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Assessing Forest Cover Loss Using Landsat Images and GIS: A Case Study in Colombian Protected Area

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ABSTRACT
The Corchal "El Mono Hernández" Fauna and Flora Sanctuary is a protected area located in northern Colombia and is home to the *Pterocarpus officinalis* forest. In this area, however, this forest cover is declining due to natural causes associated with a change in the hydrosedimentological dynamics. Multitemporal assessment was performed to quantify the forest loss. Landsat images from the years 1986, 1998, 2003, 2013, and 2018 were downloaded and a supervised classification was performed using four cover classes: "*Pterocarpus officinalis*," "other vegetation," "waterbody," and "bare land." The results showed that the vegetation cover of *Pterocarpus officinalis* forest had an initial extent of 865.26 ha in 1986; for the 1986–1998 period, the reduction was 60.30 ha; for the 1998–2003 period, it was 399.15 ha; and for the 2003–2013 period, it was 78.30 ha. Contrary to previous periods, in the 2013–2018 period, 79.65 ha of forest was recovered. In conclusion, *Pterocarpus officinalis* forest lost approximately 50% of its cover during the 1986–2018 period. This information is of great importance for government and management entities for decision-making in the conservation and restoration of this forest.

KEYWORDS
Landscape ecology; natural forest; natural reserves; remote sensing; sustainable; maximum likelihood

Introduction

Protected areas are zones designated for the conservation of species and ecosystems and have been established as a strategy to prevent deforestation and biodiversity loss (Branquart et al., 2008; James et al., 1999; Leberger et al., 2020). Currently, there are more than 200,000 protected areas, covering 16.64% of terrestrial ecosystems (UNEP-WCMC & IUC, 2021), which provide important social and ecosystem services, and constituting a key strategy for the preservation and conservation of global forest cover (Geldmann et al., 2015; Pimm et al., 2014).

However, forests inside protected areas are susceptible to a variety of threats, such as agriculture, logging, fires, and even mismanagement, which cause habitat loss and fragmentation resulting in ecological degradation of the protected forest (Buřivalová et al., 2021;

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Dolman, 2000; Hooper et al., 2012). It has been reported that more than one-third of protected areas worldwide have experience disturbance, losing 3% of their forest cover in protected areas during the 2001–2012 period (Jones et al., 2018; Wade et al., 2020). Although the principal disturbances of protected areas are linked to anthropogenic activities (Dolman, 2000; Hooper et al., 2012), patterns of change in forest vegetation inside protected areas could be associated due to natural causes (e.g., natural disasters and climate change; Boutin et al., 2009; Desrochers et al., 2012; Tews et al., 2004).

In Corchal "El Mono Hernández" Fauna and Flora Sanctuary (FFS), a protected area of Colombia, the forest cover of the species *Pterocarpus officinalis* (Figure 1) is declining due to natural causes. This loss is a concern because it threatens the existence of the homogeneous *Pterocarpus officinalis* forest area in the Colombian Caribbean (Guzmán Peña et al., 2019; Parques Nacionales Naturales de Colombia, 2017b). This forest is part of the coastal swamp forest ecosystem, integrating the Dique Channel ecoregion, a zone influenced by the freshwater runoff from the Dique Channel, a man-made extension of the Magdalena River.

The Corchal "El Mono Hernández" FFS receives massive sediments from the Dique Channel. A large amount of sediments are transported to this protected area (Busnelli & Schuurman, 2020), affecting the freshwater inflow in Corchal "El Mono Hernández" FFS (Guzmán Peña et al., 2019; Parques Nacionales Naturales de Colombia, 2017b). Therefore, the estuarine waters of this swamp ecosystem have been disturbed and the water salinity has increased (Guzmán Peña et al., 2019). The cover forest of *Pterocarpus officinalis* is declining as this tree has a low salinity tolerance (Colón-Rivera et al., 2014; Eusse & Aide, 1999a; Rivera-Ocasio et al., 2007; Saint-Etienne et al., 2006). This cover loss is against the conservation goals of this protected area due to importance of the ecological value of this forest that is the habitat of native fauna (Aguilar-Cano et al., 2016; Cano, 2008; Ministério de Medio Ambiente, 2002; Parques Nacionales, 2017.

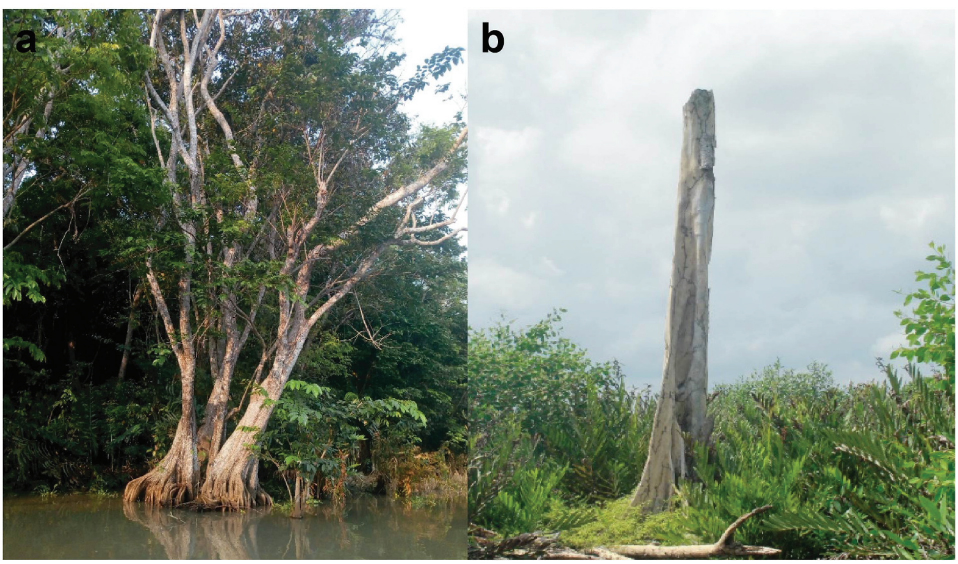

Figure 1. a) *Pterocarpus officinalis* tree in the Corchal "El Mono Hernández" FFS. b) A dead individual of *Pterocarpus officinalis* within the Corchal "El Mono Hernández" FFS protected area.

Moreover, the access and transportation within this area is a challenge, for its remote location and the flooded and swampy terrain, making in situ monitoring activities difficult (Parques Nacionales Naturales de Colombia, 2017b). Faced with this scenario, it is necessary to implement strategies that contribute to the study and monitoring of this protected area. Using Landsat images together with geographic information systems (GIS) represents an important alternative to assess the *Pterocarpus officinalis* forest cover loss (Fisher et al., 2021; Southworth et al., 2018). These images are widely used to assess spatiotemporal changes in protected areas due to their temporal resolution and data availability (Da Ponte et al., 2017; Southworth et al., 2018; Wiens et al., 2009; Willis, 2015).

The spectral data in satellite images allows to calculate vegetation index such as the Normalized Difference Vegetation Index (NDVI), which is used for measuring vegetation greenness and forest health, detecting changes, and analyzing fragmentation. Also, this index has been used in change vegetation studies to assess protected areas disturbances (Islam et al., 2021; Khatancharoen et al., 2021; Shaharum et al., 2018; Sieber et al., 2013). Therefore, this study uses the above tools in the Corchal "El Mono Hernández" FFS to analyze the *Pterocarpus officinalis* forest cover changes and to quantify the area of loss.

Materials and methods

Study area

The Corchal "El Mono Hernández" FFS is a protected area in the Colombian Caribbean; 67% of the territory is distributed in the Department of Sucre, and the remaining 33% is in the Department of Bolívar (Figure 2). This protected area was established in 2002, with the purpose of conserving the genetic resources of flora and fauna of Colombia. The sanctuary's main

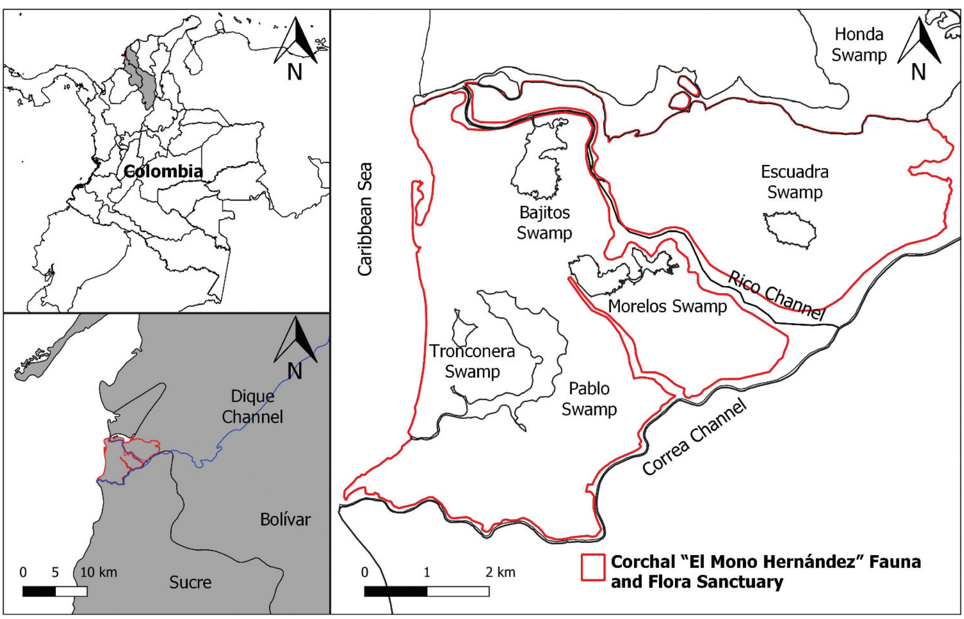

Figure 2. Location of the Corchal "El Mono Hernández" Fauna and Flora Sanctuary in the Colombian Caribbean.

objective is the conservation of the mangrove forests and *Pterocarpus officinalis* of the lower Dique Channel Delta, as well as alluvial and fluviomarine plains, marshes, swamps, and wetlands; and the associated fauna (Cano, 2008; Ministério de Medio Ambiente, 2002). The sanctuary has approximately 3870 hectares and is characterized by a flooded ecosystem, so swamp vegetation is the main flora in Corchal "El Mono Hernández" FFS. The mangrove species that can be found in this area belong to Rhizophora mangle (red mangrove), *Avicennia germinans* (black mangrove), *Laguncularia racemosa* (white mangrove), *Conocarpus erectus* (button mangrove), and *Pelliciera rhizophorae* (tea mangrove), and the species *Acrostichum aureum* (golden leather fern, Matatigre). Additionally, this protected area has the only homogeneous forest of *Pterocarpus officinalis* (corchal tree) in the Colombian Caribbean and is the representative species of the sanctuary, which took its name after the corchal tree (Sánchez Herrera, 2018). On the other hand, the sanctuary has 57 hectares of fluviomarine plains, 312 hectares of swamps, and 28 hectares of fresh and brackish water channels (Banco Franke Ante, 2016; De la República, 2006).

In this area of freshwater swamp forest, the climate is semiarid tropical with an average rainfall of 1100 mm/year, which usually occurs from April to November, with an intermediate dry period occurring in June and July (Gamba Cubides et al., 2006). The sanctuary is located on a deltaic system from which its tributaries derive from, contributing to the formation of channels and water distribution in the area. The landscape is also shaped by channel floodplains formed by sediments deposition from Correa, Rico, Burro, and Portobelo channels. Also, the coastal floodplains influence in the estuarine systems where the mangrove forest is located (Martínez Insignares, 2011). Within the sanctuary, there are five main swamps called Pablo, Tronconera, Bajitos, Morelos, and Escuadra. On the other hand, this zone is characterized by fauna typical of estuarine environments. There are 153 species of birds and different mammals, such as *Procyon cf. cancrivorus* (crab-eating raccoon), *Hydrochoerus hydrochaeris* (capybara), *Alouatta seniculus* (red howler monkey), *saguinus sp*. (tamarin), and *Cuniculus paca* (spotted paca), among others (Aguilar-Cano et al., 2016; Parques Nacionales, 2017).

Field survey

A field reconnaissance of the Corchal "El Mono Hernández" FFS area was carried out in June 2018, to examine the condition of vegetation, including the five (5) species of mangroves and the species *Pterocarpus officinalis*.

During the years 2018 (August 24 and October 14) and 2019 (March 16 and June 4), ground points were obtained from the field and used as references for landscape types and spatial locations. Also, these reference points were collected with the aim to be used as training and validation data. The ground points were recorded with Global Position System (GPS) (Waypoint) Garmin Etrex 20x handheld receiver (www.garmin.com.co/eTrex-20x). Since the sanctuary has accessibility issues, flights with drones were conducted to record vegetation cover. The flights were carried out with drones DJI Phantom 3 pro (www.dji.com/phantom-3-pro) and DJI Phantom 4 (www.dji.com/phantom-4).

Satellite Images

Landsat satellite images available from https://landsat.usgs.gov were downloaded. Landsat 5-TM scenes for 1986 and 1998, Landsat 7-ETM+ for 2003 and Landsat 8-OLI for 2013 and 2018 were downloaded. All Landsat images had a spatial resolution of 30 m (Table 1) and the cloud cover was not to exceed 22% (Sieber et al., 2013). Images from Landsat 7-ETM+ collected after May 31, 2003, were avoided due to these images have data gaps, as the Scan Line Corrector (SLC) failed. These images were then preprocessed in the ENVI 5.3 program (www.l3harrisgeospatial.com) to remove clouds and shadows. In addition, radiometrically correction was performed to convert the digital number (DN) to surface radiance (Knorn et al., 2012).

The atmospheric correction of the Landsat images was performed using FLAASH, which incorporates the MODTRAN6 radiative transfer code to estimate surface radiance values. The FLAASH algorithm requires ancillary data such as atmospheric conditions, date acquisition, scene location, sensor location, and elevation data. These auxiliary data are used as input data to convert surface radiance to surface reflectance (Jensen, 2015; Schowengerdt, 2012).

Classification and change detection assessment

To detect changes in landscape cover, a supervised classification was performed. Maximum likelihood was chosen as the classification algorithm that is typically used in cover vegetation studies (Ghebrezgabher et al., 2016; Ghobadi et al., 2014; Strahler, 1980). This algorithm assigns pixels to the classes that have the highest probability. To establish the training data, bibliographic and cartographic information available for the area was used. Also, the reference points that were taken from field were used as input data for classification algorithm. These reference points were chosen considering the mobility and access conditions during the field survey. Then image pixels were assigned to the following cover classes: waterbody, bare land, *Pterocarpus officinalis*, and other vegetation. Regarding this latter class, other vegetation class includes all vegetation that is not *Pterocarpus officinalis*, such as mangrove forests, flooded grasslands, tall graminoid vegetation, *Typha domingensis*, and *Acrostichum aureum* (PNNC, 2006).

Confusion matrix, also known as the error matrix was created with the reference points, which were obtained from field, Google Earth images, maps of the area and aerial images, which were taken using DJI Phantom 3 pro and DJI Phantom 4 drone (Tiberiu Paul Banu et al., 2016). Next, overall accuracy of the classification was verified, which was calculated by summing the number of pixels correctly classified and dividing by the total number of pixels. Likewise, the Kappa index, κ, the user's accuracy and the producer's accuracy were determined (Da Ponte et al., 2017). Finally, vegetation cover maps were created using ArcMap 10.6 software (https://desktop.arcgis.com). The changes detected were estimated by the difference in existing covers between 1986, 1998, 2003, 2013, and 2018. Also, NDVI maps were generated to assess the vegetation changes operating the near-infrared (NIR) and red bands (Eq. 1). NDVI value ranges from −1 to 1 where negatives values represent non-vegetated land areas and positive values represent a healthy vegetation cover area (Pettorelli et al., 2005; Shilong et al., 2003; Willis, 2015).

$$\mathrm{NDVI} = \frac{NIR - Red}{\mathrm{NIR} + \mathrm{Red}} \tag{1}$$

Table 1. Specifications of the satellite imagery used.

Sensor	Scene ID	Spatial resolution (m)	year	Path/Row	Acquisition date
TM	LT50090531986075XXX04	30	1986	009/053	1986–03-16
TM	LT50090531998076CPE00	30	1998	009/053	1998–03-17
ETM+	LE70090532003066EDC00	30	2003	009/053	2003–03-07
OLI	LC80090532013091LGN02	30	2013	009/053	2013–04-01
OLI	LC80090532018099LGN00	30	2018	009/053	2018–04-09

Table 2. Fragmentation metrics (McGarigal, 2015).

Metric	Description	Equation	
Number of patches	Number of patches (n) of the cover or class i	$NP = n_i$	Eq. 2
Patch density	Number of patches of the class i, divided by the total landscape area (A)	$PD = \frac{n_i}{A} \times 100$	Eq. 3
Mean patch size	Sum of the areas of the patches (a) of class i divided by the number of patches	$MPS = \frac{\sum^{a_i}}{n_i}$	Eq. 4
Largest patch index	Percentage occupied by the largest patch	$LPI = \frac{a_{max}}{A} \times 100$	Eq. 5
Total edge	Total edge length (e) of the class i	$TE = \sum e_i$	Eq. 6
Edge density	Sum of the lengths of edges divided by the total landscape area	$ED = \frac{\sum e_i}{A}$	Eq. 7
Total core area	Sum of core areas (c_i) of each patch	$TCA = \sum c_i$	Eq. 8
Mean core area	Sum of core areas of each patch of class i, divided by the number of patches of class i	$CORE_{MN} = \frac{\sum c_i}{n_i}$	Eq. 9
Mean Euclidean nearest-neighbor distance	Equivalent to the distance of the patch to its nearest neighbor of the same type or class (h), divided by the number of patches.	$ENN_{MN} = \frac{\sum h_i}{n_i}$	Eq. 10
Shannon Evenness	Landscape heterogeneity index. Where P_i is the proportion of class occupied by class i	$SHDI = \sum P_i \times \ln(P_i)$	Eq. 11

Fragmentation and landscape metrics

Fragmentation can be measured through various landscape metrics indices that represent or describe the forest structure. The landscape metrics used in this study were: number of patches, patch density, nearest neighbor and proximity indices, among others (Table 2). This metrics indices were calculated using equations 2–11 and the free software FRAGSTATS 4.2 (Da Ponte et al., 2017). In ArcGIS, the vector layers were converted to GeoTIFF raster, which is a file format recognized by FRAGSTATS 4.2. A 100 m edge depth was set, and 8-neighbor pixel connectivity was used to perform the landscape structure analysis (Couvillion, 2005; De Figueiredo, 2020; Margret, 2009; Mulatu, 2013).

Additionally, the annual rate of forest cover change was determined by Eq. 12:

$$r = \frac{1}{t_2 - t_1} \times \ln\frac{A_2}{A_1} \tag{12}$$

where A_1 and A_2 correspond to the areas of forest cover at time t_1 and t_2 (Puyravaud, 2003), respectively.

Results

Classification and change detection

The cover maps for the years 1986, 1998, 2003, 2013, and 2018 were obtained by supervised classification using the maximum likelihood algorithm (Figure 3). Four classes were defined: waterbodies, bare land, *Pterocarpus officinalis* forest, and other vegetation. The latter class presented the largest area in each of the years evaluated, followed by the class *Pterocarpus officinalis*. The overall accuracy obtained was 98.69%, 99.95%, 99.67%, 99.06%, and 99.78% for 1986, 1998, 2003, 2013, and 2018, respectively. In 1986 and 2013, the Kappa index was 0.98, while in 1998, 2003, and 2018, this value was 0.99. Likewise, the user's and producer's accuracy values were between 0.89 and 1 (Table 3).

1: *Pterocarpus officinalis*, 2: Other vegetation cover, 3: Waterbody, 4: Bare land,w_i: proportion of area, UA: user's accuracy, PA; producer's accuracy.

The changes detection analysis showed a decrease in the extent of the *Pterocarpus officinalis* forest during the period 1986–1998, with a loss of 60.30 ha. Specifically, 244.26 ha was converted to another type of vegetation cover during this period. Likewise, the class other vegetation presented a loss of 265.86 ha, and its main transition was to bare land, which means an increase of 332.37 ha of bare land areas in this period. With respect to waterbodies, there was a small decrease of 6.21 ha.

During the 1998–2003 period, the area covered by *Pterocarpus officinalis* forest was considerably reduced, with a loss of 399.15 ha. This represents a reduction of 49.58%. A total of 423.63 ha of *Pterocarpus officinalis* was converted to other vegetation, which is why this class showed an increase of 545.94 ha, equivalent to an increase of 24.15%. Another considerable change occurred in the transition of 272.52 ha from bare land to other

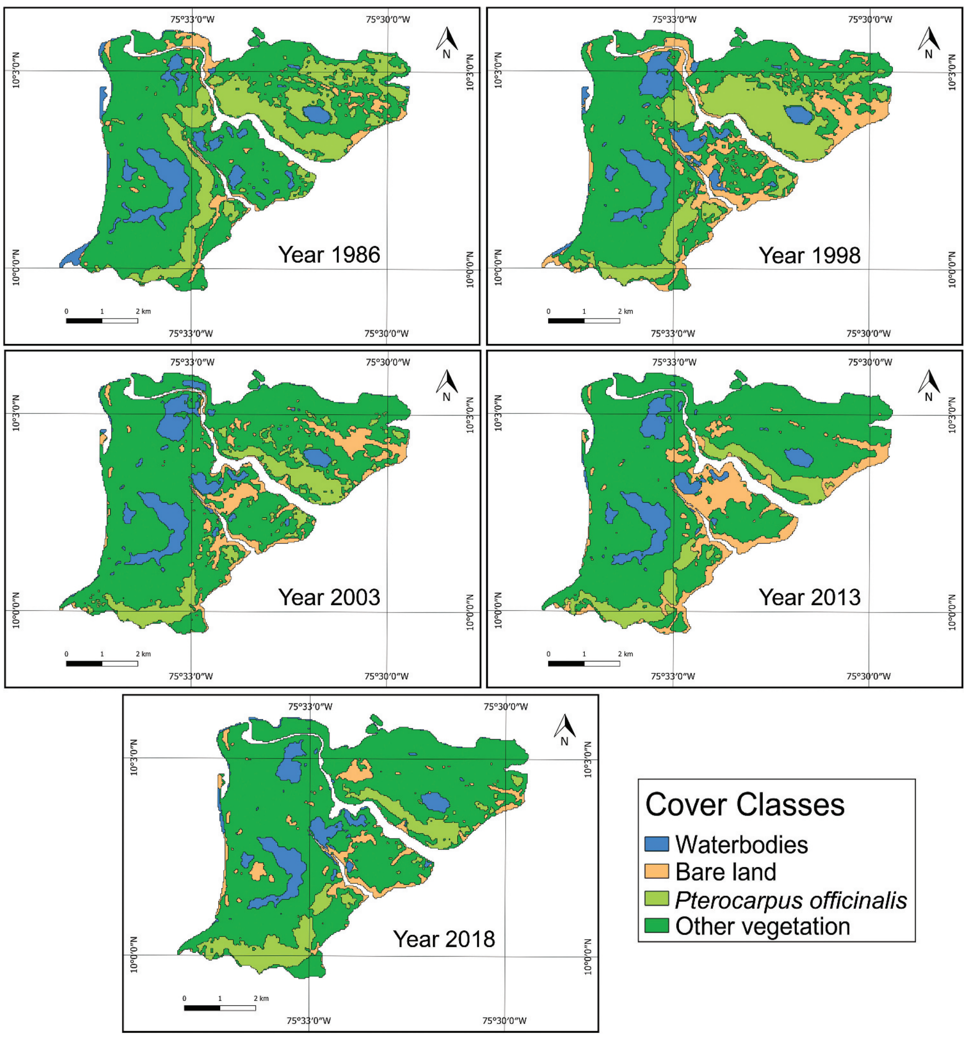

Figure 3. Cover map of the Corchal "El Mono Hernández" FFS for the years 1986, 1998, 2003, 2013 and 2018.

Table 3. User and producer accuracy values obtained for each of the classes evaluated.

		Years														
		1986			1998			2003			2013			2018		
		w_i	UA	PA	w_i	UA	PA	w_i	UA	PA	w_i	UA	PA	w_i	UA	PA
Class	1	0.22	0.99	0.99	0.20	1	1	0.10	1	0.96	0.08	1	0.89	0.10	0.99	0.99
	2	0.64	1	0.99	0.57	0.98	1	0.71	0.99	0.99	0.71	0.97	1	0.71	1	0.99
	3	0.09	1	1	0.09	1	0.99	0.09	1	1	0.07	1	1	0.09	1	1
	4	0.05	0.99	1	0.14	0.99	0.93	0.10	0.99	1	0.13	1	0.95	0.10	0.99	1

vegetation class, in contrast to waterbodies, whose increase was 4.59 ha, being the class presenting the smallest change.

For the period 2003–2013, the main change was evident in the areas classified as bare land, which increased by 35.73%, followed by *Pterocarpus officinalis* forest cover, which continued to decrease to 327.51 ha, signifying a reduction of 19.30%. In contrast to the previous period (1998–2003), the other vegetation class remained relatively stable between 2003 and 2013. However, this period also showed the greatest reduction in the water body class within the protected area. Finally, for 2018, the forest cover of *Pterocarpus officinalis* presented an area of 407.16 ha, which represents an increase of 24.32% compared to the cover existing in 2013. Specifically, 79.65 ha of forest was recovered for 2018. Likewise, the other vegetation class gained 152.64 ha, while bare land shrank by 259.02 ha. Therefore, during this period, there was a growth of the existing vegetation of the sanctuary, both as forested area, as well as shrub vegetation and grassland (Table 4).

Landscape metrics and fragmentation analysis

Of the landscape metrics analyzed (Figure 4 and Figure 5), number of patches (NP) and mean patch size (MPS) indicated the presence of fragmentation. During 2003, there was an increase in NP, while in 2013, the number decreased to 13 and then increased to 23 in 2018 (Figure 5). This same pattern occurred with patch density (PD) because density depends on the number of existing patches in the landscape. PD values were highest from 1986 to 2003 compared to those recorded during 2013 and 2018, which indicates a higher fragmentation rate during the first 17 years compared to the most recent period (Figure 4a). Regarding MPS values, in 1986 it was 14.42 ha, going to 15.48 in 1998, to 6.15 in 2003, to 25.19 in 2013 and to 16.96 in 2018. The higher number of patches in 2003 indicates fragmentation as a result of the loss in cover recorded in that year. Additionally, the increase in NP and the decrease in MPS in this year also indicate fragmentation (Figure 5).

Discussion

One of the main objectives of the Corchal "El Mono Hernández" FFS is conserving the *Pterocarpus officinalis* forest. For this reason, it is of great interest to detect and quantify the changes in cover. During 1986, the initial area was 865.26 ha, of which 6.97% was lost by 1998 at an annual rate of 0.60%; likewise, for 1998–2003, there was a 49.59% loss of *Pterocarpus officinalis* forest at an annual rate of 13.70%. During the 2003–2013 period, 78.3 ha was lost at an annual rate of 2.14%, contrary to what happened in the 2013–2018

Table 4. Extent of area obtained for each of the classes mapped through the supervised classification for each of the years studied.

	Area (ha)			
Year	Other vegetation cover	*Pterocarpus officinalis*	Bare land	Waterbody
1986	2526.48	865.26	202.14	358.83
1998	2260.62	804.96	534.51	352.62
2003	2806.56	405.81	383.04	357.21
2013	2808.99	327.51	520.02	296.1
2018	2961.63	407.16	261	322.83

period, in which 79.65 ha was gained at an annual rate of 4.35%. This indicates that the greatest loss occurred between 1998 and 2003 when forest cover was reduced by approximately half and at a higher rate.

The 2013 results were compared with the cover information of the Dique Channel System Restoration project (Consorcio Dique & Fondo Adaptacion, 2016), finding similarities in the areas of *Pterocarpus officinalis* identified for this year, which were 324.88 ha compared to the 327.51 ha identified in this study. Within this restoration project, loss of *Pterocarpus officinalis* was also identified in the vicinity of the Pablo swamp during the 80′s and 90′s, as well as increasing trend of *Pterocarpus officinalis* cover along Correa channel. The above confirms and validates the classification obtained in this research.

On the other hand, the largest decrease in *Pterocarpus officinalis* cover occurred in the northeastern area of the sanctuary near Rico channel, where forest and vegetation conversion have been more dynamic, which is reflected in the normalized difference vegetation index (NDVI) variations (Figure 6), whose map of change for the period 2013–2018 shows a considerable decrease in the index in an area containing *Pterocarpus officinalis*. This decrease in NDVI is due to mortality, which was recorded photographically in surveys carried out in 2018 (Figure 7). The above shows the validity of the data estimated and obtained in this report through remote sensing products with those observed in the in situ survey.

The *Pterocarpus officinalis* forest located near Rico channel is one of the main patches of the sanctuary. However, due to sedimentation and obstruction of the channels, the freshwater inflow has decreased, which has caused an increase in the salinity of water, and therefore the increase in the mortality rate of adult trees, given that the survival of this species depends on the exchange of fresh and salt water (Colón-Rivera et al., 2014; Lambs et al., 2015). When salinity increases, it affects the production of flowers and leaves (Eusse & Aide, 1999b), as well as the growth of roots and their symbiotic associations, reducing the colonies of mycorrhizal-arbuscular fungi (Colón-Rivera et al., 2014; Fougnies et al., 2007; Saint-Etienne et al., 2006).

The waters that flow through the Correa channel and Rico channel derive from the Dique Channel, which is an artificial channel created in 1650 that has undergone various interventions and rectifications with the aim of improving its navigability. However, along with these interventions was a greater sediment and transport of approximately 10 million tons/day (Mejía Monterroza, 2015). This large contribution of sediment has caused a decrease of freshwater inflow to swamps and the channels associated with the Dique Channel ecoregion (Ordóñez, 2009; Roa et al., 2007). Thus, these hydrological and hydrosedimentological changes have affected the survival of the *Pterocarpus officinalis* forest, which are of great

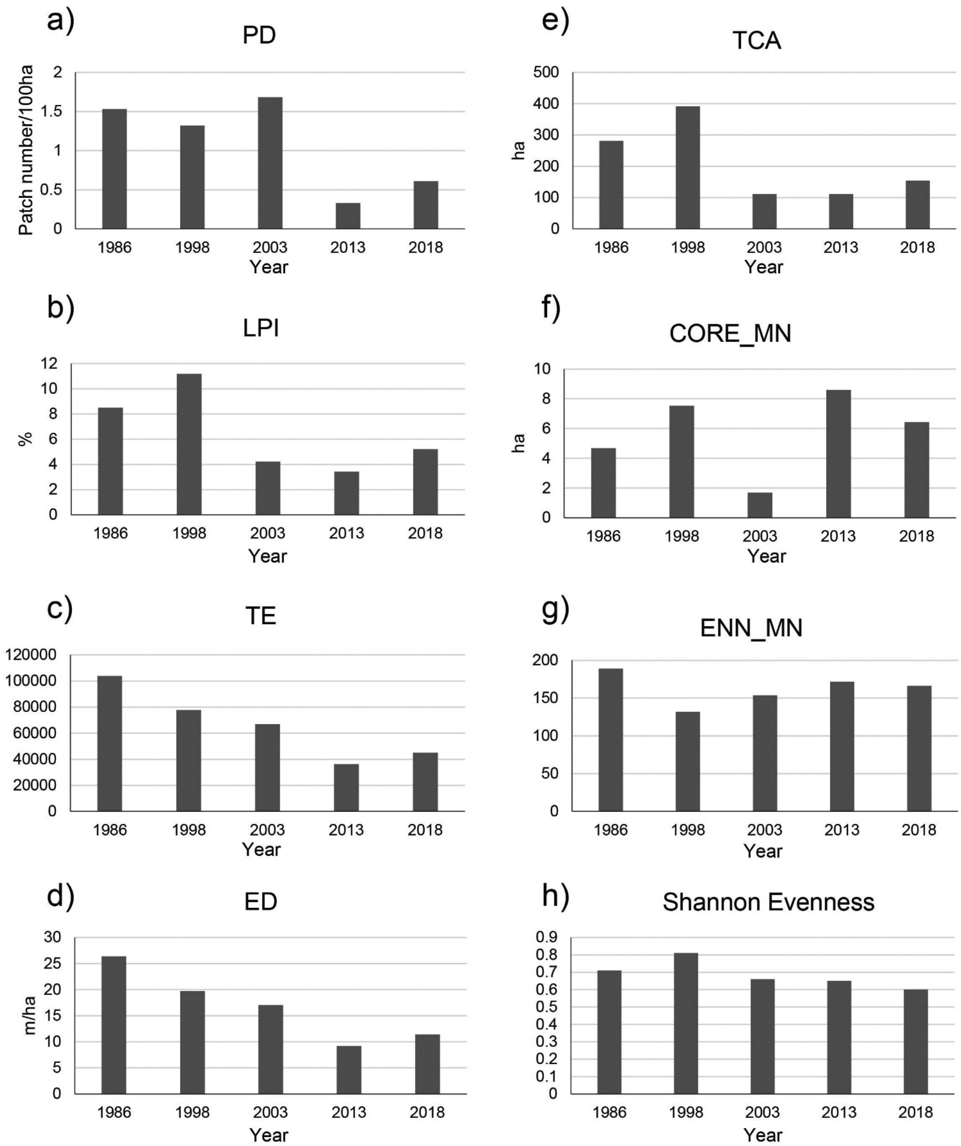

Figure 4. Landscape metrics of the class *Pterocarpus officinalis* a) Patch density (PD), b) Largest patch index (LPI), c) Total edge (TE), d) Edge density (ED), e) Total core area (TCA), f) Mean core area (CORE_MN), mean Euclidean nearest-neighbor distance (ENN_MN) and Shannon Evenness.

interest for environmental and forest science, given that they fix atmospheric nitrogen (Galiana et al., 2019) and help protect marine ecosystems by trapping and filtering harmful sediments (Bacon, 1990). According to Cintron (1983) and Rivera-Ocasio (2007), the species *Pterocarpus officinalis* helps control coastal and riparian erosion, in addition to providing refuge for different animal species. Therefore, this species has a significant value for coastal ecosystems and biodiversity conservation; however, this forest is affected by the pressures described above.

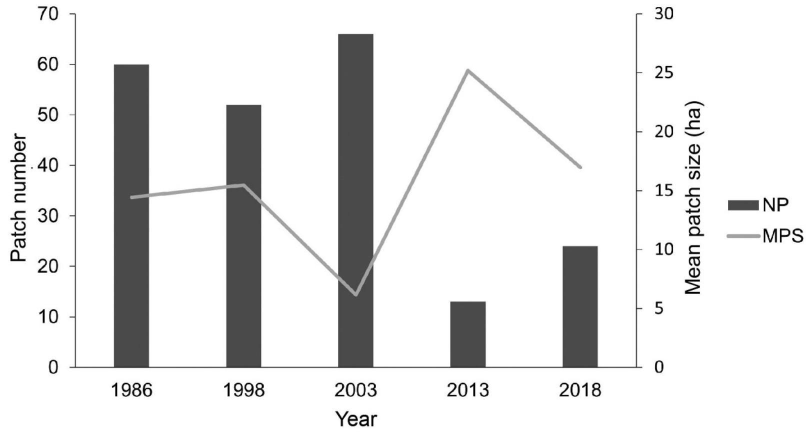

Figure 5. Trend in the number of patches (NP) and mean patch size (MPS) during the period 1986–2018.

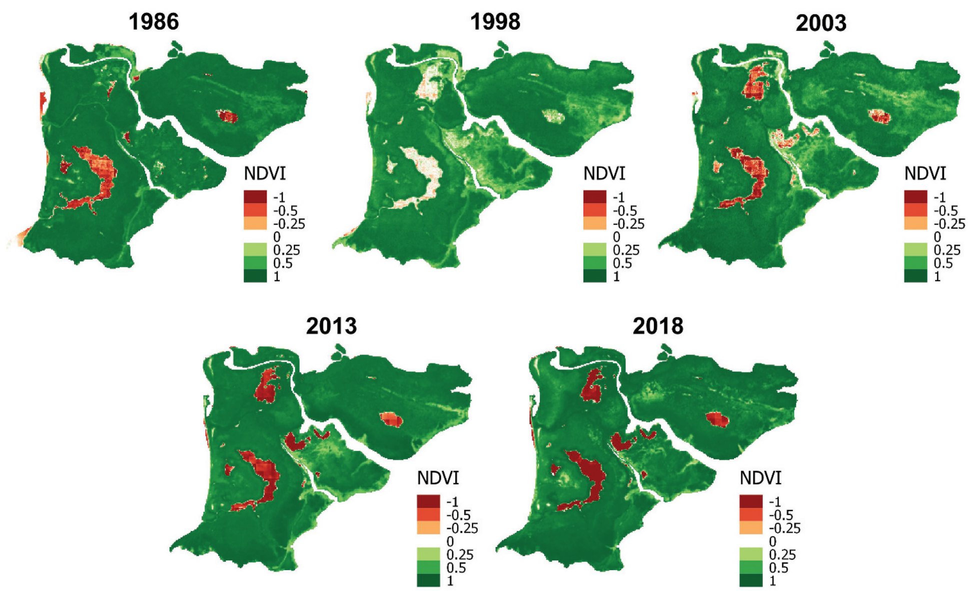

Figure 6. Normalized Difference Vegetation Index (NDVI) map, for the years 1986, 1998, 2003, 2013 and 2018 in the protected area Corchal "El Mono Hernández" FFS. NDVI < 0 represents non-vegetated areas; NDVI = 1 represents healthy vegetation.

The constant loss of *Pterocarpus officinalis* cover over time has been considerable, and most of the area converted to other vegetation cover, which include swampy vegetation capable of resisting higher levels of salinity (Consorcio Dique & Fondo Adaptacion, 2016). However, the identification and quantification of this information warrants a more detailed study to understand the processes of ecological succession presented in the area under study. In addition, the need arises to know if this expanding flora includes species native to the sanctuary or invasive species. However, it is presumed that there is increase in the species *Acrostichum aureum* in the protected area due to its notorious presence in field

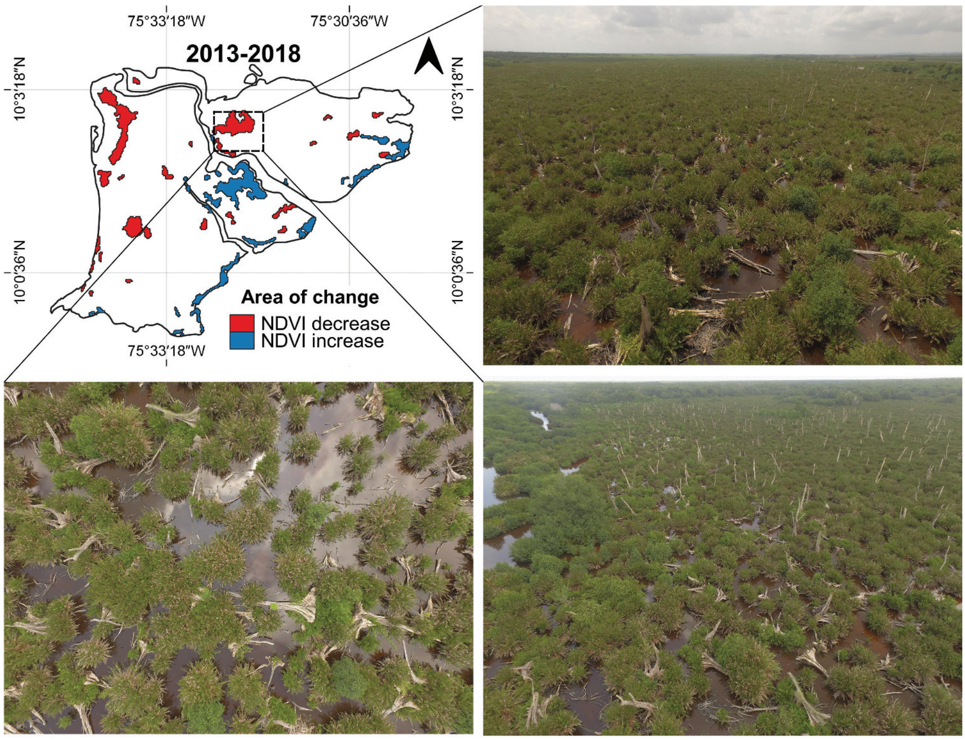

Figure 7. Photographic evidence of *Pterocarpus officinalis* mortality recorded with a drone in the area of lower NDVI.

surveys and its constant presence in aerial imagery, in which this species was captured in conjunction with *Pterocarpus officinalis* and even in areas where mortality has occurred in this forest. This could indicate the colonization of this species; for this reason, a more exhaustive investigation should be proposed to determine the distribution and abundance of this native species within the flora and fauna sanctuary.

On the other hand, during the period 2013–2018, the *Pterocarpus officinalis* cover class gained area, perhaps associated with the conservation actions carried out within the protected area, which have focused on the rehabilitation of the water flows in the main channels (Parques Nacionales Naturales de Colombia & El Fondo de Patrimonio Natural). However, it is necessary to continue monitoring the cover of these forests given their high vulnerability to avoid large losses, such as those that occurred in Puerto Rico, where the *Pterocarpus officinalis* forest was reduced to only 5 ha (Cintrón, 1983) or cases such as the one that occurred in the Colombian protected area, Vía Parque Isla de Salamanca, where 5930 ha of mangrove forests were impacted by the construction of the Barranquilla-Ciénaga highway, which interrupted the connectivity of hydrological flows of the delta-lagoon system (PNNC 2017a).

Therefore, it is necessary to implement effective conservation measures to stop and mitigate their mortality, including restoration projects, monitoring systems and regulation of fresh and saltwater exchange, control of sediment transit and saltwater intrusion, in addition to systems for adapting to climate change and sea level rise. Currently, the Dique macroproject is being developed, which includes the reopening of the flow of Rico channel, an action that could aid in the ecological restoration of the sanctuary and according to the updated Watershed Land Use and Management Plan (POMCA) for Dique Channel, these territories would become marine biotope restoration areas run by the Macroproject (POMCA Canal del Dique, 2018). It is worth noting the importance of this research, since it contributes to the understanding of the landscape changes in this protected area, generating quantitative information of great importance for government and management entities for decision-making in the conservation and restoration of this ecosystem. Also, these findings add new data to the literature on *Pterocarpus officinalis* forest cover since there are a few existing publications related to this forest.

Conclusion

Cover maps for the years 1986, 1998, 2003, 2013, and 2018 were obtained using Landsat images and a supervised classification applying the maximum likelihood algorithm, presenting high values of accuracy. The *Pterocarpus officinalis* cover was identified and classified, which had an initial area of 865.26 ha in 1986; however, by 2018, this cover was reduced by approximately 50%. It should be noted that during the period 2013–2018, the forest recovered 79.65 ha at a rate of 4.35%. This recovery may be due to the restoration and conservation actions implemented in the protected area. Additionally, through landscape and class metrics, it was possible to estimate the presence of fragmentation during the period 1986–2003; however, during 2013 and 2018, this fragmentation was less because the most isolated patches were lost, leaving the largest strips of forest around Rico channel and Correa channel. This decrease in fragmentation may also be associated with the gain in cover during the last period 2013–2018.

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References

Aguilar-Cano, J. R., Barriga, J. C., Córdoba, D., Corzo, G., Diaz-Pulido, A., Forero, F., García Martínez, H., González, M. F., Hernández, O. L., & Medina, S. (2016). Volumen III. *Catálogo de biodiversidad de la región Caribe*.

Bacon, P. R. (1990). Ecology and management of swamp forests in the Guianas and Caribbean region. *Ecosystems of the World*.

Boutin, S., Haughland, D. L., Schieck, J., Herbers, J., & Bayne, E. (2009). A new approach to forest biodiversity monitoring in Canada. *Forest Ecology and Management*, *258*, S168–S175. https://doi.org/10.1016/j.foreco.2009.08.024

Branquart, E., Verheyen, K., & Latham, J. (2008). Selection criteria of protected forest areas in Europe: The theory and the real world. *Biological Conservation*, *141*(11), 2795–2806. https://doi.org/10.1016/j.biocon.2008.08.015

Buřivalová, Z., Hart, S. J., Radeloff, V. C., & Srinivasan, U. (2021). Early warning sign of forest loss in protected areas. *Current Biology*, *31*(20), 4620–4626.e3.

Busnelli, M., & Schuurman, F. (2020). Hydro-morphological management to improve navigation and ecological functions on the 'Canal del Dique'Colombia. In *River Flow 2020* (pp. 2135–2143). CRC Press.

Cano, M. D. (2008). Conflicto de ocupación en áreas protegidas. Conservación versus derechos de comunidades. *Opinión Jurídica*, *7*(14), 53–69. https://revistas.udem.edu.co/index.php/opinion/article/view/765

Cintrón, B. B. (1983). *Coastal freshwater swamp forests: Puerto Rico's most endangered ecosystem* (pp. 249–282). Los Bosques de Puerto Rico.(AE.

Colón-Rivera, R. J., Feagin, R. A., West, J. B., López, N. B., & Benítez-Joubert, R. J. (2014). Hydrological modification, saltwater intrusion, and tree water use of a Pterocarpus officinalis swamp in Puerto Rico. *Estuarine, Coastal and Shelf Science*, *147*, 156–167. https://doi.org/10.1016/j.ecss.2014.06.012

Couvillion, B. R. (2005). *Spatial Heterogeneity in Forested Landscapes : An Examination of Forest Fragmentation and Suburban Sprawl in the Florida Parishes of Louisiana*. Issue August. Louisiana State University.

Da Ponte, E., Roch, M., Leinenkugel, P., Dech, S., & Kuenzer, C. (2017). Paraguay's Atlantic Forest cover loss – Satellite-based change detection and fragmentation analysis between 2003 and 2013. *Applied Geography*, *79*, 37–49. https://doi.org/10.1016/j.apgeog.2016.12.005

de Figueiredo, F. (2020). *Fine-Scale Analyses for Improving Conservation and Sustainability Efforts in Agricultural Landscapes of Neotropical Savannas*. UNIVERSITY OF CALIFORNIA.

de la República, B. (2006). *Subregiones productivas del Caribe colombiano* (J. V. de la Hoz, ed.). Colección). Banco de la República.

Desrochers, A., Tardif, J., & Mazerolle, M. J. (2012). Use of large clear-cuts by wilson's warbler in an eastern Canadian boreal forest. *Avian Conservation and Ecology*, *7*(2) art1. https://doi.org/10.5751/ACE-00521-070201

Dique, C., & Adaptacion, F. (2016). Estudios y diseños definitivos para construcción de las obras del plan de manejo hidrosedimentológico y ambiental del sistema del Canal del Dique.

Dolman, P. (2000). Biodiversity and ethics. In *Environmental science for environmental management* (pp. 119–148). Prentice Hall Harlow.

Eusse, A. M., & Aide, T. M. (1999a). Patterns of litter production across a salinity gradient in a Pterocarpus officinalis tropical wetland. *Plant Ecology*, *145*(2), 307–315. https://doi.org/10.1023/A:1009832627166

Eusse, A. M., & Aide, T. M. (1999b). Patterns of litter production across a salinity gradient in a Pterocarpus officinalis tropical wetland. *Plant Ecology*, *145*(2), 307–315. https://doi.org/10.1023/A:1009832627166

Fisher, A. G., Mills, C. H., Lyons, M., Cornwell, W. K., & Letnic, M. (2021). Remote sensing of trophic cascades: Multi-temporal landsat imagery reveals vegetation change driven by the removal of an apex predator. *Landscape Ecology*, *36*(5), 1341–1358. https://doi.org/10.1007/s10980-021-01206-w

Fougnies, L., Renciot, S., Muller, F., Plenchette, C., Prin, Y., de Faria, S. M., Bouvet, J. M., Sylla, S. N., Dreyfus, B., & Bâ, A. M. (2007). Arbuscular mycorrhizal colonization and nodulation improve flooding tolerance in Pterocarpus officinalis Jacq. seedlings. *Mycorrhiza*, *17*(3), 159–166. https://doi.org/10.1007/s00572-006-0085-2

Franke Ante, R. (2016). *Corchal Sanctuary*. United Nations Environment Program UNEP 812. http://www.car-spaw-rac.org/IMG/pdf/informe_final_-_corchal_sanctuary.pdf

Galiana, A., Geoffroy, A., Sanguin, H., & Bâ, A. M. (2019). Mycorrhizal networks and nitrogen fluxes between *Pterocarpus officinalis* and Taro in swamp forests of Guadeloupe. 4th World Congress on Agroforestry. Montpellier, France. CIRAD. (pp. 812) . https://agritrop.cirad.fr/593059

Gamba Cubides, N. J., Pineda Vargas, J., Bedoya, D. M., & Martinez, L. A. Parques Nacionales Naturales de Colombia - Dirección Territorial Caribe. (2006). Santuario de Flora y Fauna el Corchal "El Mono Hernández." Plan Básico de Manejo 2006 –2010. https://fondodeaguacartagena.org/2017/10/08/plan-de-manejo-santuario-de-fauna-y-flora-el-corchal-el-mono-hernandez-plan-basico-de-manejo-2006-2010/

Geldmann, J., Coad, L., Barnes, M., Craigie, I. D., Hockings, M., Knights, K., Leverington, F., Cuadros, I. C., Zamora, C., Woodley, S., & Burgess, N. D. (2015). Changes in protected area management effectiveness over time: A global analysis. *Biological Conservation*, *191*, 692–699. https://doi.org/10.1016/j.biocon.2015.08.029

Ghebrezgabher, M. G., Yang, T., Yang, X., Wang, X., & Khan, M. (2016). Extracting and analyzing forest and woodland cover change in Eritrea based on landsat data using supervised classification. *The Egyptian Journal of Remote Sensing and Space Science*, *19*(1), 37–47. https://doi.org/10.1016/j.ejrs.2015.09.002

Ghobadi, Y., Pradhan, B., Shafri, H. Z. M., Bin Ahmad, N., & Kabiri, K. (2014). Spatio-temporal remotely sensed data for analysis of the shrinkage and shifting in the Al Hawizeh wetland. *Environmental Monitoring and Assessment*, *187*(1), 4156. https://doi.org/10.1007/s10661-014-4156-0

Guzmán Peña, K., Correa Uribe, L. M., & García Calderón, L. M. (2019). Gobernanza y gestión de áreas protegidas. In G. L. Worboys, M. Lockwood, A. Kothari, S. Feary, & I. Pulsford (Eds.), *situ Parques Nacionales Naturales de Colombia*. ANU Press. https://doi.org/10.22459/GGAP.2019

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E. K., Hungate, B. A., Matulich, K. L., Gonzalez, A., Duffy, J. E., Gamfeldt, L., & Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature*, *486*(7401), 105–108.

Islam, M. A., Jimmy, A. N., Alam, M. S., & Khan, N. A. (2021). The use of multi-temporal Landsat normalized difference vegetation index (NDVI) data for assessing forest cover change of Lawarchara National Park. *Environment, Development and Sustainability*, *23*(12), 17702–17722. https://doi.org/10.1007/s10668-021-01408-x

James, A. N., Green, M. J. B. B., & Paine, J. R. (1999). A global review of protected area budgets and staff. *WCMC Biodiversity Series*, *10*, 39. http://www.unep-wcmc.org/resources/PDFs/10_all.pdf

Jensen, J. R. (2015). *Introductory digital image processing: A remote sensing perspective*. Prentice Hall Press.

Jones, K. R., Venter, O., Fuller, R. A., Allan, J. R., Maxwell, S. L., Negret, P. J., & Watson, J. E. M. (2018). One-third of global protected land is under intense human pressure. *Science*, *360*(6390), 788–791. https://doi.org/10.1126/science.aap9565

Khatancharoen, C., Tsuyuki, S., Bryanin, S. V., Sugiura, K., Seino, T., Lisovsky, V. V., Borisova, I. G., & Wada, N. (2021). Long-Time Interval Satellite Image Analysis on Forest-Cover Changes and Disturbances around Protected Area, Zeya State Nature Reserve, in the Russian Far East. *Remote Sensing*, *13*(7), 1285. https://doi.org/10.3390/rs13071285

Knorn, J., Kuemmerle, T., Radeloff, V. C., Szabo, A., Mindrescu, M., Keeton, W. S., Abrudan, I., Griffiths, P., Gancz, V., & Hostert, P. (2012). Forest restitution and protected area effectiveness in post-socialist Romania. *Biological Conservation*, *146*(1), 204–212. https://doi.org/10.1016/j.biocon.2011.12.020

Leberger, R., Rosa, I. M. D., Guerra, C. A., Wolf, F., & Pereira, H. M. (2020). Global patterns of forest loss across IUCN categories of protected areas. *Biological Conservation*, *241*, 108299. https://doi.org/10.1016/j.biocon.2019.108299

Margret, C. (2009). *Defending Public Interests in private forests: Land-Use policy alternatives for the Xingu river headwaters region of southeastern Amazonia*. [University of Florida]. https://ufdc.ufl.edu/UFE0041028/00001

Martínez Insignares, A. J. (2011). *Potencial turístico de Arjona, Bolívar* [Universidad Tecnológica de Bolívar]. http://repositorio.utb.edu.co/handle/20.500.12585/3399

McGarigal, K. (2015). FRAGSTATS help. In *University of Massachusetts* (pp. 182). USA.

Mejía Monterroza, G. E. (2015). *Hidrocarburos aromáticos policíclicos en la Costa Caribe colombiana y posibles fuentes de contaminación* (Vol. 151).

Ministério de Medio Ambiente. (2002). *Resolución 0763 de 2002. Por la cual se reserva, alindera y declara el Santuario de Fauna y Flora El Corchal "El Mono Hernández"* (Resolución 0763). http://www.parquesnacionales.gov.co/portal/wp-content/uploads/2013/08/res_0763-de-2002.pdf

Mulatu, K. (2013). *Remote sensing opportunities for biodiversity monitoring in REDD+ MRV. In GRS-2013-10*. Wageningen University. Issue April

Nacionales, P. (2017). *Santuario de Flora y Fauna El Corchal ¨El Mono Hernández¨*. www.parquesnacionales.gov.co

Ordóñez, J. I. (2009). Diseño final de las Obras de Control Sedimentologicos del Canal del Dique. 1–19.

Parques Nacionales Naturales de Colombia. (2006). *Bases de licitación No 039 - Para la contratación de servicios de consultoría para la elaboración de una propuesta de acondicionamiento de caños en el SFF El Corchal "El Mono Hernández" Tendiente a la restauración bosques de corcho y mangle*.

Parques Nacionales Naturales de Colombia. (2017a). *Plan de Manejo Vía Parque Isla de Salamanca 2017-2022*.

Parques Nacionales Naturales de Colombia. (2017b). Portafolio de proyecto de investigacion Santuario de Fauna y Flora El Corchal "El Mono Hernández"

Parques Nacionales Naturales de Colombia, & El Fondo de Patrimonio Natural. (nd). *Bases de licitación No 039 - Para la contratación de servicios de consultoría para la elaboración de una propuesta de acondicionamiento de caños en el SFF El Corchal "El Mono Hernández" Tendiente a la restauración bosques de corcho y mangle*.

Paul Banu, T., Florian Borlea, G., & Banu, C. (2016). The Use of Drones in Forestry. *Journal of Environmental Science and Engineering B*, *5*(11). https://doi.org/10.17265/2162-5263/2016.11.007

Pettorelli, N., Vik, J. O., Mysterud, A., Gaillard, J.-M., Tucker, C. J., & Stenseth, N. C. (2005). Using the satellite-derived NDVI to assess ecological responses to environmental change. *Trends in Ecology & Evolution*, *20*(9), 503–510. https://doi.org/10.1016/j.tree.2005.05.011

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., Raven, P. H., Roberts, C. M., & Sexton, J. O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection. *Science*, *344*(6187), 1246752. https://doi.org/10.1126/science.1246752

POMCA Canal del Dique. (2018). *Ajuste y/o actualización del POMCA Canal del Dique*. http://www.crautonoma.gov.co/documentos/pomcas/canaldedique/4faseformulacion/Fase_Formulacion_Canal_Del_Dique_2903_vf.pdf

Puyravaud, J.-P. (2003). Standardizing the calculation of the annual rate of deforestation. *Forest Ecology and Management*, *177*(1), 593–596.

Rivera-Ocasio, E., Aide, T. M., & Rios-López, N. (2007). The effects of salinity on the dynamics of a Pterocarpus officinalis forest stand in Puerto Rico. *Journal of Tropical Ecology*, *23*(5), 559–568. https://doi.org/10.1017/S0266467407004361

Roa, Z., Duarte, J. A., Cortés, D., Roa Díaz, Z. D. P., Duarte, J. A., & Cortés, D. (2007). *Multifuncionabilidad escasez y riqueza territorial. Revista Finanzas y Política Económica, 1*(1 SE-Artículo de Reflexión), 116–128. https://revfinypolecon.ucatolica.edu.co/article/view/569

Saint-Etienne, L., Paul, S., Imbert, D., Dulormne, M., Muller, F., Toribio, A., Plenchette, C., & Bâ, A. M. (2006). Arbuscular mycorrhizal soil infectivity in a stand of the wetland tree Pterocarpus officinalis along a salinity gradient. *Forest Ecology and Management*, *232*(1–3), 86–89. https://doi.org/10.1016/j.foreco.2006.05.046

Sánchez Herrera, G. (2018). *Plan de Manejo Santuario de Flora y Fauna El Corchal "El Mono Hernández" 2018–2023*. Parques Nacionales Naturales de Colombia. https://www.parquesnacionales.gov.co/portal/wp-content/uploads/2020/10/plan-de-manejo-sff-el-corchal-el-mono-hernandez.pdf

Schowengerdt, R. A. (2012). Remote sensing: Models and methods for image processing: Second edition. *Remote Sensing: Models and Methods for Image Processing: Second Edition*. https://doi.org/10.1016/C2009-0-21902-7

Shaharum, N. S. N., Shafri, H. Z. M., Gambo, J., & Abidin, F. A. Z. (2018). Mapping of Krau Wildlife Reserve (KWR) protected area using Landsat 8 and supervised classification algorithms. *Remote Sensing Applications: Society and Environment*, *10*, 24–35. https://doi.org/10.1016/j.rsase.2018.01.002

Shilong, P., Jingyun, F., Liming, Z., Qinghua, G., Mark, H., Wei, J., Yan, L., & Shu, T. (2003). Interannual variations of monthly and seasonal normalized difference vegetation index (NDVI) in China from 1982 to 1999. *Journal of Geophysical Research: Atmospheres*, *108*(D14). https://doi.org/10.1029/2002JD002848

Sieber, A., Kuemmerle, T., Prishchepov, A. V., Wendland, K. J., Baumann, M., Radeloff, V. C., Baskin, L. M., & Hostert, P. (2013). Landsat-based mapping of post-Soviet land-use change to assess the effectiveness of the Oksky and Mordovsky protected areas in European Russia. *Remote Sensing of Environment*, *133*, 38–51. https://doi.org/10.1016/j.rse.2013.01.021

Southworth, J., Ryan, S. J., Bunting, E., Herrero, H. V., Nagendra, H., Gibbes, C., & Agarwal, S. (2018). Protected Areas, Climate Change, and Ecosystem Sustainability. In *Comprehensive Remote Sensing* (pp. 202–219). Elsevier. https://doi.org/10.1016/B978-0-12-409548-9.10432-4

Strahler, A. H. (1980). The use of prior probabilities in maximum likelihood classification of remotely sensed data. *Remote Sensing of Environment*, *10*(2), 135–163. https://doi.org/10.1016/0034-4257(80)90011-5

Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M. C., Schwager, M., & Jeltsch, F. (2004). Animal species diversity driven by habitat heterogeneity/diversity: The importance of keystone structures. *Journal of Biogeography*, *31*(1), 79–92. https://doi.org/10.1046/j.0305-0270.2003.00994.x

UNEP-WCMC, & IUC. (2021). *Protected Planet Report 2020*. UNEP-WCMC and IUCN:.

Wade, C. M., Austin, K. G., Cajka, J., Lapidus, D., Everett, K. H., Galperin, D., Maynard, R., & Sobel, A. (2020). What is threatening forests in protected areas? A global assessment of deforestation in protected areas, 2001-2018. *Forests*, *11*(5), 539. https://doi.org/10.3390/F11050539

Wiens, J., Sutter, R., Anderson, M., Blanchard, J., Barnett, A., Aguilar-Amuchastegui, N., Avery, C., & Laine, S. (2009). Selecting and conserving lands for biodiversity: The role of remote sensing. *Remote Sensing of Environment*, *113*(7), 1370–1381. https://doi.org/10.1016/j.rse.2008.06.020

Willis, K. S. (2015). Remote sensing change detection for ecological monitoring in United States protected areas. *Biological Conservation*, *182*, 233–242. https://doi.org/10.1016/j.biocon.2014.12.006