands are a group of 128 islands that together shape the Farasan bank, totalling an area of 3,000 km^2 (Bantan, 1999). Khalil (2012) estimated the complete land area covered by the islands as 600 km^2. Three islands are currently inhabited; namely, Qummah, 14.9 km^2; Sajid, 109 km^2; and Farasan Al-Kabir, 369 km^2. There are other islands that are large in size but remain uninhabited, except by the coast guard — the Disan, Zifaf, and Dumsuk Islands. The other islands are used for picnicking and by fishermen as rest areas. The population of the island is estimated at 4,500 and the majority of the inhabitants are either fishermen or work in agriculture (Alwelaie et al., 1993). An uplifted coral reef is the main material that forms the islands mostly consisting of fossils, with coastal regions comprising eroded coral sands or cliffs and also coral surfaces along the wadis and runnels (Dabbagh et al., 1984). The mean surface level is not greater than 20 m a.s.l. but attains a height of 70 m in certain regions (Alwelaie et al., 1993). The two species investigated in this study, *R. mucronata* and *A. marina*, can be found in the lagoons and the island shorelines (Al-Mutairi et al., 2012). Al-Mutairi et al. (2012) describes *A. marina* as the main species of mangrove established in this region either as a pure community or mixed with *R. mucronata*. Two other studies describe a massive monospecific *R. mucronata* population in the north-eastern region of the Farasan Jetty (Alwelaie et al., 1993; El-Demerdash, 1996).

The climate in the islands is between arid and subtropical. Temperatures of 29.8°C are reached on average with rainfall being less than 130 mm yr^{-1}. Most of the year is humid with high humidity levels of between 70% to 80% in winter and decreasing in summer to between 65% and 78% (El-Demerdash, 1996). The saltiness of the water in the region of the Farasan Islands in the southern area of the Red Sea is also high ranging from ca. 37% in winter to ca. 38% in summer (Bantan, 1999). Abu-Zeid et al. (2011) described how owing to the raised levels of evaporation in the islands'

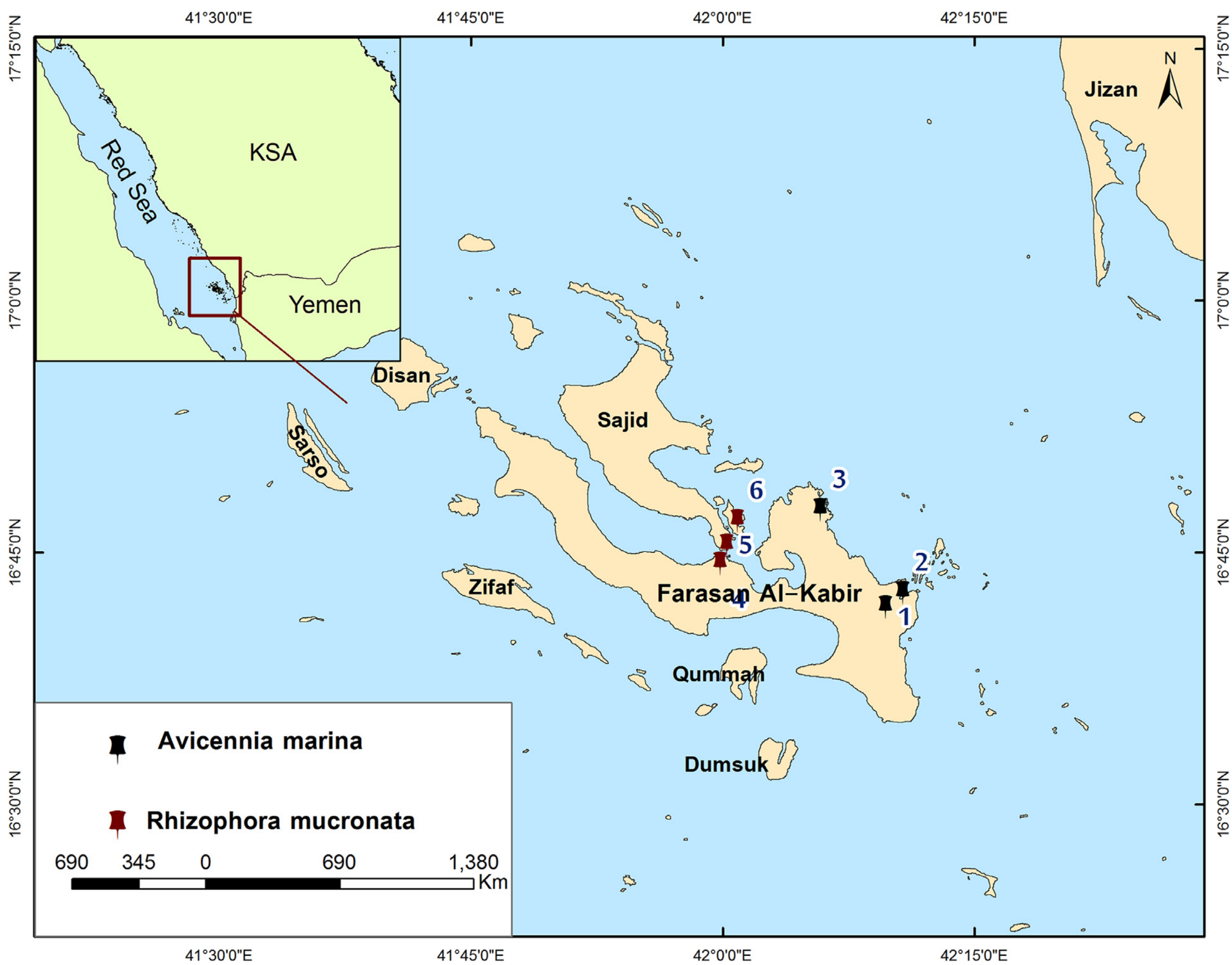

Figure 1 Map of the study area indicating the six sampling locations.

lagoon, the salinity tends to reach levels as high as 40%. The tide is every 11 h (diurnal) and low, with a mean spring range of 0.5 m at the region of the Farasan Islands (Bantan, 1999). As the Farasan Islands are located in the southern Red Sea, their climate is a monsoon climate. The region is influenced by the monsoon winds coming from the north in the summer and those coming from the south-east in the winter (Marcos, 1970). This results in the southern Red Sea surface water currents being affected around the Farasan Islands. In this region, the surface currents flow in a southerly direction in the summer months and in a northerly direction in the winter months.

2.2. Sediment sampling

Six locations were selected for sampling which was conducted in July of 2019. Locations representative of the *A. marina* monospecific stands were 17°48′28.7″N, 41°51′57.0″E; 17°47′31.5″N, 41°53′18.5″E; and 17°10′43.9″N, 42°22′2.1″E and for the *R. mucronata* stands: 17°47′37.5″N, 41°53′49.9″E; 17°46′40.8″N, 41°54′37.3″E; and 17°13′ 17.0″N, 42°20′20.5″E. All these locations were along the coastline of the Farasan Islands (Fig. 1). Kauffman and Donato's (2012) the *Rule of Twelfths* was used for sampling during low tides (<0.3 m water depth) in locations that were accessible on foot. At each sampling location, seven sites were selected and seven soil cores were collected from each one. Samples from both *A. marina* and *R. mucronata* locations were collected using a hand sediment corer (made of stainless steel, 100 cm long with an inner diameter of 70 mm). To collect the core, the researchers pushed the corer down into the soil to a depth of 50 cm. This depth was selected owing to the presence of raised coral/rock and subfossil at depths below 50 cm. Next, the sediment core was slowly extracted out of the corer and then, divided into ten 5 cm sections: 0—5, 5—10, 10—15, 15—20, 20—25, 25—30, 30—35, 35—40, 40—45, and 45—50 cm. Each section was then stored in a plastic container which was sealed with parafilm and volatilisation loss was prevented along with a decrease in the activity of microbes by storing the container on ice until further analysis (Eid and Shaltout, 2016). Thus, from each of the ten layers of soil, one sample was collected, and this was done for each of the 42 sites that were sampled (21 sites for each species). Overall, 420 samples of sediment were gathered for the analysis of SBD, SOC stock, and SOC concentration.

2.3. Sample analysis

The sediment samples were first oven dried at a temperature of 105°C over 3 days. The samples were then weighed to establish SBD [g cm^{-3}] as per Wilke's (2005) methodology:

$$\rho_{sj} = \frac{m_j}{v_j},$$

where ρ_{sj} is the SBD [g cm^{-3}] of the j^{th} layer, m_j is the mass of the sediment sample [g] of the j^{th} layer dried at 105°C, and v_j is the volume of the sediment sample [cm^3] of the j^{th} layer. Once dried, the samples were ground and sieved to ensure particle size was below 2 mm. SOC concentrations were established for each sample by measuring the sediment organic matter (SOM). The loss-on-ignition method was used which was performed for two hours at 550°C as described by Jones (2001). SOC concentration was established as [g C kg^{-1}] = 0.50 [Pribyl, 2010] × SOM [g C kg^{-1}]. A CHN analyser was not available to the researchers at the time of the study, and therefore, as per the recommendations of Nóbrega et al. (2015), the loss-of-ignition methodology was employed instead. This was required to establish a precise approximation of SOC in the sediments from the mangroves. Next, following the methodology of Meersmans et al. (2008), SOC stock [kg C m^{-2}] was quantified using the following equation:

$$SOC_s = \frac{\sum_{j=1}^{k} \rho_{sj} \times SOC_j \times T_j}{\sum_{j=1}^{k} T_j} \times D_r,$$

where SOC_s is SOC stock [kg C m^{-2}], ρ_{sj} is the SBD [g cm^{-3}] of the j^{th} layer, SOC_j is the SOC concentration [g C kg^{-1}] of the j^{th} layer, D_r is the reference depth [= 0.5 m], T_j is thickness [m] of the j^{th} layer, and k is the number of layers [= 10].

CSR [g C m^{-2} yr^{-1}] was estimated based on the sedimentation rate, SBD, and SOC concentration (Xiaonan et al., 2008):

$$CSR_i = \rho_{si} \times SOC_i \times R,$$

where CSR_i is the CSR [g C m^{-2} yr^{-1}] of the i^{th} location, ρ_{si} is the mean SBD [g cm^{-3}] of the i^{th} location, SOC_i is the mean SOC concentration [%] of the i^{th} location, and R is the sedimentation rate in the mangrove forests (the mean for Saudi Arabia = 2.2 mm yr^{-1}; Almahasheer et al., 2017).

CSP [Mg C yr^{-1}] was calculated as follows (Xiaonan et al., 2008):

$$CSP = CSR \times A,$$

where CSP is the CSP [Mg C yr^{-1}] of the mangrove stands and A is the area [m^2] of the mangrove stands (216.4 ha; Khan et al., 2010).

2.4. Statistical analysis

Before analysis, the data was evaluated for normality of distribution and homogeneity of variance using the Shapiro-Wilk's W test and Levene's test, respectively. Log transformation was performed on the data when necessary prior to conducting analysis of variance (ANOVA). Two-way ANOVA (ANOVA-2) was used to identify statistically significant differences in SBD and SOC concentrations in *A. marina* and *R. mucronata* samples for each of the ten sediment depths. Significant difference between means among the ten sediment depths were identified using the Tukey's HSD test at $P < 0.05$. Non-linear regression and Pearson correlation coefficient were used to evaluate the association between the SOC concentration and SBD (Shaltout et al., 2019). Student's *t*-test was used to identify significant differences among the *A. marina* and *R. mucronata* locations for the total means of SOC stock, SOC concentration, SBD, CSP, and CSR. All statistical analyses were performed using SPSS 15.0 (SPSS, 2006).

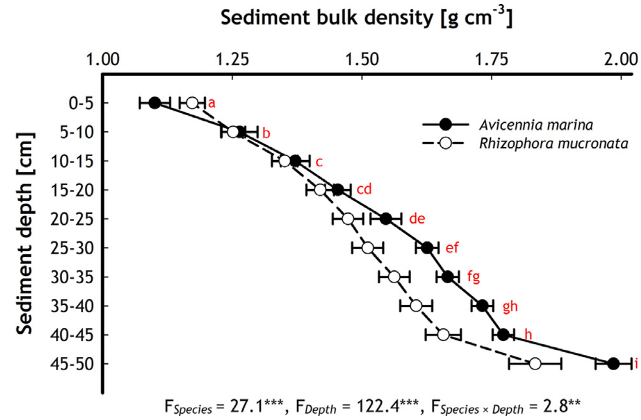

Figure 2 Distribution of sediment bulk density [g cm^{-3}] in relation to sediment depth [cm] in *Avicennia marina* and *Rhizophora mucronata* locations along the coastline of the Farasan Islands, Saudi Arabia. Horizontal bars indicate the standard error of the means [$n = 21$]. *F*-values represent results of the two-way ANOVAs. Means followed by different letters are significantly different at $P < 0.05$ according to the Tukey's HSD test. Location: *A. marina*/*R. mucronata*; Depth: 0—5, 5—10, 10—15, 15—20, 20—25, 25—30, 30—35, 35—40, 40—45, 45—50 cm. **: $P < 0.01$, ***: $P < 0.001$.

3. Results

Differences in SBD were significant between the *A. marina* and *R. mucronata* locations with a *t*-value of 4.1 ($P < 0.001$) and the greatest mean values belonging to *A. marina*; thus, these findings demonstrate that the null hypothesis is not true (Table 1). Variation in SBD at the *A. marina* locations was clearly high as it increased from 1.10 g cm^{-3} at the depth of 0—5 cm to 1.99 g cm^{-3} at the depth of 45—50 cm (Fig. 2). In comparison, for *R. mucronata*, SBD increased from 1.17 g cm^{-3} at the depth of 0—5 cm to 1.83 g cm^{-3} at the depth of 45—50 cm.

Significant differences were also observed for the SOC concentrations with *t*-values of 7.7 ($P < 0.001$) between the *A. marina* and *R. mucronata* locations. The greater average differences were in the sediments from the *A. marina* locations; thus, the null hypothesis was once again disproved (Table 1). A decrease was noted in the SOC concentrations at the *A. marina* locations from 23.4 g C kg^{-1} at the depth of 0—5 cm to 13.2 g C kg^{-1} at the depth of 45—50 cm (Fig. 3). At the *R. mucronata* locations, the SOC concentrations clearly declined from 20.4 g C kg^{-1} to 11.8 g C kg^{-1} at the depth of 0—5 cm to 45—50 cm, respectively. SOC concentration (g C kg^{-1}) and SBD (g cm^{-3}) had a significant and an inverse relationship, which was described by the following non-linear regression equations: SBD = 0.879 + 10.021 $e^{-0.172 \times \text{SOC concentration}}$ ($r = -0.633$, $P < 0.001$) and SBD = 1.059 + 9.638 $e^{-0.219 \times \text{SOC concentration}}$

Table 1 Mean ± standard error of sediment bulk density — SBD [g cm^{-3}], sediment organic carbon (SOC) concentration [g C kg^{-1}], SOC stock [kg C m^{-2}], carbon sequestration rate — CSR [g C m^{-2} yr^{-1}], and carbon sequestration potential — CSP [Mg C yr^{-1}] of *Avicennia marina* and *Rhizophora mucronata* locations along the coastline of the Farasan Islands, Saudi Arabia.

Species	SBD	SOC concentration	SOC stock	CSR	CSP
A. marina	1.55 ± 0.02 [n = 210]	16.3 ± 0.3 [n = 210]	12.3 ± 0.1 [n = 21]	5.4 ± 0.1 [n = 210]	11.8 ± 0.1 [n = 210]
R. mucronata	1.48 ± 0.02 [n = 210]	14.9 ± 0.2 [n = 210]	10.8 ± 0.2 [n = 21]	4.8 ± 0.1 [n = 210]	10.3 ± 0.1 [n = 210]
t-value	4.1***	7.7***	6.0***	10.9***	10.9***

t-values represent the Student's t-test. ***: $P < 0.001$.

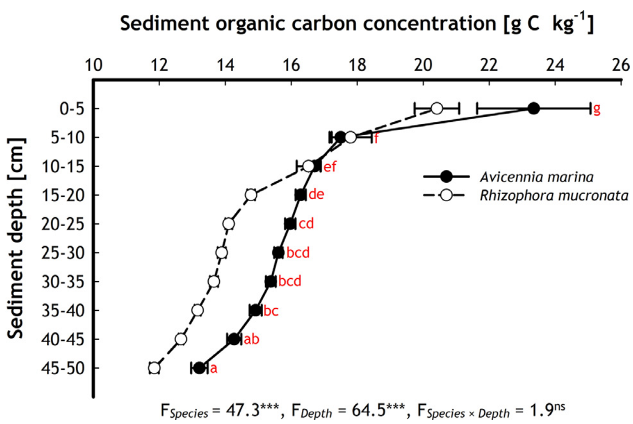

Figure 3 Distribution of sediment organic carbon concentration [g C kg^{-1}] in relation to sediment depth [cm] in the *Avicennia marina* and *Rhizophora mucronata* locations along the coastline of the Farasan Islands, Saudi Arabia. Horizontal bars indicate the standard error of the means [n = 21]. F-values represent results of the two-way ANOVAs. Means followed by different letters are significantly different at $P < 0.05$ according to the Tukey's HSD test. Location: *A. marina*/*R. mucronata*; Depth: 0—5, 5—10, 10—15, 15—20, 20—25, 25—30, 30—35, 35—40, 40—45, 45—50 cm. ***: $P < 0.001$, *ns*: not significant [i.e. $P > 0.05$].

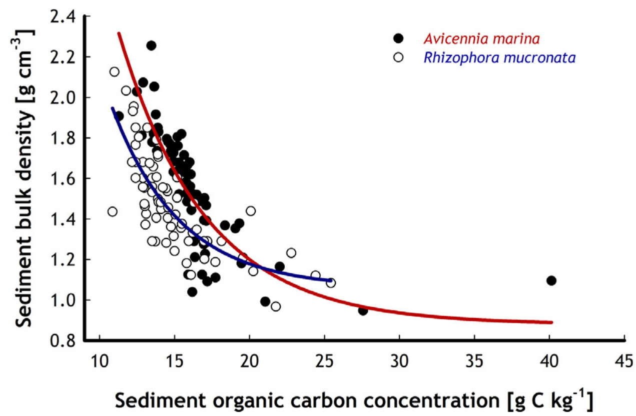

Figure 4 Non-linear correlation between sediment organic carbon concentration [g C kg^{-1}] and sediment bulk density [g cm^{-3}] of 420 sediment samples from *Avicennia marina* and *Rhizophora mucronata* locations along the coastline of the Farasan Islands, Saudi Arabia.

($r = -0.718$, $P < 0.001$) for *A. marina* and *R. mucronata* locations, respectively (Fig. 4).

Allowing for the entire sediment interval depth from 0 to 50 cm, the greatest SOC stock value was observed at *A. marina* locations (12.3 kg C m^{-2}), whereas the least value for the SOC stock was observed at *R. mucronata* locations (10.8 kg C m^{-2}). Thus, SOC stock was 114.3% greater for *A. marina* locations than for *R. mucronata* locations (Table 1). Therefore, the null hypothesis was once again disproved. A significantly greater mean CSR was observed at *A. marina* locations (5.4 g C m^{-2} yr^{-1}) than at *R. mucronata* locations (4.8 g C m^{-2} yr^{-1}) with a t-value of 10.9 ($P < 0.001$). Considering the mangrove forest area of 216.4 ha and the CSR values, the mangroves of the Farasan Islands had a total CSP ranging from 10.3 Mg C yr^{-1} at *R. mucronata* locations to 11.8 Mg C yr^{-1} at *A. marina* locations.

4. Discussion

The SBD is a dynamic feature that differs with structural conditions within the sediment (Pravin et al., 2013). It is frequently employed as a measure of two soil features, namely, ventilation and moisture (Huang, 2015). The mechanical resistance of the soil with regard to the growth of plants is also established using SBD; it may also affect the distribution of SOC concentration while playing a significant part in the approximation of this parameter (Johnston et al., 2004). The coastline of the Farasan Islands has mangrove forests along it; the SBD of this mangrove forest is within the value range described for mangroves inhabiting the Red Sea coast (Table 2). Additionally, as depth increased, so did SBD for both *A. marina* and *R. mucronata* locations. This finding has been described by other authors, including for the coast along the Egyptian Red Sea (Eid and Shaltout, 2016), along an approximate 1134 km of the coast of the Saudi Arabian Red Sea (Shaltout et al., 2019), and along the southern region of the coastline of the Saudi Arabian Red Sea (Arshad et al., 2018; Eid et al., 2016). Such variation in SBD has been attributed to the build-up of the remains of plants and tailings in the various layers of the sediment (surface and subsurface) (Sherry et al., 1998). These lead to changes in the organic matter content, porosity, and compaction (Pravin et al., 2013).

Underground roots, dropped litter, autochthonous matter, and other forms of material produced locally by mangroves are the initial contributors to SOC concentrations in mangrove forests (Alongi, 1998); as also benthic algae (Yong et al., 2011); seagrass and seaweed captured in the area (Mandura et al., 1988); and matter brought

Table 2 Mean of sediment bulk density — SBD [g cm^{-3}], sediment organic carbon (SOC) concentration [g C kg^{-1}], SOC stock [kg C m^{-2}], and carbon sequestration rate — CSR [g C m^{-2} yr^{-1}], of *Avicennia marina* and *Rhizophora mucronata* locations along the coastline of the Farasan Islands, Saudi Arabia compared with those reported for different mangrove forests around the globe.

Location	Species	SBD	SOC concentration	SOC stock	CSR	Depth [cm]	Reference
Farasan Islands, Saudi Arabia	*Avicennia*	1.55	16.3	12.3	5.4	50	Present study
	Rhizophora	1.48	14.9	10.8	4.8	50	
Red Sea coast, Saudi Arabia	*Avicennia*	1.5-1.9	14.4-18.1	6.7-10.5	5.0-6.0	50	Shaltout et al. (2019)
Southern Saudi Red Sea coast	*Avicennia*	1.66	17.7	29.2		100	Eid et al. (2019)
Southern Saudi Red Sea coast	*Avicennia*	1.53-1.66	12.6-15.7	9.9-11.5	4.4-5.1	50	Arshad et al. (2018)
Central Saudi Red Sea coast	*Avicennia*		2.0-15.0	2.5-7.6	1.5-5.5	100	Almahasheer et al. (2017)
Southern Saudi Red Sea coast	*Avicennia*	1.5	28.7	17.0	11.9	40	Eid et al. (2016)
Arabian Gulf, United Arab Emirates	*Avicennia*			10.2-15.6		100	Schile et al. (2017)
Red Sea coast, Egypt	*Avicennia*	1.4	15.5	8.5	6.1	40	Eid and Shaltout (2016)
Africa' Sahel, Senegal	*Avicennia & Rhizophora*			7.4-10.7		40	Woomer et al. (2004)
Ambanja and Ambaro bays, Madagascar	*Avicennia & Rhizophora*	0.52-1.39	6.0-61.0	32.4-51.7		150	Jones et al. (2014)
Zambezi River Delta, Mozambique	*Avicennia & Rhizophora*	0.72-0.95	14.5-23.6	27.5-31.4		200	Stringer et al. (2015)
Tropical Pacific, Micronesia	*Rhizophora*		130.0-150.0	51.7-94.7		100	Donato et al. (2012)
Yap Island, Micronesia	*Rhizophora*	0.27-0.51	73.0-137.0	12.1		100	Kauffman et al. (2011)
Pohnpei Island, Micronesia	*Avicennia*	0.10-0.43	114.6-364.9	177.1-211.6	53.0-93.0	365	Fujimoto et al. (1999)
Babeldaob Island, Palau	*Rhizophora*				69.8-369.7	60	Mackenzie et al. (2016)
Babeldoab Island, Palau	*Rhizophora*	0.18-0.36	120.0-215.0	56.7		100	Kauffman et al. (2011)
Moanaanuanu Estuary, New Zealand	*Avicennia*			6.9-7.0	31.8-57.6	40	Pérez et al. (2017)
Australian coast	*N.A.*			48.1-101.2		200	Sanders et al. (2016)
Pilbara coast, Australia	*Avicennia*		14.0-70.0	11.8		100	Alongi et al. (2000), Alongi et al. (2003)
	Rhizophora		23.0-65.0	16.9		100	
Lagoons and estuaries, Sri Lanka	*Avicennia & Rhizophora*	0.97-1.37	53.0-97.0	31.6-58.1		45	Perera and Amarasinghe (2019)
Batticaloa Lagoon, Sri Lanka	*Avicennia & Rhizophora*	0.40-1.60	3.0-51.0	100.9-784.6		80	Jonsson and Hedman (2019)
Mekong region, Vietnam	*Rhizophora*				121.0-602.7	60	Mackenzie et al. (2016)
Mekong Delta, Vietnam	*Avicennia & Rhizophora*	0.52-0.86	17.9-52.0	66.7		250	Dung et al. (2016)
Mekong Delta, Vietnam	*Avicennia & Rhizophora*	0.61-1.20	13.0-33.0	58.4-65.5		250	Tue et al. (2014)
Pulau Ubin Island, Singapore	*Avicennia & Rhizophora*	0.73	45.0	30.7		100	Phang et al. (2015)
Honda Bay, Philippines	*Avicennia*	0.48-0.62	64.3-87.3	85.2		300	Castillo et al. (2017)
Sawi Bay, Thailand	*Avicennia & Rhizophora*		22.5-54.3			100	Alongi et al. (2001)
Zhangjiang Estuary, China	*Avicennia*	0.94	12.7	9.6		100	Gao et al. (2019)
	Rhizophora	1.24	12.1	8.5		100	

(*continued on next page*)

Table 2 (*continued*)

Location	Species	SBD	SOC concentration	SOC stock	CSR	Depth [cm]	Reference
Hainan Dongzhaigang Wetlands, China	*Avicennia*			9.6		100	Xin et al. (2018)
	Rhizophora			5.6		100	
Leizhou Peninsula, China	*Avicennia & Rhizophora*	0.93-1.12	7.1-16.4	7.1-14.0	37.0-205.0	90	Yang et al. (2014)
Zhangjiang Estuary, China	*Avicennia*		10.0-12.7			60	Xue et al. (2009)
Segara Anakan Lagoon, Kongsi Island and Thousand Islands, Indonesia	*Avicennia & Rhizophora*	0.62-1.35	8.0-77.0	3.7-48.5	2.4	100	Kusumaningtyas et al. (2019)
Bay of Bengal, India	*Avicennia*	0.56	9.2	2.8		N.A.	Sahu and Kathiresan (2019)
	Rhizophora	0.57	8.4	2.6		N.A.	
Sundarbans, India	*Avicennia*		5.1-6.5	2.6		30	Ray et al. (2011)
Tampa Bay, USA	*Avicennia & Rhizophora*	0.44-0.90	63.0-110.0	10.1		50	Radabaugh et al. (2018)
La Paz Bay, Mexico	*Avicennia & Rhizophora*	0.9		10.0-23.9		45	Ochoa-Gómez et al. (2019)
Pantanos de Centle Wetland, Mexico	*Avicennia & Rhizophora*	0.49	131.0	13.7-200.3		300	Kauffman et al. (2016)
Yucatan Peninsula, Mexico	*Rhizophora*	0.11-0.95	32.0-351.0	28.6-116.6		100	Adame et al. (2013)
Ceará State, Brazil	*N.A.*			8.2		40	Nóbrega et al. (2019)
Crumahú River, Brazil	*Avicennia & Rhizophora*		163.0-266.0			80	Ferreira et al. (2010)
Tamandaré, Brazil	*N.A.*	0.61-1.33	10.9-214.2			45	Sanders et al. (2010)
Sepetiba Bay, Brazil	*Avicennia*		38.0-61.0			15	Lacerda et al. (1995)
	Rhizophora		27.0-28.0			15	
Caribbean coast, Venezuela	*Avicennia*	0.26-0.39	100.0-120.0	3.1-3.8		20	Barreto et al. (2016)
	Rhizophora	0.22-0.23	170.0-200.0	3.7-4.4		20	
Amboa Swamp, New Caledonia	*Avicennia*	0.24-0.45	51.0-115.3	25.6		100	Jacotot et al. (2018)
	Rhizophora	0.33-0.44	54.7-110.8	31.5		100	
Coastal wetlands, Dominican Republic	*Avicennia & Rhizophora*	0.30	175.0	75.3		195	Kauffman et al. (2014)

from nearby coastal areas by tidal flux, i.e. allochthonous material (Allisaon et al., 2003). All this material eventually ends up in the mangroves (Bouillon et al., 2003). On a worldwide scale, the organic carbon content of mangrove soil is highly variable with values ranging from 5 g kg^{-1} to above 400 g kg^{-1} and this is dependent on the composition and age of the forest, productivity, biomass, soil texture, geographical and morphological settings, tidal range, regional climate, and anthropogenic effects (Donato et al., 2011; Gao et al., 2019; Otero et al., 2017; Sanders et al., 2016). The average SOC concentration of 14.9 to 16.3 g C kg^{-1} observed in this study was below the global value of 22.0 g C kg^{-1} (Kristensen et al., 2008) and also below values estimated in the mangrove forests of many other nations, as shown in Table 2, including Thailand, Micronesia, Sri Lanka, Indonesia, Philippines, Australia, Palau, Brazil, Vietnam, New Caledonia, Singapore, Mexico, USA, Venezuela, Dominican Republic, and Madagascar. The values observed in this study were, however, greater than those estimated in China, Egypt, and India.

Zimmerman and Canuel (2000) described how input and decomposition systems at the various sediment depths are responsible for the concentration of SOC. SOC concentration decreased slowly starting from the surface and moving towards the bottom of the cores of the sediment in both *A. marina* and *R. mucronata* locations analysed in this study. *A. marina* sediments of the Saudi Arabian Red Sea mangroves showed a similar structuring as those of *A. marina* in Egypt (Eid and Shaltout, 2016). A study by Schile et al. (2017) also described a similar structuring in sediments of *A. germinans* and *A. marina* along the coast of the United Arab Emirates. This was further supported by previous research on the mangrove forests in Vietnam (Tue et al., 2011), Australia (Saintilan et al., 2013), the Asia-Pacific region (Donato et al., 2011), Mexico (Adame et al., 2013), and the Gulf of Mexico (Bianchi et al., 2013). It is clear that as depth increases, the SOC concentration decreases and this is thought to be due to variation in the ratio of matter obtained from the mangroves and allochthonous material, such as seston, as well as from sources of organic carbon contained in sediments and organic material mineralisation in the sediments of mangroves (Bouillon et al., 2003; Tue et al., 2012). Changes in the concentrations of SOC with increasing depth could also be caused by the interplay between various complex systems including decomposition, leaching, hydrologic regimes (sediment regimes), biological cycling, soil erosion, atmospheric decomposition, illuviation, and weathering of minerals (Girmay and Singh, 2012; Sanderman et al., 2018).

An indirect exponential association was observed between the SOC concentration and SBD at all *A. marina* and *R. mucronata* locations investigated in this study. SBD is a crucial factor for the approximation of the SOC stock; nevertheless, only a limited number of research studies have concurrently detailed SOC concentrations together with those of SBD when analysing the sediments of mangroves. The indirect association between SBD and the SOC concentration determined in this study, therefore adds a beneficial methodology for the approximation of SOC stock in mangrove forests (Donato et al., 2011). Furthermore, the negative association between the SBD and the SOC concentration also indicates that various factors may be affected by SBD including soil permeability, porosity, ventilation, and structure of the soil, all of which also affect the build-up of the soil (Gao et al., 2019). This study determined that SBD in the sediments diminishes with greater SOC concentrations. This association was also noted in previous studies including that for the *A. marina* sediments of the Saudi Arabian Red Sea (Arshad et al., 2018; Eid et al., 2016, 2019; Shaltout et al., 2019), *A. marina* sediments along the Red Sea coast of Egypt (Eid and Shaltout, 2016), and for four Chinese mangrove forests (Gao et al., 2019). This was also consistent with results of other studies including one near the United Arab Emirates coastline that was conducted on the sediments of *A. marina* and *A. germinans* (Schile et al., 2017), a study in Vietnam that examined tropical species of *Rhizophora* and *Avicennia* (Dung et al., 2016; Tue et al., 2014), an examination of species of *Avicennia* and *Rhizophora* present on the Venezuelan Caribbean coast (Barreto et al., 2016), and a study on *Rhizophora* species in the tropics (Donato et al., 2011). Thus, the negative association between SBD and SOC concentrations observed in this study are consistent with findings in different sediments in different regions of the globe.

The primary elements that contribute to establishing the SOC stock are SBD, SOC concentration, and the full depth at which the approximations are integrated (Stringer et al., 2015). Matsui (1998) described how beneficial conditions are produced for organic carbon preservation as a result of the anaerobic nature and the low rate of decomposition of mangrove sediments. The behaviour and quantity of SOC stock in the soil varies extensively in the various mangroves, and this is initiated by the particular carbon dynamics of the location and differences that result from the age of the forest, composition of the species, productivity and biomass of the vegetation, fertility of the soil, location of the inter-tide, sedimentation of suspended materials, edaphic conditions of soil (such as pH, salt content, and redox potential), and structure of the community (Alongi et al., 2003; Gleason and Ewel, 2002; Khan et al., 2007; Ray et al., 2011; Sanderman et al., 2018; Twilley et al., 2018).

Sanderman et al. (2018) noted that large amounts of soil carbon can build up in many mangrove forests but not in others. Significant differences can be found in the stocks of soil carbon in the various mangrove forests (Jardine and Siikamäki, 2014) and within the same forest of mangroves (Kauffman et al., 2011). Previous studies have also shown the differences in carbon stock across the globe in various mangrove forests (Donato et al., 2011; Kauffman et al., 2011; Khan et al., 2007). Generally, lower SOC stocks are found in mangroves located in arid and sub-humid areas than in humid areas (Schile et al., 2017). Moreover, a significant difference in the SOC stock for each unit area through the latitudinal bands was determined for mangroves from 0° to 10° S, with these two points indicating the greatest SOC stock levels of 351 Mg C ha^{-1}, whereas those between 20° and 30° N had the least, at 222 Mg C ha^{-1} (Atwood et al., 2017). Table 2 lists the data for blue carbon across the globe. It is clear that the carbon stocks from sediments of *A. marina* and *R. mucronata* that flourish in the Farasan Islands are low to fair relative to the global values. In this study, the mean SOC stock (11.6 kg C m^{-2}) determined for the mangrove sediments of the Farasan Islands was 3.1 times lower than the worldwide mean values described for mangroves, such as 36.1 kg C m^{-2} by Sanderman et al. (2018). The low level of SOC stock in the Farasan Islands may be because of small supplies of allochthonous matter or extreme environments that could have resulted in restricted levels of mangrove growth with trees that were dwarf-like with low biomass levels and increased rates of sediment respiration (Almahasheer et al., 2016a, b, 2017). Moreover, the sequestration of carbon may not have been encouraged by certain features of the habitats or by the geomorphological environment not being beneficial for their growth (Almahasheer et al., 2017). The results of Ren et al. (2010) further support this as they describe how stressful environments may lead to low SOC stocks that result from small supply levels of dead roots throughout a certain time period together with a low rate of falling debris. Furthermore, the value for SOC stock in this study may have been underestimated as a depth of over 3 m was recorded for the mangrove organic sediment in certain regions (Donato et al., 2011); however, this study only involved coring to depths of 0.5 m.

The build-up of organic carbon in the soil is the consequence of autochthonous matter being supplied via primary processes or the deposit of allochthonous matter and this being balanced against the loss of matter through erosion, mineralisation or decomposition, and leaching (Alongi, 2014). Sediment tends to be efficiently trapped and held in mangrove forests and this is attributed to the structure of the stems and their widespread roots (Furukawa et al., 1997; Krauss et al., 2003). The low rate of decomposition of root biomass together with the sediment's anoxic environment, leads to the build-up of a lot of organic carbon (Middleton and McKee, 2001). Climatic elements may also affect the sequestration of carbon in mangroves, such as through rainfall, temperature, and evapotranspiration (Alongi, 2008). Moreover, it could occur via coastal oceanographic systems such as currents, tidal heights, and geomorphology (Alongi, 2008). Other influences include the access to nutrients as described by Lovelock et al. (2014) and the functional traits of plants, including their turnover, the production inputs both above and below ground level, and the distribution of carbon (De Deyn et al., 2008; Nordhaus et al., 2006).

In this study, significant differences in CSR were demonstrated between *A. marina* and *R. mucronata* locations with greater average values in the *A. marina* sediments. These outcomes were consistent with the findings in the Vellar—Coleroon estuary, India, where higher results were observed for *A. marina* with a CSR that was 75% greater than that of *R. mucronata* (Kathiresan et al., 2013). The average CSR measured for the mangrove forests of Farasan Islands was from 4.8 to 5.4 g C m^{-2} yr^{-1}. This is similar to results of the study on the southern Saudi Arabian Red Sea mangrove sediments where a value of 4.7 g C m^{-2} yr^{-1} was described (Arshad et al., 2018). The results of this study are also similar to those reported by Shaltout et al. (2019) who registered a mean CSR value of 5.6 g C m^{-2} yr^{-1} in the mangroves of the Red Sea. All these values, however, are greater than those recorded by Almahasheer et al. (2017) who described a mean CSR value of 3.5 g C m^{-2} yr^{-1} for the mangrove sediments of the central region of the Red Sea. Alongi (2012) described the mean CSR values at the global level as 174.0 g C m^{-2} yr^{-1}, which is 34.1 times greater than that in the present study. The low levels determined in this study are probably caused by the following: absence of freshwater, oligotrophic aspects, low levels of allochthonous matter supplied to the Red Sea, and man-made stressors (Almahasheer et al., 2017; Kathiresan et al., 2013); extremely arid conditions affecting mangrove growth (Almahasheer et al., 2017); lower primary productivity in Red Sea mangrove forests (Arshad et al., 2018); high temperature and water eutrophication (Shaltout et al., 2019); and heavy metal pollution that may negatively influence primary production (Bouillon et al., 2008; Arshad et al., 2018). High temperature is known to have a strong influence on litter decomposition (Hanson et al., 1984). According to Schlesinger (1997), the decomposition rate doubles with every 10°C increase in temperature which means a low CSR. In addition, it has been reported that the disposal of nutrients (e.g. N and P) can accelerate the decomposition of mangrove SOM and thereby reduce SOC stocks (Feller et al., 2003; Suárez-Abelenda et al., 2014). Furthermore, coarse carbonates that are biogenic in nature are the primary building material of the mangroves in the soil of the Farasan Islands. This may also be indicated as a reason for the lower CSR levels of the mangroves that flourish in environments that are uninviting for the accretion of soil, production of biomass, and preservation than in mangroves from subtropical or temperate regions (Almahasheer et al., 2017).

5. Recommendations

The past five decades have seen the loss of one-third of the Earth's mangroves, which makes them one of the highly threatened ecosystems (Alongi, 2002). This is because these ecosystems are the source of fuel, food, timber, and medicine (Alongi, 2002). The change in the manner of mangrove land utilisation results in the loss of stored carbon into the atmosphere. It also affects other benefits provided by this ecosystem such as a habitat for marine life, greater levels of biodiversity, and protection at the coastal level (Castillo et al., 2017). This study evaluated the CSP of *R. mucronata* and *A. marina* locations along the coastline of the Farasan Islands and estimated values of 10.3 and 11.8 Mg C yr^{-1}, respectively. By employing conversion factor of 3.67 by Sahu and Kathiresan (2019), it was determined that *R. mucronata* and *A. marina* can sequester CO_{2e} yr^{-1} at levels of 37,801 and 43,306, respectively, which is equivalent to 6.3×10^{-8} and 7.2×10^{-8}, respectively, of the complete carbon emissions of Saudi Arabia for 2014 (601,047 kt CO_{2e}) (Trading Economics, 2019). It is, therefore, imperative that these ecosystems are not only protected but also re-established to ensure they play their part in the sequestration of carbon and for all the other benefits that they provide.

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