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Greer, Bethan

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Aboveground biomass of a South West Florida mangrove stand

Greer, B. R., 2019

Full name: Bethan Rachel Greer

Supervisor: Dr Nathalie Fenner

Institution: Bangor University

Abstract

Mangroves are salt-tolerant tree species found in tropical and sub-tropical regions around the globe. Whilst also important for many roles, such as storm protection, poverty alleviation and nurseries for many commercially important fish species, at present there is a focus on their potential for climate mitigation through carbon sequestration. However, to know how much carbon can be stored, it is important to know the aboveground biomass (AGB) of localised areas to monitor global carbon sequestration and produce climate mitigation procedures as AGB values can be used to estimate carbon sequestration when used in conjuncture with region-specific mangrove carbon content values. Whilst Florida's mangrove forests are under legal protection, there have been no recent studies of AGB and the most recent in 2004 was found to contain errors. This study aimed to produce estimates of the aboveground biomass of a Southwest Florida mangrove stand containing *Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa*. It was found that AGB ranged between 4.9-104.2 t/hectare which was consistent with other research based in Florida. The most important factors influencing Florida mangrove growth are considered to be temperature and impact of storm events such as hurricanes. However, it is unclear how future effects of climate change including rising sea levels, increased surface temperatures and increased intensity and frequency of storm events will alter AGB of the forests, and their carbon storage potential. Planned projects for the future will consider standing carbon stocks for use in international mitigation policy and estimating mangrove greenhouse gas emissions.

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Mangroves Ecosystem Services

Mangroves are facultative halophytes found around the equator and subtropical latitudes (Alongi, 2012; Siikamäki, 2013). Mangrove habitats are ecotones acting as a transition zone between terrestrial and marine ecosystems, with 'stands' consisting of individual mangrove 'stems' colonizing intertidal areas of coastlines, estuaries and river deltas (Siikamäki, 2013; Lawrence, 2012). The term 'mangroves' is used interchangeably to refer to either the coastal forest ecosystem, or to the 70 species of tree and shrub from plant families which have evolved to live in these coastal regions, including Rhizophoraceae and Avicenniaceae (Siikamäki, 2013; FAO, 2007), but there is a growing consensus that true mangroves are only those species that are viviparous (Tomlinson, 2016). Adaptations for this environment include; glands on leaves to excrete salt, specialised aerial roots which aid in direct uptake of atmospheric gases, vivipary and trapping nutrients (Siikamäki, 2013; FDEP, 2017). Mangroves have a 'bottom-heavy' tree form which allows the stems to stay upright in the soft sediment (Komiyama et al, 2008). In areas close to their environmental limits, mangroves may only grow into small shrubs, whereas in more favourable conditions their canopies can reach heights between 30 and 40m (FAO, 2007).

Mangroves account for 0.7% of tropical forest cover, with the most recent study, Giri et al (2010) estimated there are 13,776,000 hectares of mangrove forest globally, with the highest density found in Asia and Australia (FAO, 2007; Lawrence, 2012). Mangrove forests are considered valuable due to the large number of ecosystem services they provide, with an estimated economical value of over US$900,000/km² per annum (Estrada and Soares, 2017). In regions at risk of extreme weather events including hurricanes, flooding and tsunamis, mangroves provide storm protection for human settlements and other coastal habitats as their unique root system absorbs wave energy before they reach shore and traps sediments from upstream sources, reducing the risk of coastal erosion (Simard et al, 2006; Lawrence, 2012; Siikamäki, 2013; FAO, 2007). The roots also create a suitable habitat for fish to use as nursery grounds, which helps provide food security as in mangrove areas the average yield of fish and shellfish can be up to 90kg per hectare (Siikamäki, 2013; FAO, 2007). Mangroves provide fuel for coastal communities and opportunities for making revenue from tourism. This makes these ecosystems important resources for poverty alleviation, as many of the regions with the highest densities of mangroves are developing countries (Lawrence, 2012). The most globally relevant service mangroves is carbon sequestration (Kristensen et al, 2008). Mangroves contain 3-4 times the volume of carbon (C) found in other types of forest,

including tropical forests, making them important stores of coastal 'blue carbon' (Kristensen et al, 2008; Siikamäki, 2013). In mangrove forests, between 50 and 90% of the total C sequestered is contained within the sediment and soil, whilst the rest is stored as living biomass (Lawrence, 2012). This C storage has potential to reduce atmospheric levels of CO_2 and is therefore important to climate change policies (Siikamäki, 2013).

In the last 50 years it is estimated that 50% of global mangrove area has been lost through habitat degradation and deforestation, with deforestation rates of up to 80% in tropical regions (Alongi, 2012; RAMSAR, 2010). This can partially be explained by the presence of dense human populations along low-latitude coastlines in tropical regions, where less affluent communities depend upon mangroves for resources (Alongi, 2012). This loss of mangrove habitat prevents more carbon from being sequestered and the loss leads to increased total carbon emissions released through deforestation. Mangroves may contribute 10% of these emissions due to increased bacterial activity in disturbed mangrove sediments and destruction of standing C stocks (Siikamäki, 2013). Mangroves are most commonly displaced to provide land for urban development, aquaculture and also by sea-level rise, leading to habitat fragmentation and disturbance of natural processes (Alongi, 2012; FAO, 2007; Simard et al, 2006).

Whilst mangroves are considered a particularly hardy environments, factors including salinity, topography, temperature, tidal flushing and tidal flow ultimately impact the scale of the forests and their total biomass, making mangroves vulnerable to change (Alongi, 2012). Habitat degradation has the potential to switch mangrove forests from carbon sinks to carbon sources of if the environment becomes stressed or damaged (Macreadie et al, 2017). Natural habitat disturbance can also impact the sequestration of C when increased frequency of extreme weather events, insect outbreaks and disease damage mangrove stands and reduce habitat area. However, these forests are more susceptible to degradation when stressors such as pollutants are introduced to the ecosystem (Alongi, 2012; Macreadie et al, 2017). Some of these pollutants can come from industries such as aquaculture, which is prevalent in areas where mangrove density has been historically high (Lawrence, 2012; Siikamäki, 2013).

Despite raised awareness of the need to protect these environments, conservation efforts are hampered by a lack of data on factors such as mangrove status and distribution, making it difficult to prepare projects enough to be successful (FAO, 2007). In other areas, mangroves are degraded by unsustainable management which impacts communities and habitats surrounding these forests (FAO, 2007). One way to potentially

improve how mangrove forests are monitored and conserved could be to include them in global policies on forest carbon storage.

Global Carbon Policy

Policy makers are increasingly turning to 'non-conventional' ecosystems such as coastal wetlands to reduce the impact of anthropogenic emissions on climate change (Wylie et al, 2016). Carbon markets are a form of climate finance, where climate change mitigation projects in developing countries are supported financially by the governments of developed countries or by private entities (Thomas, 2014). These markets assume that adding a financial incentive to conserve a valuable ecosystem will lead to improved management by application of scientific measures to estimate stored C, this carbon can then be sold as carbon credits, allowing buyers to offset their own emissions (Wylie et al, 2016). The most common form of a C market is the use of Payment for Ecosystem Services (PES), where the total C sequestered is quantified and converted into credits, primarily used in voluntary C markets, as coastal projects are not typically large enough in scale to qualify for compliance market sale (Friess, 2013; Wylie et al, 2016). The total carbon sequestered can be estimated by multiplying the AGB values of the mangrove habitat by the region-specific C content estimate (IPCC, 2014). PES particularly focus

on the rate of C accumulated from the atmosphere and long-term storage through the prevention of deforestation, which makes it difficult to achieve C targets in short-term projects (Alongi, 2011).

Despite the issues with inclusion of coastal wetland systems, the continuing development of the United Nations Framework Convention on Climate Change (UNFCCC), there has been great progress in the incorporation of C markets and PES schemes into international law. The Kyoto Protocol set guidelines for ratifying member nations, as well as participating industry members, to reduce their greenhouse gas (GHG) emissions. Projects such as the United Nation's Reducing Emissions from Deforestation and Forest Degradation scheme (REDD+) and the Clean Development Mechanism (CDM) also encourage the conservation and regeneration of forest land that has been previously degraded or deforested (Friess, 2013; Wylie et al, 2016). These nations can alter their emissions by modifying their land use, either using national assessments of GHG emissions and C sequestered in standing stocks, or by funding conservation projects (Wylie et al, 2016). The UNFCCC also has the Green Climate Fund (GCF) which works to provide investment for climate mitigation projects (Wylie et al, 2016). The Intergovernmental Panel on Climate Change (IPCC) uses the work of collaborating scientists to advise the UNFCCC on methodologies for its C projects,

with different tiers of assessment to increase reliability of C sequestration estimates (IPCC, 2014). These methods can be used by members of the Kyoto agreement to produce national greenhouse gas assessments (IPCC, 2014). It has been suggested these methodologies are rigorous enough that they could be expanded for use in coastal wetland projects (Crooks et al, 2011).

REDD+ is often considered to be the best project for coastal wetland environments such as mangroves to be incorporated into, with strategies developed by over 40 countries (Alongi, 2011). These projects tend to focus on afforestation as the early stages of forest development are typically characterised by increased C sequestration compared to later successional stages (Alongi, 2011). However coastal wetland systems may be considered too dynamic for current REDD+ projects due to the influence environmental factors and stressors have on C storage potential, as well as difficulties creating easily replicated methodologies for surveying large areas of habitat both above- and belowground (Friess, 2013). Whilst methodologies for wetlands need to be developed for REDD+, the CDM allows Annex 1 countries (developed) to implement Nationally Appropriate Mitigation Actions (NAMAs) in developing countries as a method for receiving carbon credits for offsetting their own emissions using ecosystems such as mangroves (Wylie et al, 2016). Some countries have developed their

own policies for monitoring of standing C stocks which are used alongside guidelines produced for the Kyoto agreement. An example of this would be the US inclusion of total forested area carbon estimates in their National GHG Inventory as well as the collection of stock change data by the US Department of Agriculture Forest Services Inventory and Analysis program (Smith et al, 2013). Whilst these policies are designed to help prevent further release of C sequestered from the atmosphere, it is not always easy to produce these estimates and apply standards for policy making. These policies are most effective for conservation when based on reliable and robust scientific evidence (Friess, 2013).

Practical Problems of Mangrove Measurements

A large problem for creating accurate estimates of AGB in mangrove forests globally is that there are many variables impacting the accuracy of studies. It is difficult to study how each individual species at a site may impact stand C storage potential, as AGB data from single species plantations and managed forests are excluded from global averages. This makes calculating total mangrove ABG, and therefore standing C stocks difficult (Estrada and Soares, 2017). Variation could also be explained by differences in the age of stands measured (Estrada and Soares, 2017).

During the early stages of mangrove growth there is a higher rate of aboveground biomass (AGB) growth until maximum leaf cover causes it to peak (Estrada and Soares, 2017). After this point, AGB accumulation (ANPP) reduces to a much slower rate as gross primary productivity stabilizes, with only wood respiration increasing (Estrada and Soares, 2017).

The latitudinal limits of mangrove species are due to several key factors (Webber and Ferreira, 2016; Lopez-Medellin and Ezcurra, 2012). As latitude increases there is generally a reduction in solar radiation and air temperature, which leads to dwarf forms of mangroves and higher risk of mortality from extreme cold weather events (Webber and Ferreira, 2016; Estrada and Soares, 2017). However, particularly arid areas with low annual rainfall may also produce stunted mangrove forms (Estrada and Soares, 2017). It is suggested that soil moisture results in submersing of the roots which increases allocation of mangrove biomass to the root system (Komiyama et al, 2008), but this soil moisture is best for mangroves when it is from freshwater. Increased salinity reduces biomass productivity of mangrove forests and reduces the level of carbon sequestered in biomass (Rahman et al, 2015). Most mangrove species require low to medium salinity levels of 2-18ppt to reach their maximum potential productivity, although some species can survive at 35ppt (Estrada

and Soares, 2017). Rahman et al (2015) found that mangrove plots in freshwater zones had significantly higher stocks of carbon than those in more saline conditions. High salinity levels often coincide with low nutrient levels, which increases stress on mangrove stands and reduces growth (Feller et al, 2003). This is because Tidal influxes of nutrients often fail to reach the interior of mangrove forests, specifically nitrogen (N) and phosphorous (P), which are limiting variables for mangrove growth and biomass accretion (Feller et al, 2003).

On the other hand, excess nutrients can result from anthropogenic disturbance such as from use of fertilizers, combining with environmental variables to change the flux of nutrients in the forests and their uptake by mangrove stems (Feller et al, 2003). Other natural causes of disturbance include fire disturbance and lightning, which is known to create canopy gaps in mangroves by causing mortality of trees surrounding the area struck (Sherman et al, 2000). On a larger scale, storm events such as hurricanes and tsunamis can create disturbance over a greater area. One of the immediate results of storm action is the uprooting of trees, reducing the total biomass and C sequestration of an area (Smith et al, 2009). Sediment deposition may have a longer lasting impact on mangrove forests as the layers of sediment brought in by flooding and storm surges can cover the roots of

mangroves and suffocate large areas of forest (Smith et al, 2009).

Florida

The USA, including its territories, contains 195,150 ha of mangrove forest along its coastlines (FAO, 2007). However, countries such as Indonesia and the Philippines, which have 2,900,000 ha and 240,000 ha of mangroves respectively, exhibit considerably higher biodiversity with 43 and 35 species respectively, compared to just 6 in the USA and its territories (FAO, 2007). The lower number of mangrove species is most likely due to the USA mangrove forests being at higher latitudes than these other countries. The higher latitudes would reduce the available suitable habitat for mangroves as they are unable to survive particularly cold temperatures that may occur (FDEP, 2019; Osland et al, 2017). There are also frequent extreme weather events in the USA which can damage or destroy areas of mangrove forest. These events either break the stems with high winds or suffocate the mangroves by covering the aerial roots with sediment (Smith et al, 2009).

Of the 195,150 ha of mangroves in the USA, roughly 189,798 of those hectares are found in the state of Florida (FDEP, 2019). Mangroves in Florida are found along both coastlines, with the highest densities found to the South in the Everglades National Park (US

Fish and Wildlife Services). There are 3 species of mangrove present in the state: *Rhizophora mangle*, *Avicennia germinans* and *Laguncularia racemosa* (FDEP, 2019). At the northern distribution limits, *Avicennia germinans* is the most abundant species and is more tolerant of cooler temperatures (US Fish and Wildlife Services). As distance from the equator and salinity decreases *Rhizophora mangle* becomes the more dominant species, with *Laguncularia racemosa* found more sporadically and in lower densities as elevation decreases (US Fish and Wildlife services).

Mangrove forests in Florida are vital ecosystems that support a wide variety of species including American Alligators, Ospreys, Raccoons, species of Mud Fiddler Crabs and American Crocodiles which are listed as vulnerable by the IUCN (US Fish and Wildlife Services). They also provide an important function in providing some protection to Florida coastal communities from hurricanes (FDEP, 2019; US Fish and Wildlife Services). However, mangroves are under threat of degradation and deforestation even in developed countries such as the USA.

Whilst destruction of mangrove forests to leave room for human development is the most commonly known threat to mangrove systems, one factor that also contributes to mangrove degradation is wake production from boats passing these fragile ecosystems (Shahbudin et al, 2011; Manis et al, 2014). As

boats pass mangrove ecosystems, the wake caused can erode the sediment surrounding the roots due to the energy of the wave and the stress it puts on the sediment (Manis et al, 2014). In Florida, 59% of the state's coastline is considered eroded (Manis et al, 2014). Both natural and anthropogenic processes are considered responsible for this occurrence. Although shorelines are naturally eroded by tidal activity, wind and currents, the construction of boating inlets and jetties, as well as commercial and recreational boating are considered a major cause of shoreline ecosystem degradation in Florida and around the world (Shahbudin et al, 2011; Manis et al, 2014). To prevent erosion, it has been suggested that mangroves should be planted, and this has been implemented in some areas in the state (Goforth and Thomas, 1980). However, there are other factors that can impact mangrove health and growth including pests and eutrophication (Rehm and Humm, 1973; Brooks, 2004; Lovelock et al, 2009; Schaffelke et al, 2005).

Eutrophication as a result of both hurricanes and excess run-off of fertiliser and other pollutants containing nitrogen (N) is a particular concern in the state of Florida (Lapointe and Clark, 1992; Lovelock et al, 2009; Harmon et al, 2013). When climatic variables are favourable for growth, N can increase productivity. But when the environment is less favourable, potentially due to hypersaline conditions, low

atmospheric humidity or periods of low rainfall, excess N can lead to increased stem mortality (Lovelock et al, 2009).

But environmental scientists and lawmakers have been able to create important policies for the protection of Floridian mangrove forests. Since 1996, all mangroves in the state of Florida are protected by the Mangrove Trimming and Preservation Act, enforced by the Florida Department of Environmental Protection (FDEP, 2019). This act was created to protect Florida's mangroves from unregulated defoliation, destruction and removal by those who are unlicensed and unqualified (FDEP, 2019). Florida residents who wish to have mangroves on their property trimmed must either apply for a general license from the FDEP, or from a local government that the FDEP has found to have "adequate resources for the administration and enforcement of a delegated mangrove regulatory program" (FDEP, 2019). The use of herbicides on mangroves is strictly prohibited, as is the trimming of mangroves on uninhabited land that is owned publicly or by local governments, apart from where exemptions are made, such as to maintain public safety or to keep public waterways clear (FDEP, 2019). Where mangroves are found to have been defoliated, destroyed or removed, the responsible party has 3 options, they must restore, mitigate or offset the costs. For restoration, mangroves of the same species impacted must be replanted at the

same location and need to achieve a canopy area equivalent to the area affected within 5 years of the damage. Additional planting is required until the survival rate is 80%. If neither of these options is practicable, the cost of the damage caused must be offset by a monetary donation to a mangrove project for restoration, creation, enhancement or preservation, or through purchasing mitigation credits double the worth of the mangroves affected by the illegal damage to the mangroves (FDEP, 2019). One important factor for deciding the worth of mangroves, is the carbon sequestration potential, which requires the estimation of aboveground biomass. Allometric equations have been created more recently for Florida mangrove species but whilst these can be used to calculate AGB, the authors gave no examples or study data (Smith and Whelan, 2006; Dai et al, 2018).

Previous studies that have taken place in Florida have produced a range of values for AGB. Florida estimates were taken from Taylor Slough (Coronado-Molina et al, 2004), Biscayne National Park (Ross et al, 2001), Rookery Bay and Ten Thousand Islands (Lugo and Snedaker, 1974), with estimates of aboveground biomass ranging from 7.9 Mg ha^{-1} - 135.5 Mg ha^{-1}. However, the most recent study by Coronado-Molina et al (2004) was found to contain errors in the reporting of AGB values for their study sites and may not be considered a reliable source as a result.

There have also been a few studies that present the rate of aboveground biomass accumulation or aboveground woody biomass accumulation per year without stating the current AGB of the mangrove stands studied so cannot be used to calculate C content for these sites (Day et al, 1987; Pool et al, 1977; Castañeda-Moya et al, 2013).

As climate conditions change and new methods of calculating AGB are put into the literature, it is important to assess whether the AGB estimates created in these studies are still relevant and usable today and to produce an estimate of mangrove belowground biomass (BGB) for Southwest Florida. Therefore, this study aims to estimate the aboveground biomass of a Southwest Florida mangrove stand.

Methods

The study took place in the mangrove forests located around the Florida Gulf Coast University Vester Marine and Environmental Science Research Field Station in Bonita Springs, SW Florida (26° 19 N, 81° 50 W), approximately 1.75km from Estero Bay. Southern Florida is classified as subtropical savannah (Castañeda-Moya et al, 2013) with a defined dry season from December until May, followed by a wet season from June until November. Measurements were taken during late April and early May of 2019 during the start of the 2019 wet season. Daytime land

Figure 1: Aerial image of the study area, located at the Vester Field Station in Bonita Springs, Florida. Image taken from Google Maps (August 2019)

temperatures during the sampling period ranged from 19-31°C (66-88°F) and daytime surface water temperatures ranged between 25-28°C (77-82°F). The mangrove stand was comprised of 3 of the native species of mangrove: *Avicennia germinans* (Black mangrove), *Rhizophora mangle* (Red mangrove) and *Laguncularia racemosa* (White mangrove). *Conocarpus erectus* (Buttonwood) was not present in this area. On the edge of the stands, *R. mangle* was the most abundant with individuals of *A. germinans* and *L. racemosa* scattered along the edge of the water. As distance from the water increased*, A. germinans* became more prevalent but *R. mangle* remained dominant. Approximately 20m into the stand, *L. racemosa* became more dominant, with *A. germinans* becoming absent and *R. mangle* density reducing.

Canoes were used for transportation to study plots to produce minimal wake and enter

shallower areas of water. Twelve study plots of 5 x 5m were chosen through stratified sampling based on ease of access for collection of measurements for calculating AGB. For each stem within the study plots, stem species, tree height and diameter at breast height (DBH) were recorded using an inclinometer and measuring tapes (Donato and Kauffman, 2012).

To calculate aboveground and belowground biomass of the study sites, the common allometric equations from Komiyama et al (2005) was chosen due to the structure of the canopy. These equations were:

$$W(top)= 0.251\rho D^{2.46}$$

Where:

$W(top)$ = AGB

ρ = Species-specific wood density

D = DBH

$$W(R)= 0.199\rho^{0.899}D^{2.22}$$

Where:

$W(R)$= BGB

This circumvented the need for accurate estimation of crown coverage which is required for most other commonly used AGB allometric equations. Wood density values (ρ) for each species were taken from the Encyclopaedia of Life database. Normality was tested using Kolmogorov-Smirnov and Anderson-Darling tests and a One-way ANOVA was used to test for significant differences on

transformed AGB data whereas a Kruskal-Wallis test was used for BGB of the different species. In total, an area of 300m² was included in the study of the mangrove forests surrounding the Vester Field Station. In this area, 101 stems were recorded in total including 68 *R. mangle* stems, 14 *A. germinans* stems and 19 *L. racemosa* stems (Table 1).

Results

On the edge of the stands, *R. mangle* was the most abundant with individuals of *A. germinans* and *L. racemosa* scattered along the edge of the water. As distance from the water increased, *A. germinans* became more prevalent but *R. mangle* remained dominant. Approximately 20m into the stand, *L. racemosa* became more dominant, with *A. germinans* becoming absent and *R. mangle* density reducing. *R. mangle* was the dominant species in the area comprising 67.33% of the mangrove stems, whilst *A. germinans* and *L. racemosa* contributed 13.86% and 18.81% of the remaining total. Canopy height ranged from between 1.5 and 12.4m. Despite being the most dominant species, *R. mangle* had both the lowest average DBH and height when compared to both *A. germinans* and *L. racemosa* (Table 1). The AGB of individual study plots ranging from 4.9-104.2 Mg/ha^{-1}

Table 1: The density and distribution of each mangrove species at the Vester Field Station study site and the aboveground biomass (AGB) of each quadrat. *R. mangle* had the highest stem density of 68 and was the most dominant species, whereas *A. germinans* and *L. racemosa* had similar stem densities (14 and 19), but slightly differing distribution patterns, with *A. germinans* more common in plots closer to water. AGB of the 12 quadrats ranged from 4.904-104.215 Mg/ha^{-1} whilst BGB ranged from 2.366-34.153 Mg/ha^{-1}.

Quadrat	*A.germinans* individuals	*R.mangle* individuals	*L.racemosa* individuals	Total number of trees	Aboveground biomass (Mg/ha^{-1})	Belowground biomass (Mg/ha^{-1})
1	0	4	0	4	4.904	2.366
2	2	6	1	9	26.065	11.372
3	1	6	0	7	22.987	9.477
4	2	2	0	4	47.566	17.306
5	2	4	0	6	104.215	34.153
6	1	3	0	4	24.182	9.622
7	1	2	1	4	20.779	8.587
8	2	7	2	11	31.555	13.604
9	0	8	5	13	44.2	18.329
10	3	7	2	12	70.899	26.744
11	0	9	4	13	17.811	8.167
12	0	10	4	14	56.336	22.279
Total	14	68	19	101	471.499	187.006

Table 2: The averages and ranges of DBH, height and AGB of each mangrove species at the Vester Field Station study site. *A. germinans*, *R. mangle* and *L. racemosa* each contribute 44.66%, 17.92% and 37.42% of the total AGB of the site, which was calculated using the DBH of each stem in the study plots. *A. germinans* contributed 40.9% of the BGB with an average mass of 159.5kg BGB for the species whilst *R. mangle* and *L. racemosa* each contributed 21.75% and 37.35% with average BGB values of 17.5kg and 107.3kg respectively.

	A.germinans	*R.mangle*	*L.racemosa*
Average DBH (cm)	18.7	7.1	19
DBH range (cm)	6.4-49.9	2.7-17.7	9-40
Average height (m)	6.8	3.7	7
Height range (m)	3.6-12	1.5-10	1.8-12.4
Average AGB (kg)	451	37	278
AGB range (kg)	18.7-2908.9	2.6-262.4	33.9-1340.4
Average BGB (kg)	159.5	17.5	107.3
BGB range (kg)	9.7-926.4	1.6-105.6	16.7-460.8

with an average AGB of 39.3 Mg/ha^{-1} (Table 2). Even though *A. germinans* and *L. racemosa* had the fewest stems in the study area, 44.66% and 37.42% of the total AGB was comprised of *A. germinans* stems and *L. racemosa* stems respectively, whereas the more common *R. mangle* stems only comprised 17.92% of total AGB. The Anderson-Darling test found that the AGB data for all 3 species studied was non-normally distributed and was then transformed before the One-way ANOVA was performed. The results of the ANOVA showed there was no significant difference in the AGB of *A. germinans*, *R. mangle* and *L. racemosa*
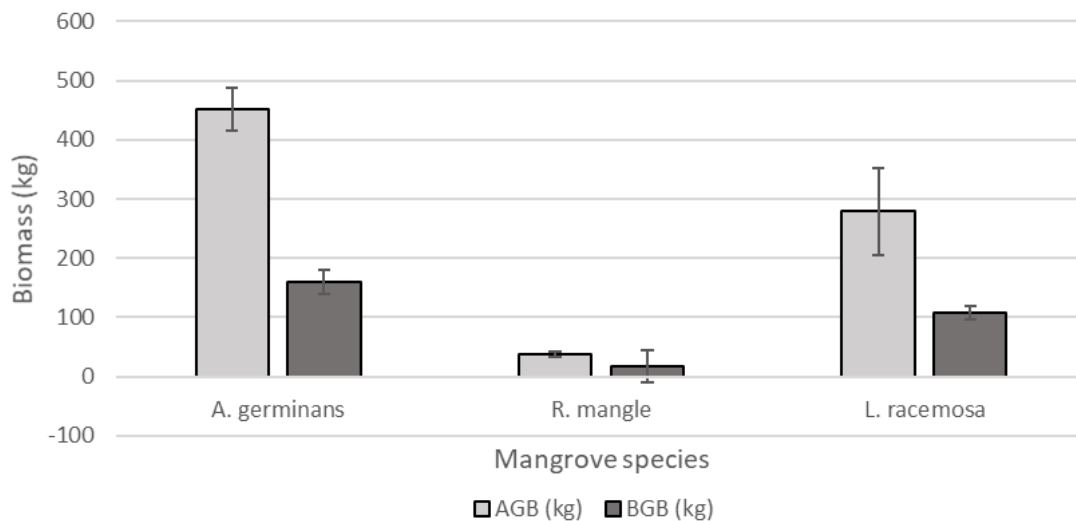

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Figure 2: The averages of aboveground and belowground biomass were analysed and there was a statistically significant difference between both *A. germinans* and *L. racemosa* when compared to *R. mangle* for both aboveground and belowground biomass ($p < 0.05$). While A. germinans had the lowest number of stems in the study area it had the highest average AGB and BGB at 451 kg and 159.5 respectively.

value=0.1, P-value=0.907). Separate ANOVAs were conducted to test for significant differences in AGB between *A. germinans* and *L. racemosa, A. germinans* and *R. mangle* and *L. racemosa* and *R. mangle* for future considerations in replanting projects where total biomass could be used a measure of success. However, these also showed no significant differences (F-value=0, P-value=0.970; F-value=0.08, P-value=0.774; F-value=0.14, P-value=0.706 respectively).

The BGB of the plots ranged from 2.37- 34.15 Mg/ha^{-1} with an average of 15.17 Mg/ha^{-1}. As with AGB, A. germinans contributed the most BGB with 40.9% compared to L. racemosa and R. mangle with 37.35% and 21.75% respectively. The Kolmogorov-Smirnov test showed that the BGB data was non-parametric (P<0.05) and therefore a Kruskal-Wallis was used. The results of the Kruskal-Wallis test showed significant differences in the BGB of the species (H value=46.401, df.=2, p value=0.00). When differences between species were tested for, it was found that there were significant differences in the BGB between R. mangle and A. germinans as well as R. mangle and L. racemosa (H value= 22.046, df.= 1, p= 0.00; H value= 32.982, df.= 1, p= 0.00). But there was no significant difference between L. racemosa and A. germinans for BGB (H value= 0.033, df.= 1, p= 0.855).

This study found the AGB of the study plots at the Vester Field station were consistent with the range of values given in previous studies on Floridian mangrove forests. The lowest value in prior research was 7.9 Mg/ha^{-1} from Lugo and Snedaker (1974) at an area of scrub mangrove in Florida. However, quadrat 1 produced a value of 4.9 Mg/ha^{-1}, making it the lowest AGB value for a Florida mangrove site. This is most likely due to the small number of *R. mangle* stems present at the site which was potentially in an earlier stage of succession. These trees each had wide reaching prop roots which would prevent the establishment of more stems within the study area. The site would also be inundated at high tide which would stop the establishment of any *A. germinans* or *L. racemosa* seedlings which are less tolerate of such conditions. The highest value from the study area was at quadrat 5, where despite having fewer stems than other study plots, the presence of the largest *A. germinans* stem found contributed disproportionately to the total AGB of the quadrat. The range of values also suggests the mangrove forest surrounding Vester field station is made up of two different forest structures described by Lugo and Snedaker (1974), fringe and basin forest structures.

When combined with the mangrove C content value of 44% from Ewe et al (2006) the estimated C standing stocks of the AGB plots ranged from 2.16- 45.86 Mg C/ha^{-1}, with an

average of 17.3 Mg C/ha^{-1}. As the individuals of *A. germinans* had on average the highest aboveground biomass values, the species also had the highest estimates of AGB carbon stocks ranging from 8.2- 1279.9 kg C. *L. racemosa* and *R. mangle* AGB C estimates ranged from 14.9- 589.8 kg C and 1.1- 115.5 kg C respectively. The value of mangrove carbon storage is defined here as the price per tonne of carbon stored through preservation of aboveground forest area instead of harvesting the wood as laid out in Richards and Stokes (2004). At $10 per tonne of carbon the average aboveground biomass of the study area would provide $173 per hectare for one-off payments for the carbon sequestered in the forest. These one-off payments are the most valuable for mangrove carbon as the annual accruement rates of carbon in mangrove habitats are negligible at <1 cent per hectare. This shows that for mangrove forests, the long-term sequestration potential of these sites is the most important for climate mitigation and for future carbon credit projects that include coastal wetlands.

There were fewer studies found for the belowground biomass of the mangrove forests in Florida, however the average value of 15.17 Mg/ha^{-1} was lower than the BGB range of 24.0±3.3- 46.7±7.5 Mg/ha^{-1} from Castañeda-Moya et al (2013). This may be due to several of the plots having lower densities of stems and therefore lowering the

average for belowground biomass over the study site. However, as these studies took place at several sites spanning Central and Southern Florida there may be site-specific variation in these estimates due to geographical differences between comparison sites and the study site of this project.

 Several of the quadrats had low AGB values within a range similar to those found by Ross et al (2001) in their study of mangroves at Convoy Point in Florida. Hurricane Andrew hit in 1992 and in 1995 the AGB recorded at a fringe forest directly in the storm's path and a dwarf forest opposite was 18 Mg/ha^{-1} and 16 Mg/ha^{-1} respectively. In 1997 the areas were re-studied, and it was found the AGB had increased in both plots, with the AGB of the fringe forest at 56 Mg/ha^{-1} and the AGB of the dwarf forest at 22 Mg/ha^{-1} (Ross et al, 2001). It was noted that in the mangrove stands at the study site, overturned and dead trees were present along with gaps in the canopy consistent with storm damage, which suggests that when Hurricane Irma made landfall in September of 2017 the mangrove stands were impacted by the category 4 storm (Ross et al, 2001; Smith et al, 2009). Although most values found at the site were similar to Ross et al (2001), quadrats 5 and 10 had larger values for AGB which were more consistent with Lugo and Snedaker (1974). Whilst the Florida mangrove scrub included in their study had a low AGB value, the range of values for fringe, riverine and overwash mangrove

stands was found to be 86.2-173.9 Mg/ha^{-1} (Lugo and Snedaker, 1974). It could be theorised that study plots where damage was observed could have had much higher values before Hurricane Irma. However, without any retrospective data for the area before the hurricane this theory could not be investigated.

The mangrove scrub forest studied by Lugo and Snedaker (1974) was described as similar to the site of Coronado-Molina et al (2004) at Taylor Slough, where the range of AGB values was low, ranging between 7.9-23.2 Mg/ha^{-1}. However, there are errors in this paper, including the AGB value range being listed as 7.9-21.2 Mg/ha^{-1} and incorrect values listed in their table for the Ross et al (2001) study. Despite these errors, the AGB values found are comparable to the data from several quadrats studied at Vester.

Mexico, Puerto Rico and Vietnam are found on similar latitudes to Florida, however the range of values for AGB have been found to be lower in Mexico and more similar to the mid-range AGB estimates in Puerto Rico, with the range of 6.14-12.52 Mg/ha^{-1} for AGB for Mexico (Day et al, 1987; Lopez-Portillo and Ezcurra, 1985) whereas Golley et al (1962) found that the AGB value for Puerto Rico was 62.9 Mg/ha^{-1}, but Vietnam which is slightly lower in latitude has a range of 31.88-170.65 Mg/ha^{-1}. On the other side of the equator Australia covers a range of latitudes including those similar to Florida and AGB values

varying from 7.1-128.4 Mg/ha^{-1} (Murray, 1985; Briggs, 1977; Goulter and Allaway, 1979). Countries closer to the equator such as Panama, Indonesia and Malaysia have higher AGB values (279.2 Mg/ha^{-1}, 169.1-436.4 Mg/ha^{-1} and 147-314 Mg/ha^{-1}) where there are less environmental limitations that would reduce the potential growth of mangrove forests (Golley et al, 1975; Komiyama et al 1988; Ong et al 1981).

2 key factors that influence the AGB stock of mangroves in Florida are temperature and frequency of storm events. Whilst hydrology and salinity are also key factors in the growth of mangroves, they are outside the scope of this project. Mangroves are a mostly tropical group of species that are sensitive to cold temperatures and stochastic freezing events, although different species show variation in their level of physiological tolerance and response to lower temperatures (Duke et al, 1998; Osland et al, 2017). These cold temperatures and freezing events can have several different effects on the physiology of mangroves, including reduction in AGB, lower metabolic and reproductive rates, and in cases of particularly hard frosts and low temperatures mortality can occur (Ross et al, 2009; Lovelock et al, 2016; Osland et al, 2017). In a study by Stuart et al (2007) comparing the growth of *A. germinans* in Florida, where winter temperatures are fairly mild, but frost events can occur, and *A. marina* in Australia where winter

temperatures on average are lower but there is less risk of frost, it was found that *A. germinans* had larger xylem vessel diameters, and showed more rapid growth than *A. marina* in Australia. Whilst these adaptations may be beneficial when the temperatures are mild in Florida, the mangroves are at a higher risk of mortality when frost events occur, with several incidences where winter frosts have caused major mortality events in the past few decades (Stevens et al, 2006; Quisthoudt et al, 2012). As mangroves move towards their poleward limits, incidence of frost and temperatures meeting the lowest absolutes that mangroves can survive reduces species richness and diversity as only scrub-like forests can tolerate such temperatures before the mangroves can go no further in their ecological range (Quisthoudt et al, 2012; Osland et al, 2017). In Florida, the most frost-tolerant of the mangrove species present is *A. germinans*, which can survive minimum temperatures of -6.7°C, which is why they are found further North than the other native species (Stevens et al, 2006).

It has been predicted that climate change could lead to increased hurricane wind strength and elevated sea surface temperatures, creating more intense and frequent tropical storm events (Scavia et al, 2002). Hurricanes are measured between category 1 and 5, with the number and strength of hurricanes varying greatly in consecutive years (Scavia et al, 2002). Whilst these storm events can have many impacts on mangrove forests, there are 3 categories that most damage falls under; damage from wind, storm surges and deposition and redistribution of sediments (Castañeda-Moya et al, 2010). Wind damage is the most common and visually obvious way hurricanes can impact mangrove forests. With the strong winds characterised by these storms, it is common for branches to be broken off, large stems to be blown over and for partial or total defoliation to occur (Smith et al, 2009). Whilst in some areas where hurricanes hit may be protected by topographic features such as mountains, Florida is a relatively flat state with few naturally occurring large features that could prevent the forests taking the brunt of the storms (Doyle et al, 1995; Castañeda-Moya et al, 2010). However, it has been found that whilst larger stems can see significant reductions to the average height of the foliage, it has been found that dwarf forests and shorter stems are likely to sustain little damage as a result of high winds (Doyle et al, 1995). It has also been found that whilst all Florida mangrove species are susceptible to high winds, the species most at risk of taking wind damage is *L. racemosa* (Doyle et al, 1995). This may be due to the fact that L. racemosa can grow to greater heights than *R. mangle* and *L. racemosa* does not have the characteristic thicker trunk associated with *A. germinans*.

Storm surges have 2 particular ways that they can impact the mangroves they hit. The first is that as the surge moves into the mangrove forest, the kinetic energy can cause stems to be pushed over, particularly larger ones (Smith et al, 2009). Smaller stems that lie under the storm surge are less likely to be uprooted, but if the water remains high for a prolonged period of time, the covered roots can lead to the mortality of the stem (Smith et al, 2009). A similar problem can occur with the deposition of sediment carried in the storm surge. This sediment can cover aerial roots and seedlings, preventing sufficient soil gas exchange, suffocating them and causing them to die (Smith et al, 2009). This is a particular risk where the storm surge is carrying fine sediment (Ellison, 1998). However, the sediments deposited as a result of a hurricane are actually an important factor that controls the potential growth of mangroves, especially around the South coast of Florida. The allochthonous mineral inputs caused by storm events are known to enhance the concentrations of phosphorous and lower the ratio of nitrogen to phosphorous, which are important limiting factors for mangrove growth (Castañeda-Moya et al, 2010). In South Florida there is also a very limited supply of carbonate in the terrestrial sediment, partially due to soil-building in the area being reliant on the biomass production of mangroves (Middleton and McKee, 2001). This carbonate is brought in by storm surges and is an important part of the biogeochemistry of the region as it is one of the few sources of P available to the mangroves (Koch and Snedaker, 1997; Castañeda-Moya et al, 2010). It has been found that mangrove mortality can occur many years after a storm event, as trees eventually die as a result of the damage they sustained during the hurricane, although some are able to re-sprout (Doyle et al, 1995), whilst gaps in the canopy are often filled by seedlings that were in the understorey of larger stems.

Whilst the biomass of Florida mangroves has been researched for use in calculating the carbon potential of mangroves in most countries where they are present, it is unclear as to how mangroves will be affected by the predicted effects of climate change. Atmospheric CO_2 concentrations have now exceeded 400 ppm as a result of anthropogenic activity since the beginning of the industrial revolution and this may be beneficial for many plant communities, including mangrove forests (Gilman et al, 2008; Field, 1995; Komiyama et al, 2008). It has been suggested that the increase in atmospheric CO_2 levels will result in increases in the growth and net primary productivity of many species of mangroves (Field, 1995; Komiyama et al, 2008). Farnsworth et al (1996) found under high CO_2 conditions total plant biomass of *R. mangle* seedlings increased by nearly 50% and maximum

photosynthetic rates were significantly higher than seedlings under ambient CO_2 conditions. But long-term exposure to the higher CO_2 conditions lead to lower photosynthetic responsiveness, suggesting downregulation was occurring overtime (Farnsworth et al, 1996).

However, it is unclear whether there would be any significant change on the ecological zonation and composition of mangrove forests due to this increase (Ball et al, 1997). It has also been reported that there will be an increase of annual global precipitation of 25% by 2050, although this will be seen unevenly around the world, with some areas experiencing a decrease compared to others (Houghton et al, 2005; Solomon et al, 2007). One of the regions expected to see an increase in precipitation is North and South America, which would mean that the mangroves in Florida would see more rainfall than in previous decades (Solomon et al, 2007). It was predicted by Duke et al (1998) that as the extra rain would result in increased biomass and biodiversity in mangrove forests as well as the colonization of previously unsuitable habitat. This increase in growth and diversity would be due to a higher supply of nutrients and fluvial sediments from the rain, as well as lower sulphate concentrations and reduced salinity (Field, 1995; Ellison, 2000). There would also be an increase the level of peat produced due to the greater input of freshwater into the

systems (Mörner, 1994; Snedaker, 1995). Whilst for poleward ecosystems the global increases in average temperatures are predicted to have extremely damaging consequences, for mangroves the rising global temperatures are likely to have several positive impacts. The IPCC has predicted that as the global surface temperatures rise, there will be reduced number of extreme cold events, which would lead to fewer frosts causing tree mortality at the poleward limits of mangrove forests (Solomon et al, 2007). The as extreme cold temperatures reduce in frequency, mangroves may be able to move further into habitat previously unsuitable due to climatic limitation and become more productive in already established areas, with increases in the rate of photosynthesis and alterations in the species compositions of mangrove forests and phenological patterns of reproduction (Field, 1995; Ellison, 2000; Solomon et al, 2007).

But these potential positives will be useless if mangrove forests are unable to adapt to rising sea levels or survive more extreme storm events. It has already been established that the rise in global sea levels is already occurring, although it is variable around the world. But several different models have predicted that this rise will accelerate in the decades to come (Thomas et al, 2004; Church and White, 2006; Solomon et al, 2007). There have already been papers that report that at present around the Western Caribbean and

Atlantic coastlines the mangrove forests are unable to maintain a rate of sediment deposition and surface accumulation that is above the current rate of change in relative sea level (Gilman et al, 2008; Cahoon et al, 2006; McKee et al, 2007). From the research conducted it has been determined that the most substantial non-anthropogenic cause of mangrove habitat loss in the future will be relative sea level change, which is estimated to cause the loss of 10-20% of global mangrove forests in the future (Gilman et al, 2006; Gilman et al, 2008). Another major threat to mangroves will be the predicted increase in the frequency and strength of tropical storm events. It has been theorised that as a result of climate change tropical cyclones will become more intense (Emanuel, 1999; Webster et al, 2005; Mann and Emanuel, 2006). If sea surface temperatures rise, it is likely that hurricane wind strength would increase and cause an increase the height and strength of storm surges as a result (Houghton et al, 2001; Solomon et al, 2007). But it has also been theorised that because during El Niño years there are fewer Atlantic storms due to the raised sea surface temperatures in the Pacific, that there would be a decrease in the frequency of tropical storms as sea surface temperatures rise globally (Pielke and Landsea, 1999). However, even if hurricanes became less frequent, they would be far more severe as the storm surges would be higher and the winds would be

stronger (Scavia et al, 2002; Gilman et al, 2008). These storms would be much more likely to cause defoliation, branch breakage and tree mortality, but they could also cause erosion, peat collapse and soil compression (Smith et al, 1994; Baldwin et al, 2001; Sherman et al, 2001; Cahoon et al, 2006; Piou et al, 2006). Potentially this could lead to a positive feedback loop, where erosion and loss of smaller stems caused by rising sea levels could result in more damage from hurricanes over a larger area, where re-establishment of stems would be prevented by further changes in sea level. Where sea level or storms cause mass tree mortality with little to no chance of regeneration or establishment of saplings, it is possible that there may be permanent ecosystem conversion, releasing large volumes of CO_2 into the atmosphere and furthering the potential future damage climate change may cause (Cahoon et al, 2003). One factor that could make global temperature increase detrimental to mangroves is the potential increase in algal bloom frequency and size. In other parts of the world where mangroves are present, large diebacks of mangroves have been attributed to algal blooms caused by eutrophication (Schaffelke et al, 2005). These blooms effectively smothered adult stems by blocking their aerial roots, as well as preventing growth of seedlings (Schaffelke et al, 2005). As global sea surface and increased surface temperatures increase, efficiency and

rate of photosynthesis in algae will increase, leading to larger algal blooms that are less regulated by cooler temperatures. With increasing algal bloom events in Florida, it could be theorised that local mangrove species are experiencing similar problems.

The effects of climate change present a significant challenge to the preservation of coastal habitats and the conservation of the species dependent on these areas. But mangroves have the potential to be used to stabilize shorelines of their habitats due to their ability to trap sediment in their roots, leading to the formation of land (Romañach et al, 2018).

The first stage would be for the chosen area for restoration to be studied for any reason why natural restoration may not have occurred, such as water pollution (Erftemeijer and Lewis, 1999). The site would need to also be evaluated for ecologically or economically important species that may be negatively impacted by the mangrove habitat restoration.

For initial stabilization of coastal land in Florida, the species *R. mangle* would be the most effective as it grows faster than the other native species and its' aerial roots would be at lower risk of being smothered by sediment brought in through tidal or storm activity (Doyle et al, 1995; Erftemeijer and Lewis, 1999). For long term conservation goals, the larger *A. germinans* would be able to grow larger root structures that would support newly formed soil. *A. germinans* is also less likely to be impacted by high winds and storm events than *L. racemosa*.

For future work, a follow-on study will be designed to use the estimated AGB this study produced to calculate the carbon content of the standing mangrove stock. Standing C stocks data will contribute to informing policy decisions and creating estimates of mangrove carbon sequestration potential and mangrove greenhouse gas emissions for national reports as these stocks create long-term C storage when undisturbed. These results will allow for the creation of more accurate financial incentive estimates for the inclusion of mangrove forests in international mitigation policies such as Joint Implementation and REDD+ projects which at present exclude mangrove forests. Inclusion of mangroves in mitigation schemes could be key to allowing developing countries to access the green economy and create sustainable economic growth by preserving naturally occurring resources.

Conclusions

This study found the aboveground biomass range of 4.904-104.215 Mg/ha^{-1} in the mangrove stands around the Vester Field Station were similar to those found at other sites in Florida. However, as there was some variation it would be advised that AGB should

be calculated separately at any study site, rather than trying to apply values from nearby sites due to potential error this may cause. AGB of Florida mangroves has been found to be most affected by temperature and storm events compared to forests in other areas of the world. However, as the effects of climate change become more apparent, mangroves have an uncertain future highly dependent on whether or not they can keep up with the rise in sea level or survive extreme storm events that are predicted to become more frequent. But there is still potential for mangroves to be used to help mitigate increases in atmospheric CO_2 concentrations and shoreline erosion. Future studies must look at standing carbon stocks and greenhouse gas emissions to create mitigation plans and environmental policies surrounding carbon storage.

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