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*SIT Study Abroad*

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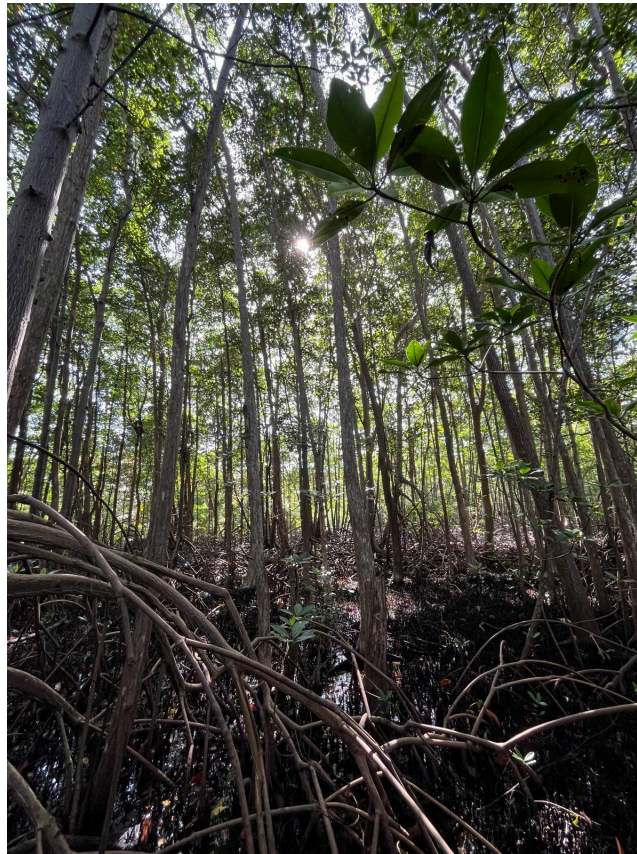
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**Variations in salinity and forest structure between marginal and cloistered mangrove forests in Boca del Drago, Bocas del Toro, Panama.**



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ISPR 3000: Independent Study Project  
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## Abstract

Mangroves are incredibly important ecosystems that have an outsized impact on their surroundings, providing crucial ecosystem services like carbon sequestration and coastal protection. At the same time, mangroves are some of the most threatened habitats due to their valuable position along tropical coastlines. Globally, mangrove forests are deforested and degraded by aquaculture, tourism developments, urbanization, and overharvesting. Due to high rates of mangrove loss, mangrove deforestation is now illegal in Panama, though this legislation is inconsistent and allows for exceptions, especially in tourism developments. This study aims to probe the impacts of localized coastal developments on a small mangrove forest in Boca del Drago, Bocas del Toro, which is obstructed from the ocean by a manmade walking path. Measurements of salinity, seedling density, height, diameter at breast height (DBH), and species composition were taken for five plots and compared to previous measurements taken in a nearby undisturbed mangrove forest. Analysis revealed that average height, DBH, prevalence of the mangrove species *Laguncularia racemosa*, and salinity were significantly greater in the cloistered plot. In combination, these results suggest that the walking path obstructs seawater input, leading to completely freshwater conditions within the forest. The repercussions of this are uncertain, but could include population dynamic changes leading to a reduction in the ecosystem services provided by this forest. It is important to understand how coastal developments directly and indirectly affect mangrove forests, especially as these ecosystems become increasingly important with climate change.

## **Acknowledgments**

I would like to give enormous thanks to Flora Schwartzman Miles for her invaluable support and companionship throughout our many improvised meals, post-mangrove debriefs, and hours of bus rides, and to Zahara Goldkin bringing their cheerful company and delicious culinary inspiration to the last few days of our stay in Boca del Drago. Thank you to Yariza Jiménez for coordinating my housing for the duration of my independent study, and to Señora Juany and the staff at Hospedaje Yarisnori and Drago Beach House for accommodating me during my long stay in Boca del Drago.

Many thanks to Dr. Eric Manzané-Pinzón for providing guidance in the development of my research question and methods, and to Aly Dagang for helping me develop my focus for this project and for greatly facilitating my investigation by connecting me with past research conducted in the area. Thank you to Yuritza Lara for providing me with the equipment needed to complete this study. Finally, huge thanks to Ben Dwyer for giving me access to his ISP data, and thank you to all of my classmates for graciously providing me with their data, and also for being wonderful people and making this semester as amazing as it was.

# **Introduction**

## **Mangroves Globally**

Worldwide, mangroves provide a wealth of ecosystem services that support, protect, and enrich communities, including timber provisioning, nutrient cycling, coastal protection, carbon sequestration, and recreation (Bimrah et al., 2022). Estimations of the economic value of these ecosystem services vary, but range as high as \$50,000 per hectare per year (Mukherjee et al., 2014). Mangroves also play key roles in ecological resilience and connectivity, providing food for a great number of facultative terrestrial species (Rog et al., 2017), nursery habitat for commercially and ecologically important fish species (Sandilyan & Kathiresan, 2012), and habitat for resident and migratory bird species (Aloysius et al., 2023). Furthermore, current research continues to uncover additional benefits provided by mangroves in the face of climate change, including their potential role as protective environments for corals (Stewart et al., 2022) and in bolstering the diversity of fish communities on degraded reefs (Seemann et al., 2018). This developing understanding of mangrove services underscores the inadequacy of our current valuation of mangroves, indicating the need for further research and increased prioritization of conservation.

Despite the clear importance of mangroves, mangrove forests are under threat globally due to direct and indirect effects of anthropogenic activity. Mangroves are a frequent target of deforestation and degradation, both because of the economic value of their natural resources and also because of their valuable position along sheltered tropical shores. Products like timber, lumber, salt, and seafood have historically been extracted from mangroves, leading to wide-scale deforestation and overfishing (Borges & Ferreira, 2019). Mangroves are also destroyed for the construction of tourism destinations, the installation of urban infrastructure, and the creation of ponds for aquaculture. Shrimp aquaculture is particularly harmful as a major source of deforestation, sedimentation, and nutrient pollution in mangrove forests. Upstream activities like urban development, industrial development, agriculture, and hydrologic alterations can produce pronounced effects on nutrient and water regimes within mangrove forests (Borges & Ferreira, 2019; Urrego et al., 2018). As a result of these various pressures, global mangrove cover has declined by about 38% as of 2020 (Bunting et al., 2022; Hagger et al., 2022).

## **Mangroves in Panama**

Panama receives an estimated two billion USD from mangrove ecosystems each year (ANAM-ARAP, 2013), comparable to the reported 2.5 billion USD that the Panamanian government received from the Panama Canal in 2023 (Autoridad del Canal de Panamá, 2023). Despite this valuation, Panama has experienced a 43% decrease in mangrove area since 1969, from 320,000 ha to around 170,000 ha (ANAM-ARAP, 2013). In Panama, mangroves are legally protected by the Ministerio de Ambiente, with the purported intention of conserving mangroves and their ecosystem services. Among other measures, legal authorization is required for development in mangrove forests and hefty fines are imposed for illegal deforestation. However, the legislation surrounding mangroves is poorly enforced and contains multiple exceptions, allowing for the continuation of mangrove deforestation both legally and illegally (Chamberland-Fontaine et al., 2022). The government's ability to selectively approve tourism developments in areas forested by mangroves is especially concerning, as tourism is becoming

an increasingly impactful market in Panama. In 2023, international tourism in Panama increased by almost 30% (Autoridad de Turismo de Panamá, 2023). Among other tourism hotspots in Panama, the mangrove-rich province of Bocas del Toro lies at risk in the conflict between economic and environmental priorities.

## **Bocas del Toro**

The province of Bocas del Toro is located in the northwest of Panama, bordered to the north by the Caribbean Sea, to the east by the Comarca Ngäbe-Buglé, to the south by the province of Chiriquí, and to the west by Costa Rica and the Comarca Naso Tjër Di. Bocas del Toro has a land surface area of 4,654 km<sup>2</sup>, comprising 4,368.9 km<sup>2</sup> of contiguous mainland and 285 km<sup>2</sup> of islands, known as the Bocas del Toro Archipelago (Instituto Nacional de Estadística y Censo Panamá, n.d.). The entire region of Bocas del Toro is classified as tropical moist forest under the Holdridge life zones system (Solano, 2021). Average monthly temperatures vary between 25.6-27.3°C, with around 3350 mm of rainfall per year (Instituto de Meteorología e Hidrología de Panamá, n.d.). With such low temperature variability throughout the year, seasonality is determined by patterns of rainfall. Being on the Caribbean side of Panama, Bocas del Toro receives rainfall throughout the entire year, but drier seasons occur between January and April, and during the months of September and October.

The Bocas del Toro Archipelago was formed by sea level rise 6-9 million years ago, possibly caused by the extension of the tectonic plate underlying Panama and Costa Rica (O'Dea et al., 2016). Today, the archipelago is composed of nine main islands and numerous mangrove cays, all lying well below 500 masl (Olson, 1993). The position of the Bocas del Toro Archipelago creates two major bodies of water: the Bahía del Almirante between the mainland and the archipelago, and the Laguna de Chiriquí between the archipelago and the Valiente Peninsula to the east. These bodies of water are sheltered by the surrounding land structures, so tidal ranges are low, while currents are determined by local factors and thus are highly variable (Guzman et al., 2005).

Around 96% of the archipelago is covered by tropical forest and wetlands, with relatively minimal land clearing for agriculture or construction (Brandt et al., 2023). As of 2010, the Bocas del Toro Archipelago contained 13.76 km<sup>2</sup> of mangrove forest (Bunting et al., 2018), which is about 5% of the archipelago's surface area. The mangrove forests in Bocas del Toro are classified as dwarf forests with canopy heights between 1-5 meters, overwhelmingly dominated by the species *Rhizophora mangle*, with other mangrove species being quite rare (Baca et al., 2014; Guzman et al., 2005; Lovelock et al., 2005; Stewart et al., 2022). While large-scale deforestation is not currently a potent problem, Bocas del Toro is still at risk of deforestation with increasing pressure from tourism. The effects of localized coastal development impact the physical conditions of the mangroves, leading to changes in ecological conditions. Historic disturbances among Panamanian mangroves have resulted in areas where the species *Laguncularia racemosa* is atypically dominant (Benfield et al., 2005), and elsewhere in the Caribbean, dominance of *Avicennia germinans* or the fern *Acrostichum aureum* accompany disturbance (Urrego et al., 2018). Moreover, anthropogenic changes to the hydrologic regime of mangrove forests have even resulted in total displacement of mangrove forest by terrestrial species. Coastal development and hardening also affects mangrove population dynamics, impacting propagule dispersion and landward recruitment (Timm & Araújo, 2018). It is important to understand how anthropogenic activities might produce unintentional alteration or



degradation within mangrove forests, as these changes may affect the capacity of mangrove forests to provide their crucial ecosystem services.

This study aims to probe the indirect effects of smaller-scale coastal development on mangrove forests by examining a potentially impacted forest. The forest covers about 2 ha between Playa Boca del Drago and Playa Estrella in Boca del Drago, Isla Colón, Bocas del Toro, about 800 meters away from the mangrove forests in Ground Creek. A maintained walking path runs along the seaward edge of the forest (Fig. 1), restricting water movement between the ocean and forest to a single small creek (Fig. 2). Records of past disturbances in this area could not be found, but between Playa Boca del Drago and Playa Estrella, land clearing for agricultural purposes occurred as recently as 2004 (Google Earth Pro 7.3.6.9796, 2024). This forest is termed a “cloistered” forest for the purpose of this study, because it is partitioned off from the ocean by the walking path. In comparison, the mangrove forest along Ground Creek is termed as a “marginal” forest because the edge of the forest lies directly on the ocean. By comparing the parameters of species composition, tree height, diameter at breast height (DBH), seedling density and salinity to those of the marginal mangrove forest, the study will determine whether this cloistered mangrove forest exhibits significantly different population dynamics than the typical mangrove forests that are not obstructed from the ocean.



**Figure 1. The walking path.** This path runs along the entire edge of the cloistered mangrove forest. *R. mangle* and *L. racemosa* are visible on the right, while *Cocos nucifera* grows along the edge of the water.



**Figure 2. The intersection between the walking path and the creek.** This creek serves as the only connection between the mangrove forest (right) and the ocean (left).

## Research Question

Is there a significant difference in forest structure (represented as DBH, canopy height, species composition, and seedling density) between a marginal mangrove forest located along Ground Creek and a cloistered mangrove forest located between Playa Boca del Drago and Playa Estrella on Isla Colón, Bocas del Toro, Panama?

## Methods

### Site and Plot Establishment

Data was collected from three study sites situated in the Boca del Drago corregimiento on Isla Colón, Bocas del Toro (Fig. 3). Site 1 (9°24'44" N, 82°19'49" W) represents the cloistered mangrove forest between Playa Boca Del Drago and Playa Estrella, distinguished by a walking path which runs along the edge of the forest, inhibiting water movement between the ocean and the forest apart from a small creek. Site 2 (9°24'08" N, 82°18'47" W) and Site 3 (9°24'06" N, 82°18'34" W) represent the marginal mangrove forest located in Ground Creek.



**Figure 3. Map of the three study sites.** Sites with gray markers represent local landmarks for reference. Modified from (Google Earth 10.52.0.0, 2023).

Site 1 consisted of five five-meter radial plots with central coordinates (9°24'44" N, 82°19'49" W), (9°24'43" N, 82°19'48" W), (9°24'44" N, 82°19'50" W), (9°24'43" N, 82°19'49" W), and (9°24'42" N, 82°19'47" W). Using satellite imagery, plots were selected to form an approximate 30 m x 30 m grid (Fig. 4). To establish each plot, the central point was located *in situ* using GPS and marked with a piece of rebar. A four-armed PVC structure was placed on top of the rebar and aligned with the four cardinal directions to divide the plot into four quadrants. Data was collected within a five meter radius of the central point, creating a circular plot with a total area of 78.54 m<sup>2</sup> (Kauffman & Donato, 2012). Data was collected over the period of April 10-14, 2024.





**Figure 4. Map of Plots 1-5.** Plots are numbered in order of completion of data collection.  
Modified from (Google Earth 10.52.0.0, 2023)

Data from Sites 2 and 3 was previously collected by SIT students over the periods of February 19-20, 2024 and February 26-28, 2023 respectively. Data was collected using the same radial sampling technique as used in Site 1. Precise coordinates for plots could not be obtained, but Site 2 contained six plots and Site 3 contained nine plots.

### Salinity

Salinity was measured using an optical salinity refractometer. The instrument was first calibrated using fresh water. Standing water was sampled from the central point of each plot in Site 1. Three drops of the sample were pipetted onto the prism and the plastic cover was closed to evenly distribute the sample over the prism. The salinity reading was viewed through the eyepiece, with the border between the blue and white fields indicating the salinity of the sample in parts-per-thousand (‰). The darkness of the understory necessitated shining a flashlight into the prism to view the salinity reading. Using the same methodology, a classmate took salinity readings within Site 2, at a distance of 10 m from the shore along five parallel transects.

### Mature Trees

For the purposes of this study, “mature” mangrove trees were defined as any individual with DBH > 1.0 m. This classification groups together saplings and adult trees for comparison against seedlings. DBH, height, distance from center, and angle relative to North were recorded for each mature tree within Plots 1-5. Distance from center was measured in meters using a reeled measuring tape, where one end was tied to a stake placed at the central point and the other end was pulled to the base of the tree, ensuring that the measuring tape formed a straight, taut line. Angle relative to North was measured using a compass, aligning the compass with the straight line between the tree and the central point, with the direction of travel pointing towards the tree. These two measurements combined to give the exact position of each tree within the

plot. DBH was measured using a calibrated diameter tape at breast height, which was standardized as all trees were measured by the same observer. If trees were split into multiple trunks at breast height, DBH for each trunk was measured and summed (Snedaker & Snedaker (Eds.), 1984). For trees above four meters, height was measured using the Nikon Prostaff 1000 rangefinder, standing directly under the portion of the canopy being measured to avoid inaccurate readings (Snedaker & Snedaker (Eds.), 1984). The observer's height at eye level was summed to each reading to properly reflect tree height. Trees below four meters were measured using a reeled measuring tape, with measurements rounded to the nearest meter to match the output of the rangefinder.

## **Data Analysis**

Analysis was performed in the Spyder IDE using the SciPy and pandas libraries. The Shapiro-Wilk normality test was used to analyze the distribution of the salinity and seedling data sets because  $n < 50$  in both cases, while the Kolmogorov-Smirnov normality test was used for the mature tree data set because  $n > 50$  (Mishra et al., 2019). When comparing across different sites, significant differences between normally distributed variables were tested using an independent samples t-test (Thukral et al., 2023), while the Kruskal-Wallis test was used when variables had non-normal distributions (Dodge, 2008). All within-site comparisons were conducted using the Kruskal-Wallis test because all variables had non-normal distributions. Correlation analysis was performed using the Pearson correlation coefficient (Schober & Schwarte, 2018) and Ordinary Least Squares regression (Fomby et al., 1984).

## **Ethics**

The most significant ethical concern involved in this research was the destructive impact of moving through the mangrove forest. It was nearly impossible to walk through the forest without stepping on prop roots, pneumatophores, or seedlings. Nonetheless, damage was reduced by working strategically to minimize movement, walking slowly and carefully, and retracing steps. Propagule data was also intentionally collected last, serving as a further incentive to avoid stepping on propagules. It was also very important to be careful of where hands and equipment were placed, not only for personal safety but also to avoid crushing the many snails, crabs, and insects that live amongst the trunks and prop roots. Use of sunscreens and insect repellants on the hands was also avoided to prevent introducing harmful chemicals into the environment. After data collection was complete, all introduced materials were removed from the site and flagging tape was taken down.

As this study did not involve human subjects, submission to the School for International Training's Internal Review Board only entailed verification of this point. Consent was obtained from all authors whose data was used in this study.

## Results

### Characterizing the Cloistered Site

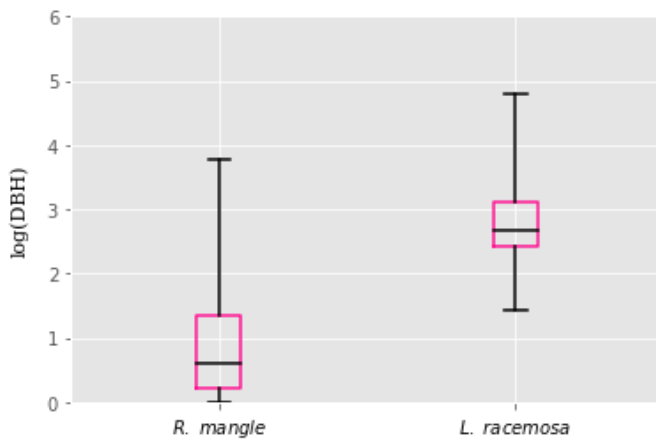
In total, 127 mature trees were measured within Site 1, of which 83 were *R. mangle* and 44 were *L. racemosa*. Site 1 formed a mosaic of *L. racemosa*-dominated canopy to the northwest (Plots 1 & 3), *R. mangle*-dominated canopy on the edges of the creek (Plots 2 & 4), and a mixed canopy to the southeast (Plot 5) (Fig. 4). The maximum canopy height lay between 15-20 meters in all five plots, and mean mature tree density was  $0.32 \pm 0.18$  trees/m<sup>2</sup> (Table 1).

**Table 1. Some descriptive statistics for Site 1.**

<b>Metric</b>	<b>Mean</b>	<b>Standard Deviation</b>	<b>Maximum Value</b>	<b>Minimum Value</b>
<b>Seedling Density (seedlings/m<sup>2</sup>)</b>	2.21	2.31	5.98	0.33
<b>Mature Tree Density (trees/m<sup>2</sup>)</b>	0.32	0.18	0.53	0.09
<b>All Species DBH (cm)</b>	9.44	14.27	119.4	1.0
<b>All Species Height (m)</b>	8.47	6.39	20.0	0.5
<b><i>R. mangle</i> DBH (cm)</b>	4.04	6.68	44.0	1.0
<b><i>R. mangle</i> Height (m)</b>	5.26	5.44	20.0	0.5
<b><i>L. racemosa</i> DBH (cm)</b>	19.61	18.69	119.4	4.2
<b><i>L. racemosa</i> Height (m)</b>	14.52	2.41	19.0	8.0

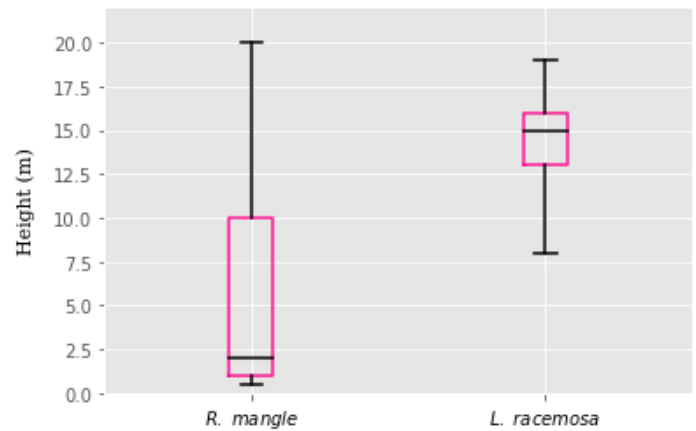
*L. racemosa* was present in all plots aside from Plot 2, and on average, *L. racemosa* DBH and height were greater than those of *R. mangle* ( $p = 2.47 \times 10^{-16}$ ,  $p = 6.48 \times 10^{-14}$ ) (Fig. 5, 6). The maximum *R. mangle* DBH (44.0 cm), maximum *R. mangle*/overall height (20.0 m), and greatest seedling density (5.98 seedlings/m<sup>2</sup>) were recorded in Plot 2. *R. mangle* and *L. racemosa* seedlings were both present in each plot, and on average, *R. mangle* seedling density was higher than *L. racemosa* seedling density ( $p = 0.0088$ ).

**DBH of Cloistered *R. mangle* and *L. racemosa***



**Figure 5. DBH by species in Site 1.** DBH of *L. racemosa* was significantly larger than DBH of *R. mangle* ( $p = 2.47 \times 10^{-16}$ ). Note that DBH data has been log- transformed for ease of visualization.  $n = 83$  for *R. mangle*,  $n = 44$  for *L. racemosa*.

**Heights of Cloistered *R. mangle* and *L. racemosa***

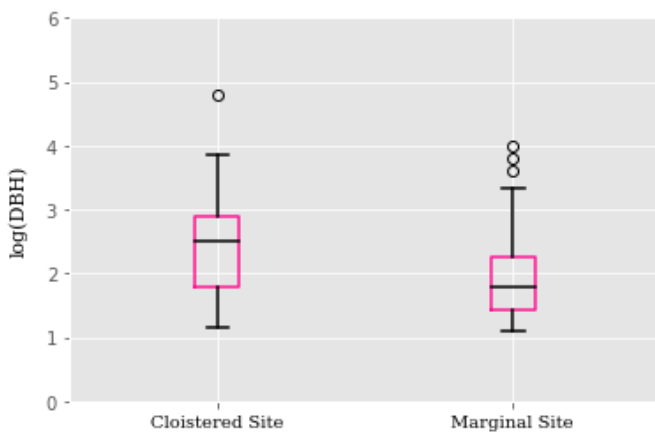


**Figure 6. Height by species in Site 1.** *L. racemosa* were significantly taller than *R. mangle* ( $p = 6.48 \times 10^{-14}$ ).  $n = 83$  for *R. mangle*,  $n = 44$  for *L. racemosa*.

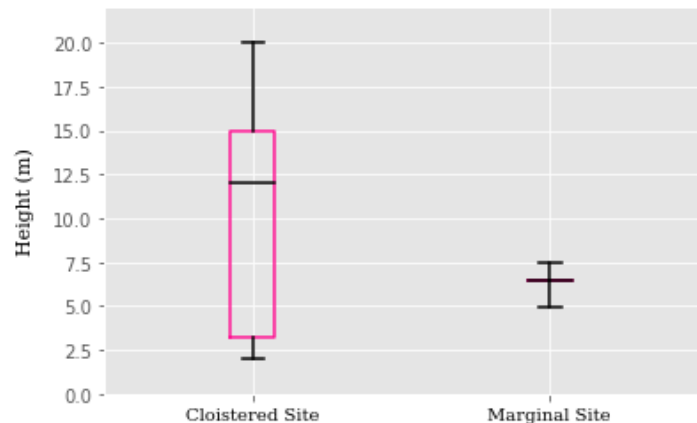
## Comparing Cloistered and Marginal Mature Mangroves

Cloistered and marginal sites were compared independently of mangrove species because much of the data from the marginal sites did not have species attribution. Comparing the entire data sets, there was no significant difference in mature mangrove DBH or height between cloistered and marginal sites ( $p = 0.27$ ,  $p = 0.059$ ). However, when the cloistered data set was filtered to conform to the same criteria as the marginal data set (excluding samples with  $\text{DBH} < 3.0$  cm,  $\text{height} < 2.0$  m), both DBH and height were significantly greater in the cloistered site ( $p = 1.01 \times 10^{-7}$ ,  $p = 4.39 \times 10^{-9}$ ). It is worth noting that filtering the data reduced the cloistered DBH sample size by 44% (Fig. 7).

**DBH in Cloistered and Marginal Mangrove Forests      Tree Height in Cloistered and Marginal Mangrove Forests**



**Figure 7. Mature mangrove DBH in cloistered and marginal sites (filtered).** DBH was significantly larger in the cloistered site ( $p = 1.01 \times 10^{-7}$ ). Note that DBH data has been log transformed for ease of visualization.  $n = 71$  for cloistered site,  $n = 251$  for marginal site.



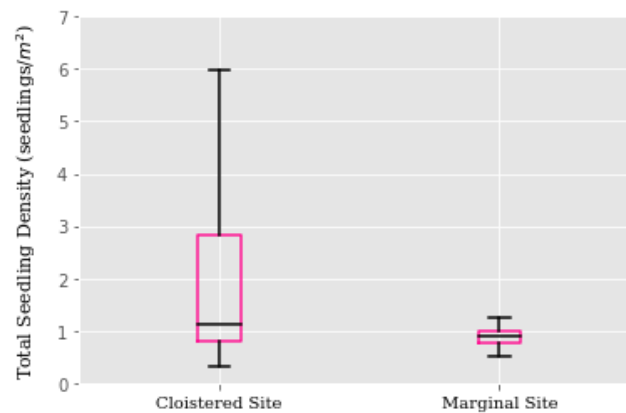
**Figure 8. Mature mangrove height in cloistered and marginal sites (filtered).** Mature trees were significantly taller in the cloistered site ( $p = 4.39 \times 10^{-9}$ ).  $n = 102$  for cloistered site,  $n = 200$  for marginal site.

A wider range of height (19.5 m) was seen in the cloistered site due to differences in data collection methods (Fig. 8). For five of the nine plots in Site 3, heights above 4.88 m were not properly represented due to equipment limitations, so the true mean height was obtained and attributed to previously collected data post factum (Dwyer, 2023). Thus, height data for the marginal site is representative of central tendency but not variation.

## Seedling Density

The maximum total seedling density was 5.97 seedlings/m<sup>2</sup> in the cloistered site and 1.25 seedlings/m<sup>2</sup> in the marginal sites, though there was no significant difference in total seedling density between the sites ( $p = 0.19$ ) (Fig. 9). However, there were significantly more *L. racemosa* seedlings in the cloistered site ( $p = 0.0027$ ).

**Seedling Density in Cloistered and Marginal Mangrove Forests**



**Figure 9. Total seedling density in cloistered and marginal sites.**  
There was no significant difference in total seedling density between sites ( $p = 0.19$ ).  $n = 5$  for cloistered site,  $n = 6$  for marginal site.

**Table 2. Seedling counts for Site 1.**

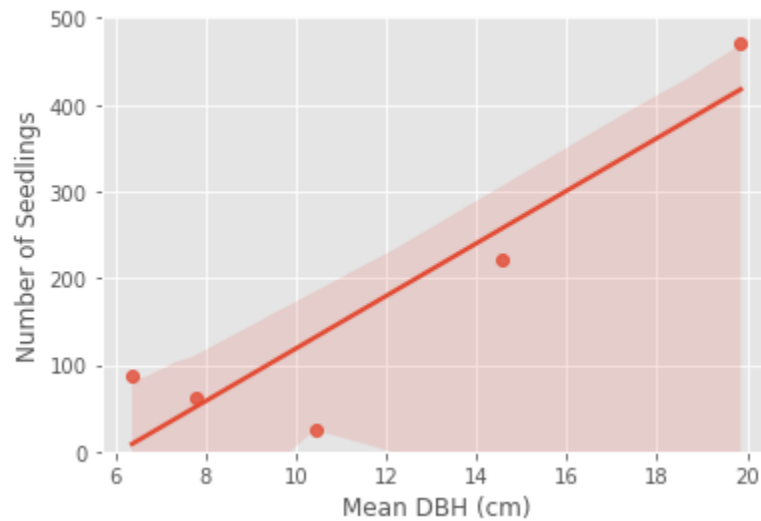
Plot	Number of Seedlings		
	<i>R. mangle</i>	<i>L. racemosa</i>	Total
1	66	21	87
2	469	1	470
3	41	21	62
4	220	2	222
5	23	3	26

94% of the seedlings observed in Site 1 were *R. mangle*, with the highest number of *L. racemosa* observed in Plots 1 and 3 (Table 2). Within Site 1, mean DBH per plot was correlated



with the number of seedlings observed in each plot ( $r = 0.91$ ,  $p = 0.03$ ). Figure 10 shows the linear regression of this relationship.

**OLS Regression of Mean DBH and Number of Seedlings in Cloistered Plots**



**Figure 10. Linear regression of mean DBH and number of seedlings per plot in Site 1.** Using an Ordinary Least Squares method with a confidence interval of 95.  $n = 5$ .

## Salinity

Salinity was recorded as ‰ (parts-per-thousand) for all plots in Site 1 (Table 3). Isolated samples of the creek and the adjacent ocean yielded salinity readings of 0‰ and 27‰ respectively. Salinity ranged between 15-20 ‰ for sampled points in Site 2, which was significantly higher than Site 1 ( $p = 0.0047$ ).

**Table 3. Salinity readings for Sites 1 and 2**

Plot	Salinity (‰)	
	Site 1	Site 2
1	0	15
2	0	12
3	0	20
4	0	20
5	0	20
Creek	0	n/a
Ocean	27	n/a

## Discussion

As a whole, the results of this study reveal the atypicality of the cloistered mangrove forest in Site 1. When compared against the marginal sites and against existing literature, mature mangroves in the cloistered site were taller and broader than is expected for the region, and *L. racemosa* was significantly more abundant (Baca et al., 2014; Guzman et al., 2005; Lovelock et al., 2005; Stewart et al., 2022). The cloistered site also did not exhibit typical patterns of zonation along salinity and inundation gradients, but instead resembled an ecological mosaic with both *L. racemosa* and *R. mangle* present in almost all plots.

This study was descriptive, not experimental, so causation cannot be established for any of these observed differences. However, the data collected on salinity was unexpectedly elucidating, suggesting possible explanations for the irregular characteristics of the cloistered site. Though limited in their scope, the salinity readings from Site 1 provide an understanding of the water regime within the site. The samples from all five plots and the creek revealed salinity readings of 0‰, suggesting that this forest exists in entirely fresh water. These samples were taken on multiple different days with different weather conditions, with salinity reading as 0‰ on both rainy and dry days, indicating that these results were not distorted by dilution from heavy rainfall.

The creek sample is particularly interesting, as it points towards possible explanations for the unique ecology and hydrology of this forest. This water salinity sample was taken where the creek intersects with the walking path at the edge of the mangrove forest. Thus, a reading of 0‰ implies that even though the adjacent ocean exhibited a salinity reading of 27‰, seawater is not flowing far enough into the creek to reach the mangrove forest. In this way, the creek might be acting exclusively as an outlet and forest would then be fed solely by rainwater, groundwater, or bodies of water further inland. Very little research exists on freshwater mangroves, as these systems are likely extremely rare due to lack of competitive ability. Some studies have described naturally occurring freshwater mangroves, though in comparison to the cloistered site, these forests cover wider areas, are dominated by *R. mangle*, and exist along large, connected bodies of water (Aburto-Oropeza et al., 2021; Bernardino et al., 2022). In the case of the cloistered site, these freshwater conditions could explain the larger average height and DBH of mature trees, as salinity stress is a growth-limiting factor for mangroves (Perri et al., 2023). Similarly, the uncharacteristic prevalence of *L. racemosa* and the lack of a typical zonation pattern could be a result of the site's lack of salinity, as the fast growing *L. racemosa* can outcompete *R. mangle* in areas with low salinity (Schaeffer-Novelli & Cintrón, 1986). Over time, freshwater conditions and lack of habitat connectivity could lead to the displacement of *R. mangle* by *L. racemosa*, or even the displacement of both species by terrestrial plants (Urrego et al., 2018).

Genetic isolation is another possible consequence of the cloistered site's occlusion from the ocean, as a lack of seawater input means that propagules from other populations cannot enter the forest. As a small population, the forest is already at risk for low genetic diversity (Albrecht et al., 2013). Furthermore, *R. mangle* and *L. racemosa* are considered self-compatible in some or all cases (Landry & Rathcke, 2007; Nadia & Machado, 2014), and for *R. mangle*, self-pollination rates can be as high as 95% (Albrecht et al., 2013). The strong correlation between seedling count and average DBH within the cloistered plots could mean that larger, older trees dominate propagule production, leading to a population with a high rate of shared parentage. In

combination, these factors create an increased risk for inbreeding depression within the cloistered forest, which could reduce the population's overall resilience.

These insights into the potentially detrimental ecological and genetic consequences of the occlusion of this site call for more extensive investigation of the forest's water regime, population dynamics, and genetic health. Time and personnel limitations restricted the scope of data collection, so it is unclear if the data set is actually representative of the populations or environmental conditions under study. Comparisons between the sites may have been complicated by inconsistent methods of data collection or classification, as data was collected by many different individuals at different times. This is especially clear with the inconsistencies in the equipment and procedures used to measure heights between and within sites. More thorough sampling is necessary to corroborate the results of this study and apply these insights to similarly impacted populations.

## Conclusion

This study serves as a preliminary description of a cloistered mangrove forest, delineating differences in forest structure and salinity when compared to nearby marginal mangrove forests. Significant differences were found in salinity, mature tree height, mature tree DBH, and *L. racemosa* prevalence as both mature trees and seedlings. These results distinguished the cloistered mangrove forest from the typical marginal mangrove forest in the Bocas del Toro region, validating the question of whether the presence of anthropogenic coastal development is responsible for the differences observed. Though data was not rigorous enough to support strong conclusions, possible explanations emerged from the salinity readings in the cloistered site, indicating a freshwater environment with no input from the ocean. These freshwater conditions are likely caused by the walking path running between the mangrove forest and the ocean. Future research in the cloistered forest could focus on describing the water regime to more rigorously confirm the presence or absence of seawater influx. Since dead trees were ignored in this study, it would also be valuable to assess the density and size (height, DBH) of standing dead trees to gain an understanding of the dynamics of tree death in this forest.

It is important to understand the possible cascading effects of anthropogenic coastal development, as changes in physical conditions can produce population dynamic shifts that lead to mangrove die-off and displacement. Mangroves exist in connection with their biological and physical surroundings, so even developments that do not directly cause deforestation still have the potential to impact physical conditions, with effects on water regime being especially important (Lewis, 2005). This would not only mean a loss of mangrove habitat, but also a loss of the disproportionately beneficial ecosystem services that mangrove forests provide.

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