

Mangrove resilience under tourism-driven land use change in a Caribbean SIDS: evidence from Samaná Bay, Dominican Republic

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Abstract

Purpose

This study examines Land Use and Land Cover (LULC) dynamics in the Samaná Bay socio-ecological system and asks what recent landscape changes reveal about mangrove resilience and vulnerability in a tourism coastal economy.

Methods

Remote sensing products, GIS analysis, and CA–Markov modeling were integrated to assess LULC transitions from 2000 to 2020, quantify mangrove extent change from 1996 to 2020, and construct exploratory mangrove scenarios to 2030.

Results

The wider landscape experienced substantial redistribution of cover classes, with built-up and cropland areas increasing by 45.37% and 39.12%, respectively, while tree cover declined by 9,220 ha. Mangroves showed a long-term net gain of 0.53% from 1996 to 2020, driven by expansion during 1996–2010 (+ 1.19%). However, 2010–2020 saw localized losses totaling 42.67 ha, attributed to agriculture, urban development, tourism infrastructure, erosion, and flooding. By 2030, scenarios range from continued decline to 6,443.08 ha under a loss trajectory to recovery up to 6,583.56 ha under a strong-gain trajectory.

Conclusion

Findings indicate conditional resilience because Samaná Bay mangroves can recover under favorable sedimentary and governance conditions, but continued tourism-driven development may push the system towards ecological thresholds. Integrated ridge-to-reef governance, coastal zoning, hydrology first restoration, and economic incentives are needed to reconcile local development with long-term coastal conservation.

1. Introduction

Coastal and marine ecosystems are central to livelihoods and welfare in Small Island Developing States (SIDS), where fisheries, tourism, coastal protection, and natural resource use are linked to local economies (Stephenson and Jones 2017), Zamboni, Noleto Filho et al. (2021). Among these ecosystems, mangroves contribute essential services, including flood protection, carbon sequestration, biodiversity conservation, water filtration, and fishery support (Avishek, Yu et al. 2012, Failler, Pètre et al. 2015, Moity, Delgado et al. 2019, Blanco-Libreros and Ramírez-Ruiz 2021, Zamboni, Noleto Filho et al. 2021, Hakim, Lubis et al. 2022, Kutir, Agblorti et al. 2022, Chopade, Mahajan et al. 2023). Between 2002 and 2017, regional declines in global mangrove areas were observed (-22.06%), predominantly due to conversion for agriculture and urban development, although some local gains (+ 4.47%) occurred (Bunting, Rosenqvist et al. 2018, Idris, Mustapha et al. 2021). These reductions have been exacerbated by escalating socioeconomic pressures and insufficient conservation policies, especially at the interface between land and sea (Soanes, Pike et al. 2021, Potapov, Hansen et al. 2022).

Despite these benefits, mangroves are highly exposed to land-use conversion because they occupy the interface between terrestrial and marine systems. Global assessment shows heterogeneous trends, including marked regional declines and localized gains with losses commonly linked to agriculture, urban development, infrastructure expansion, and weak conservation enforcement (Bunting, Rosenqvist et al. 2018, Potapov, Hansen et al. 2022). In the Dominican Republic, mangroves cover 19,184 ha (Bunting, Rosenqvist et al. 2022) and provide significant socio-economic benefits. Annual fisheries yield US\$24.88 million, with avoided damages valued at US\$8.51 million, bringing total ecosystem services to US\$69.92–95.27 million (Kauffman, Heider et al. 2014, Chopade, Mahajan et al. 2023, Ruiz de Gauna, Greño et al. 2024).

This study analyzes the Bahía de Samaná socio-ecological system (BSE) through three complementary lenses: (1) Ridge-to-reef dynamics recognizes that land-based activities can alter sedimentation, hydrological connectivity, and nutrient flows, with direct consequences for coastal ecosystems (Carlson, Foo et al. 2019, Kumari, Singh et al. 2020); (2) Ecosystem services trade-offs

between short-term economic gains from real-estate and tourism development and long-term regulating services such as coastal protection, carbon storage, and fisheries support (Lau 2013, Failler, Pètre et al. 2015, Martino, Tett et al. 2019); and (3) Socio-ecological perspective focuses on thresholds, adaptive capacity, and governance constraints that may turn recoverable disturbance into long-term degradation (Blanco-Libreros and Ramírez-Ruiz 2021, Soanes, Pike et al. 2021). The main research question is: what do LULC dynamics associated with mangrove ecosystems in the BSE during 2000–2020 reveal about spatial patterns, drivers, resilience, and vulnerability, and what scenarios can be projected for 2030 under different conservation and development trajectories?

1.1 The Samana Bay Context

The Samaná Bay socio-ecological system (BSE) covers 5,174.92 km² terrestrial and 1,394.11 km² marine area across four provinces and twelve municipalities (Fig. 1). Its coastal municipalities include 299,163 inhabitants across 3,167.84 km² (ONE 2020, IGN 2024). The BSE is strongly connected to the Yuna River, the country's highest-flow watercourse, which drains sediment-loaded waters from La Humeadora National Park through 197.2 km into BSE mangroves, creating critical ridge-to-reef connectivity linkage (Laba, Smith et al. 1997, Bautista de los Santos 2014).

Since the designation of Samaná as a tourist area in 1994 (Decree No. 91–94), hotel and real estate development have expanded along the coast, often with limited conservation measures (Francisco and Jaime 2014). The region holds 6,485.47 ha of mangroves, representing 25% of the country's total, and has undergone major land use changes due to tourism, agriculture, and extreme weather (Delanoy, Díaz-Asencio et al. 2020, Bunting, Rosenqvist et al. 2022). Sedimentation rates increased by 1.78m per year between 2003–2019, and tropical storms caused 2.17 km² of ecosystem loss (Delanoy, Díaz-Asencio et al. 2019). Despite economic growth, household poverty remains at 17–25% (ONE 2023). This makes Samaná Bay a suitable case for examining how conservation and economic development interact in Caribbean SIDS, where evidence remains more limited than for larger estuarine systems in Asia and Latin America (Nguyen 2014, Soanes, Pike et al. 2021).

2. Materials and methods

2.1. The BSE study area: a critical conservation space

The BSE encompasses diverse terrestrial, coastal, and marine ecosystems with 14 notable protected areas covering around 34,689.82 km² (IGN 2024). The Marine Area of Strict Protection, "Sanctuary of the Banks of La Plata and La Navidad," comprises 96% of this region's protected area at 33,403.29 km², underscoring a strong commitment to marine conservation (Fig. 1). This sanctuary is crucial for humpback whales (*Megaptera novangliae*), which migrate annually from the North Atlantic to the Bay of Samaná for mating and reproduction during the boreal winter, supporting approximately 1,000 individuals (Barlow 1997, Leslie F. New; Alisa J. Hall; Robert Harcourt 2015).

2.2. Data sources

The GLAD satellite data (Potapov, Hansen et al. 2022) enabled the analysis of changes in land use and cover, such as forest height, cropland, urban areas, and wetlands, at a 30 m resolution in the BSE (Table 1).

Mangrove coverage was assessed with the World Mangrove Atlas, which provides global ecosystem distribution at under 100 m per pixel. Data from 1996, 2010, and 2020 were used to compare mangrove coverage and estimate net changes. Intertemporal land use and cover analysis assessed 11 categories (Table 2). Area changes in mangrove and forest ecosystems were analyzed separately in hectares (Appendix A, Fig. 6).

Table 1
Details of the geospatial datasets used.

Satellite Product	Source	Year	Period (years)	Resolution (m)
Global Land Analysis and Discovery (GLAD)	GLAD (Potapov, Hansen et al. 2022)	2000 2020	20	30
World Atlas of Mangroves	Ocean Data Viewer (Spalding M, Kainuma M et al. 2010)	1996 2010 2020	24	30
Global Forest Watch	GFW (Potapov, Hansen et al. 2022)	2000 2020	20	30
Aster Elevation Model	ASTGM V003 (NASA, METI et al. 2019)	2019	-	Raster (.tiff)
GEBCO V3	GEBCO (GEBCO 2024)	2024	-	Raster (.tiff)
Roads	Open Street Map OSM	2024	-	Vector (.shp)
Provinces	National Geographic Institute (IGN 2024)	2020		Vector (.shp)
Municipalities	National Geographic Institute (IGN 2024)	2020		Vector (.shp)
Protected Areas (Marine and Terrestrial)	Earth Map (Morales, Díaz et al.)	2022	-	Vector (.shp)
Source: own elaboration				

2.3. Data processing

Data processing followed the workflow summarized in Fig. 2. ArcGIS Pro 3.1.0, QGIS 3.28.3, and MS Excel were used for spatial processing, area estimation, transition matrices, and scenario construction. All spatial layers were projected to WGS 84 / UTM Zone 19N (EPSG:32619), clipped to the BSE boundary, resampled to a common 30 m spatial resolution when required, and reclassified into eleven LULC categories before intertemporal comparison. TerrSet's Land Change Modeler and CA-Markov modules were used to model LULC transitions from 2000 to 2020 and to project potential patterns to 2030.

CA-Markov modeling combines a transition-probability matrix with a spatial allocation rule. The Markov component estimates the probability that each land cover class i at time t will transition to class j at time $t + 1$ (Selmy, Kucher et al. 2023, Aghababaei, Ebrahimi et al. 2024, Farhan, Wu et al. 2024, Madhusudhan, Shivapur et al. 2024, Slamet, Nababan et al. 2024). Model performance was evaluated by comparing the simulated 2020 map against the observed 2020 map using K-standard, Kno, and Klocation statistics. The transition probability matrix is expressed as:

$$\begin{bmatrix} P_{11} & P_{12} & P_{1.} & P_{1n} \\ P_{21} & P_{22} & P_{2.} & P_{2n} \\ \dots & \dots & \vdots & \dots \\ P_{n1} & P_{n2} & \dots & P_{nn} \end{bmatrix} \quad 1$$

where p_{ij} is the probability of transition from class i to class j , and each row sums to one:

$$\sum_{j=1}^n p_{ij} = 1 \quad 2$$

The expected land-cover distribution at a future time can be estimated as:

$$A_{t+1} = A_t P \quad 3$$

where A_t is the vector of land cover areas at time t , P is the transition probability matrix, and A_{t+1} is the expected land cover distribution at the next time step. The cellular automata component then allocates these transitions spatially by considering the current state of each cell, the transition probabilities, and the configuration of neighboring cells:

$$S_{t+1}(x) = f(S_t(x), N_t(x), P) \quad 4$$

where $S_t(x)$ is the land-cover state of cell x at time t , $N_t(x)$ represents the neighborhood configuration around cell x , P is the Markov transition matrix, and f is the spatial allocation function used to determine the future state of each cell. In this study, the CA–Markov procedure was implemented in TerrSet using the 2000 and 2020 LULC maps to estimate transition probabilities and generate the 2030 projection.

The other equation used is that of the cellular automata (CA) model, where $S_{(t,t+1)}$ represents the current and future state of the system, and N refers to the state of the neighboring cells. The function indicates that the future state of each cell depends on both the current state of the pixel being analyzed and the states of the cells near that pixel f .

2.4. Analysis of mangroves

Mangrove extent was assessed for 1996, 2010, and 2020 using global mangrove datasets. These products are suitable for intertemporal extent analysis but do not provide information on species composition, canopy height, forest structure, or ecological condition. Therefore, results are interpreted as changes in spatial extent rather than direct measures of ecosystem health. Future analysis should integrate NDVI or other biophysical indices, LiDAR-derived structure, socio-economic drivers, and field validation (Fig. 3).

2.4.1. Construction of restoration scenarios to 2030

Mangrove scenarios to 2030 were constructed from observed annual rates of change for 1996–2020, 1996–2010, and 2010–2020. Three exploratory scenarios were estimated: (1) Average Gain, assuming continuation of the long-term net recovery rate, (2) Strong Gain, assuming a recovery trajectory similar to 1996–2010, and (3) Continuous Loss, assuming persistence of the 2010–2020 degradation rate. These scenarios should be interpreted as plausible trajectories for conservation planning rather than deterministic forecasts (see Appendix A).

3. Results

3.1. Changes in land use and land cover (2000–2020)

Between 2000 and 2020, the BSE landscape experienced substantial redistribution among LULC classes across 416,185 ha (Table 2). Tree Cover and Dense Short Vegetation remained the dominant classes, representing approximately 55%-57% and 25–26% of the area, respectively. However, Tree Cover declined by 9,220.20 ha (-3.86%), while Dense Short Vegetation increased by 4,114.40 ha (+ 3.88%), suggesting vegetation turnover and degradation or regeneration dynamics depending on location.

Table 2
Inter-temporal analysis of land use and cover in the BSE (2000–2020)

Cover class	Value	Description	Area (ha) 2000	% 2000	Area (ha) 2020	% 2020	Net change (ha) 2020 – 2000	Variation (%)
Semi-arid	1	51% vegetation cover	15.70	0.004%	18.84	0.005%	3.14	20.00%
Dense short vegetation	2	91% vegetation cover	106,155.60	25.507%	110,270.00	26.495%	4,114.40	3.88%
Tree cover	3	15m trees	238,901.90	57.403%	229,681.70	55.187%	(9,220.20)	-3.86%
Salt pan	4	3% vegetation cover	10.05	0.0024%	18.84	0.005%	8.79	87.50%
Wetland sparse vegetation	5	27% vegetation cover	10.05	0.0024%	25.12	0.006%	15.07	150.00%
Wetland dense short vegetation	6	91% vegetation cover	32,035.52	7.6974%	27,190.88	6.533%	(4,844.64)	-15.12%
Wetland tree cover	7	15m trees	13,320.08	3.2005%	13,311.29	3.198%	(8.79)	-0.07%
Open surface water	8	60–69% of year	2,132.52	0.5124%	2,302.69	0.553%	170.17	7.98%
Cropland	9	No info	11,442.51	2.7494%	15,918.54	3.825%	4,476.03	39.12%
Built-up	10	No info	11,652.25	2.7998%	16,938.96	4.070%	5,286.71	45.37%
Ocean	11	No info	508.64	0.1222%	508.64	0.122%	-	0.00%
Total			416,184.81	100.0000%	416,185.50	100.000%	0.69	0.000165%
Source: Own elaboration								

Wetland Dense Short Vegetation declined by 4,844.64 ha (-15.12%), while Cropland and Built-up areas expanded by 4,476.03 ha (+ 39.12%) and 5,286.71 ha (+ 45.37%), respectively. These shifts indicate increasing pressure from agriculture, settlement expansion, and tourism-related infrastructure. Although the total mapped area remained constant, the internal redistribution of classes suggests growing fragmentation and potential loss of ecological connectivity and ecosystem services.

3.2. Prediction to 2030

The Markov Module estimated a transition probability matrix from 2000–2020 spatial data, which was then used in the Land Change Modeler to project transitions to 2030. Crosstab analysis produced the matrix (Table 3), while 11 conditional probability maps (one per class) show pixel transitions. The Markov Model assumes each state change depends only on the previous state (Takada, Miyamoto et al. 2010).

Table 3
Transition Probability Matrix and Markov Module Details

Class	1	2	3	4	5	6	7	8	9	10	11	Total
value												
1	0.5344	0.0567	0	0	0	0	0	0.2937	0.0298	0.0853	0	1.000
2	0.0001	0.8293	0.0489	0	0	0	0	0.0017	0.0575	0.0624	0	1.000
3	0.0001	0.1259	0.8306	0	0	0	0	0.0004	0.0026	0.0405	0	1.000
4	0	0	0	0.3573	0.0977	0	0.0388	0.5062	0	0	0	1.000
5	0	0	0	0.0511	0.4335	0.0833	0	0.432	0	0	0	1.000
6	0	0	0	0.0006	0.0007	0.7585	0.0172	0.0105	0.1994	0.0131	0	1.000
7	0	0	0	0	0	0.1155	0.8306	0.0028	0.0184	0.0326	0	1.000
8	0.0017	0.0138	0.0014	0.0017	0.0052	0.1145	0.0595	0.7973	0	0.0048	0	1.000
9	0	0.1161	0.0081	0.0001	0	0.1048	0.003	0.0016	0.743	0.0233	0	1.000
10	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	1.000
11	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	0.0909	1.000
Details of Markov Module												
First landcover image	LULC 2000											
Second landcover image	LULC 2020											
Time interval 1	20											
Time interval 2	10											
Proportional error	0.15											

Source: Own elaboration based on the Markov module in Terrset 2020

Table 3 indicates that Tree Cover and Dense Short Vegetation are highly stable (83.06% and 82.93%), due to protected areas. In contrast, Salt pans and wetlands with scattered vegetation show lower stability (35.73% and 43.35%), reflecting greater change. Semi-arid vegetation has a 29.37% chance of turning into open surface waters, while wetlands with sparse vegetation have a 43.22% likelihood. These transitions are spatially consistent with areas undergoing agricultural expansion, settlement growth, and vegetation conversion.

The 2030 projection indicates continued expansion of anthropogenic land uses and additional pressure on vegetation and wetland classes (Fig. 5). The model suggests further growth of cropland and built-up areas, while natural and semi-natural classes are expected to experience spatial redistribution. Because the projection is based on observed 2000–2020 transitions, it should be read as a business-as-usual scenario that does not explicitly include future policy shifts, climate shocks, or restoration interventions.

The projected decline of natural vegetation classes would reduce carbon storage, habitat continuity, and regulating services, while projected gains in built-up and cropland areas would intensify pressure on wetlands and mangrove-adjacent landscapes. This reinforces the need to interpret LULC projections together with spatial planning and conservation policies rather than as purely ecological trends.

Model validation showed acceptable performance for exploratory spatial planning. The comparison between the simulated and observed 2020 maps produced K-standard = 0.7226, K_{no} = 0.8443, and K_{location} = 0.9572, indicating moderate-to-high overall agreement, strong quantity agreement, and high spatial allocation accuracy. Quantity disagreement was 12.61%, whereas location disagreement was 1.66%, suggesting that the model is more dependable for spatial allocation than for exact quantity estimates.

3.3. Mangrove-specific analysis (1996–2020)

Mangrove cover increased by 0.53% from 1996 to 2020, with most growth (1.19%) occurring between 1996 and 2010 due to wetland recovery and sedimentation. From 2010 to 2020, mangroves declined by 42.67 ha, mostly because of agriculture, development, erosion, and flooding. Overall, there was a net annual gain of 1.42 ha (0.022%), with 563.21 ha gained and 529.08 ha lost (Fig. 6).

Mangrove loss accelerated to -4.28 ha/year over the last decade, highlighting growing human impacts that need to be addressed in conservation policy. Bajo Yuna Mangrove National Park (Zone 1) saw the largest gains, due to reduced human pressure (Fig. 7).

Note

1 indicates losses and 1 indicates gains

Figure 7 Spatial distribution of mangrove gains and losses in the BSE between 1996 and 2020. Values indicate loss (-1), no change (0), and gain (1). Insets show areas with higher concentration of mangrove change across the study area. Source: Own elaboration from ArcGIS Pro Raster Calculator

Miches, a municipality experiencing rapid tourism growth, has the largest mangrove loss in the study area, though its annual degradation rate is lower than prior estimates. Globally, 3.42% of mangroves were lost from 2000 to 2020, with an annual rate of 0.17% (Hamilton and Casey 2016). Myanmar, a deforestation hotspot, saw annual losses of over 3.6% between 1996 and 2016 (De Alban, Jamaludin et al. 2020). Malaysia’s Iskandar region reported 1.12% annual degradation due to urbanization and agriculture between 2000 and 2019 (Kanniah, Kang et al. 2021). On Vietnam’s southern coast, efforts reduced average annual mangrove loss from 3.6% (1998–2011) to 1.5% (2011–2023) (Tran, Reef et al. 2024).

Across the Caribbean, mangroves are declining by about 1% annually, with mainland areas seeing up to 1.7% per year (Cortés, Lorenzo-Trueba et al. 2024). The Tropical Northwestern Atlantic province experienced a 5.4% net decrease since 1996, with projections indicating further losses under climate change scenarios (Blanco-Libreros 2016, Troche, Lugo et al. 2024). Some regional reports estimate recent declines exceeding 30% (Rull 2023), and under severe sea-level rise, up to 75.9% could be submerged by 2060 (Troche, Lugo et al. 2024). In Mexico’s Mahahual-Xcalak, annual degradation was 0.85% from 1995–2007, mainly due to urban and infrastructure expansion (Hirales-Cota, Espinoza-Avalos et al. 2010).

3.4. Drivers of mangrove degradation during the period 1996–2020

Despite a net growth rate of 0.53% over 24 years, approximately 43 hectares of mangrove were lost in the last decade, driven by shifting economic activities and policy gaps (Table 4, Figs. 10, 11).

Table 4
Main drivers of mangroves degradation in BSE

Period	Net change (Ha)	Description	Anthropogenic drivers	Natural drivers
1996–2010	+ 76.81 Ha (1.19%)	Mangroves expansion due favorable conditions	Habitat restauration process Conservation of protected areas	Increased sedimentation Favorable climate conditions
2010–2020	-42.67 Ha (-0.65%)	Cover Loss due anthropogenic conditions	Deforestation and Coastal Urbanization Mass tourism Agriculture	Hurricanes Climate Change: Sea level rise Coastal erosion
1996–2020	+ 34.14 Ha (0.53%)	Balance between expansion and degradation	Conservation Policies Growth of unregulated economic activities	Climate change: Extreme events Marine pollution: Changes in salinity
Source: own elaboration based on literature revision				

Despite a net increase in mangrove extent over 1996–2020, the loss of approximately 43 ha during 2010–2020 indicates that recent pressures have altered the trajectory of the system. The observed pattern suggests a period of conditional resilience followed by localized decline, shaped by the interaction between sediment dynamics, protected-area management, and expanding economic activities (Table 4, Figs. 10 and 11). During 1996–2010, national environmental reforms, including Law 64 – 00 and the strengthening of the protected-area system, may have contributed to conservation outcomes in Bajo Yuna and Los Haitises, while favorable sedimentary conditions supported mangrove expansion. By contrast, the 2010–2020 period coincided with accelerated tourism development, urban expansion, and agricultural pressure, particularly in coastal municipalities experiencing land-market transformation.

These dynamics reflect a broader governance challenge. Mangroves generate public benefits, including shoreline protection, carbon storage, biodiversity conservation, and fishery support, yet many of these benefits are not internalized in land-use and tourism decisions. As a result, short-term private returns from real-estate development, mass tourism, and agricultural conversion can be prioritized over long-term ecosystem services. This creates social and ecological costs, including increased exposure to coastal hazards, biodiversity loss, reduced water quality, and diminished climate-regulation benefits (Hirales-Cota, Espinoza-Avalos et al. 2010, Garcés-Ordóñez, Ríos-Mármol et al. 2023).

Addressing these trade-offs requires an integrated policy response. Enforceable coastal zoning should identify non-build areas in critical mangrove belts and link setbacks to hazard exposure and blue-carbon priorities. Tourism regulation should include density controls, cumulative-impact assessment, environmental licensing, and no-net-loss requirements for mangrove conversion. Economic instruments, including payments for ecosystem services and blue-carbon finance, could help align local development incentives with conservation outcomes (Lau 2013). Finally, restoration should prioritize hydrological reconnection, including tidal-channel reopening, barrier removal, and site-appropriate planting where natural regeneration is insufficient (Simpson, Mercer Clarke et al. 2012, Moschetto, Ribeiro et al. 2021).

3.5. Restoration scenarios to 2030: mangroves of the Bay of Samana and their surroundings

Following the methodology described, mangrove restoration scenarios were created using estimated annual recovery rates (see Appendix A). The results show that the three projected scenarios differ based on conservation and restoration trends.

In the average gain scenario, mangroves would grow slowly to 6,499.75 ha by 2030 at 1.43 ha per year. The strong gain scenario projects faster restoration, reaching 6,583.56 ha with an annual increase of 9.81 ha. Conversely, continuous loss would shrink coverage to 6,443.08 ha, declining by 4.24 ha annually.

4. Conclusions

This study examined Land Use and Land Cover (LULC) changes in the Samaná Bay socio-ecological system from 2000 to 2020 and projected potential trajectories to 2030, with a specific focus on mangrove ecosystems. The findings provide three main conclusions for coastal conservation and planning in Caribbean SIDS.

First, the BSE exhibits conditional mangrove resilience with a recent localized decline. Mangroves showed a long-term net increase of + 0.53% between 1996 and 2020, with the strongest expansion during 1996–2010 (+ 1.19%), linked to favorable sedimentation and conservation policies. However, 2010–2020 recorded a loss of -42.67 ha, driven by agriculture, hotel development, urbanization, erosion, and flooding. This shows that mangrove resilience is not indefinite; once anthropogenic pressure exceeds ecological and governance thresholds, recovery becomes less certain.

Second, the 2030 scenarios reveal divergent futures shaped by trade-offs. Continued development without conservation measures would sustain mangrove decline, whereas targeted interventions, including coastal zoning, enforcement of setbacks, environmental licensing, and hydrology-first restoration, could stabilize or increase mangrove cover. The trade-off is clear: short-term private benefits from tourism and real-estate development can undermine long-term public benefits such as shoreline protection, carbon storage, fisheries productivity, and cultural values.

Third, land-sea interdependence is a central governance challenge. A ridge-to-reef perspective shows how upland agriculture, deforestation, sediment dynamics, and coastal construction cascade into mangrove degradation. Mangrove conservation in

Samaná Bay, therefore, cannot be addressed as an isolated issue within a protected area. It requires integrated governance that manages terrestrial, coastal, and marine pressures simultaneously.

Four policy implications follow from these results. Tourism and land-use regulation should strengthen coastal zoning, restrict construction in mangrove belts, enforce setbacks, and require cumulative-impact assessment for hotel and real-estate projects. Restoration strategies should prioritize community participation and hydrological reconnection through tidal-channel reopening and barrier removal. Monitoring systems should combine remote sensing, NDVI or other biophysical indicators, LiDAR where available, and ground validation to move beyond area-based assessments. Finally, economic incentives such as payments for ecosystem services and blue-carbon finance under Law 44 – 18 should be expanded to align local development incentives with ecosystem stewardship.

Overall, the results suggest that future mangrove trajectories in Samaná Bay will be strongly shaped by governance choices, particularly the capacity to regulate tourism-led land conversion, restore hydrological connectivity, and align local economic incentives with conservation. For the Dominican Republic and other Caribbean SIDS, reconciling tourism-led economic growth with coastal ecosystem conservation is urgent, feasible, and essential for long-term sustainability.

4.1. Limitations

This study has methodological and data limitations. The LULC analysis relies on available global datasets and selected temporal snapshots, which may affect transition probability estimates and reduce sensitivity to small changes or highly heterogeneous coastal areas. The CA-Markov model simplifies complex processes by estimating future change from previous states and does not explicitly incorporate future climate shocks, policy interventions, market shifts, or abrupt infrastructure development.

Mangrove analysis was based on changes in spatial extent and did not assess ecosystem condition, species composition, canopy height, biomass, or forest health. Therefore, apparent gains in area should not be interpreted automatically as ecological recovery. Future research should integrate field validation, higher-resolution imagery, LiDAR, NDVI or other vegetation indices, socio-economic drivers, and ecosystem service indicators such as fisheries, recreation, flood protection, and blue carbon. These improvements would reduce uncertainty and strengthen the use of spatial models for coastal zoning and mangrove conservation in the BSE.

Declarations

Supplementary Materials:

Appendix A provides data analysis supporting mangrove change and scenario calculations. Appendix B provides the CA–Markov model validation outputs.

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Author Contribution

C.C and V.G.V : conceptualization, methodology, formal analysis, investigation, data curation, visualization, writing—original draft, writing—review and editing. K.R and S.B.D: writing—review and editing.

Data Availability

The geospatial datasets used in this study are publicly available from the sources listed in Table 1 in the manuscript. The processed database, transition matrices, and generated spatial layers are available from the corresponding author upon reasonable request and will be deposited in an appropriate repository upon acceptance.

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Figures

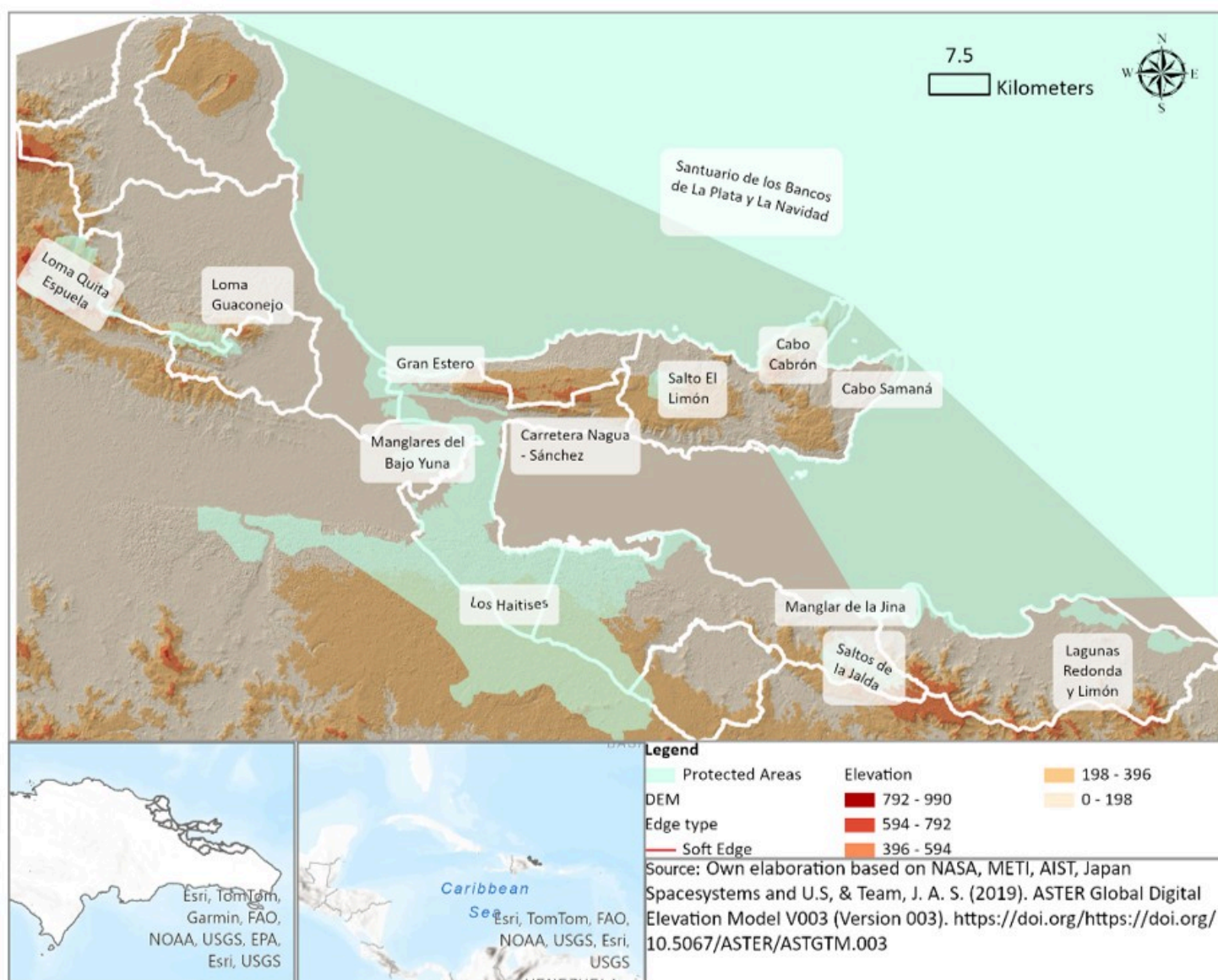


Figure 1

Study area of the Samaná Bay socio-ecological system (BSE), Dominican Republic, showing the terrestrial and marine study boundaries, protected areas, and elevation context. Elevation data are based on ASTER GDEM, and protected area boundaries were obtained from national and global spatial datasets. Source: Own elaboration.

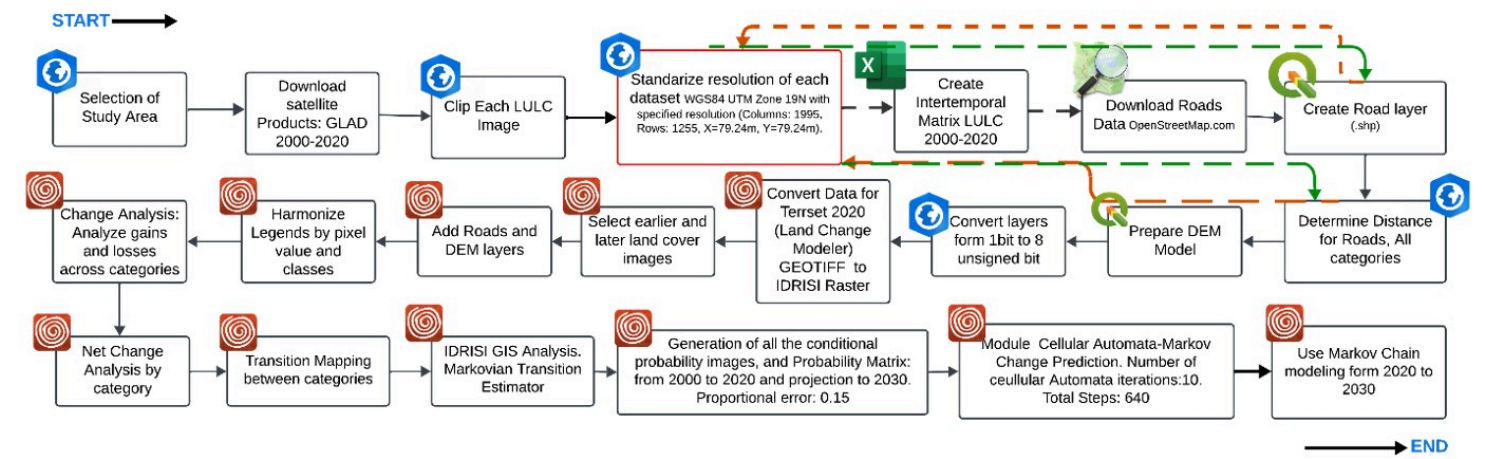


Figure 2

Workflow used for land-use and land-cover change analysis and CA-Markov projection in the Samaná Bay socio-ecological system. The process included data acquisition, spatial harmonization, reclassification, intertemporal change detection, transition-probability estimation, model validation, and projection to 2030. Source: Own elaboration.

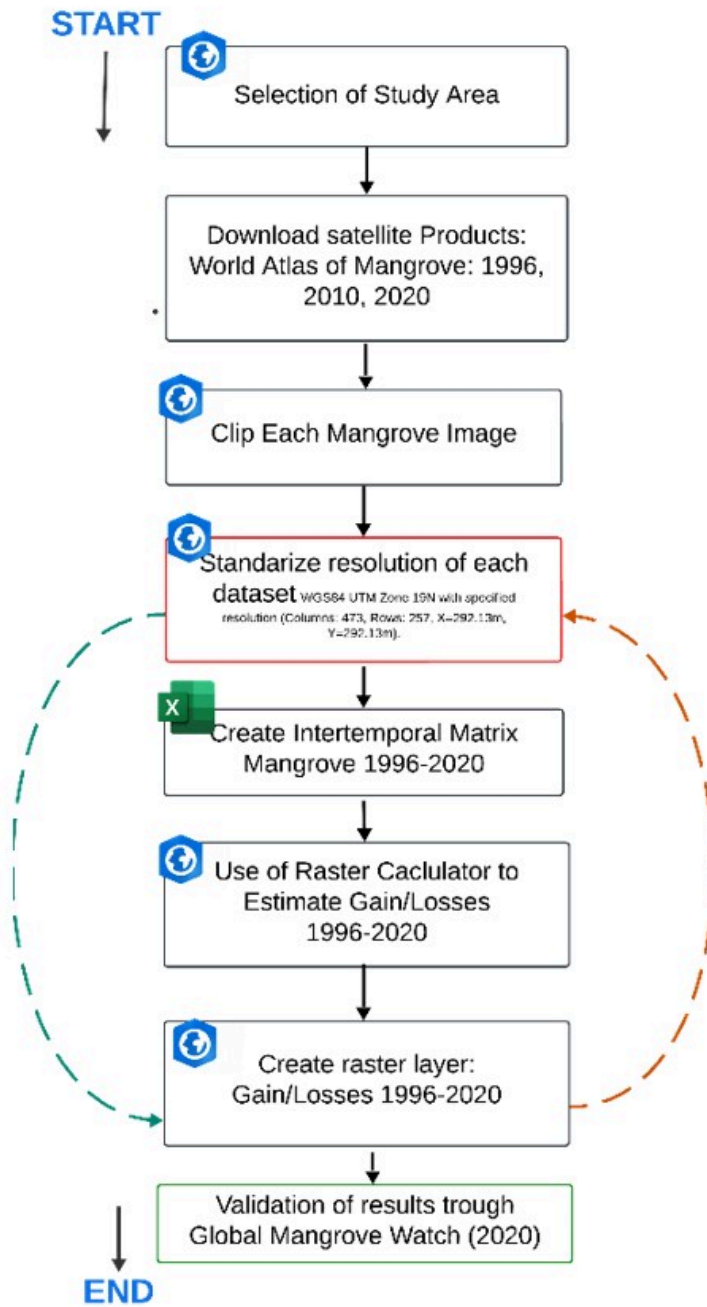


Figure 3

Workflow used for mangrove extent analysis in the BSE. Mangrove layers for 1996, 2010, and 2020 were clipped to the study boundary, spatially standardized, compared through intertemporal matrices, and processed with raster algebra to estimate gains, losses, and net change. Source: Own elaboration.

Land Use and Land Cover (LULC). Samana Bay and its Surroundings (BSE)

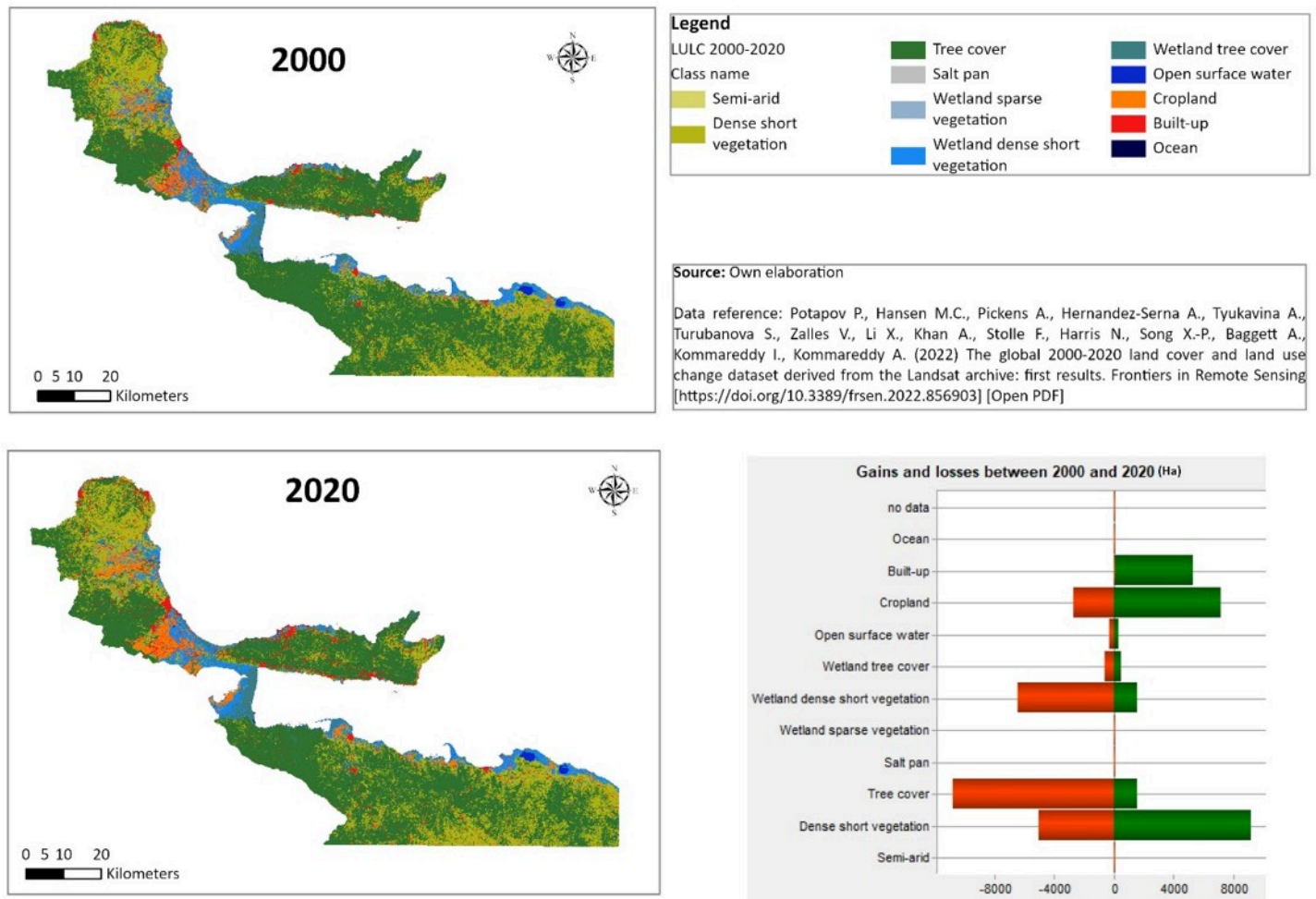


Figure 4

Land-use and land-cover distribution in the BSE in (a) 2000 and (b) 2020, and (c) net gains and losses by LULC class between 2000 and 2020. LULC data are based on the GLAD global land-cover product (Potapov et al. 2022). Source: Own elaboration.

Land Use and Land Cover (LULC) Projection. Samana Bay and its Surroundings (BSE)

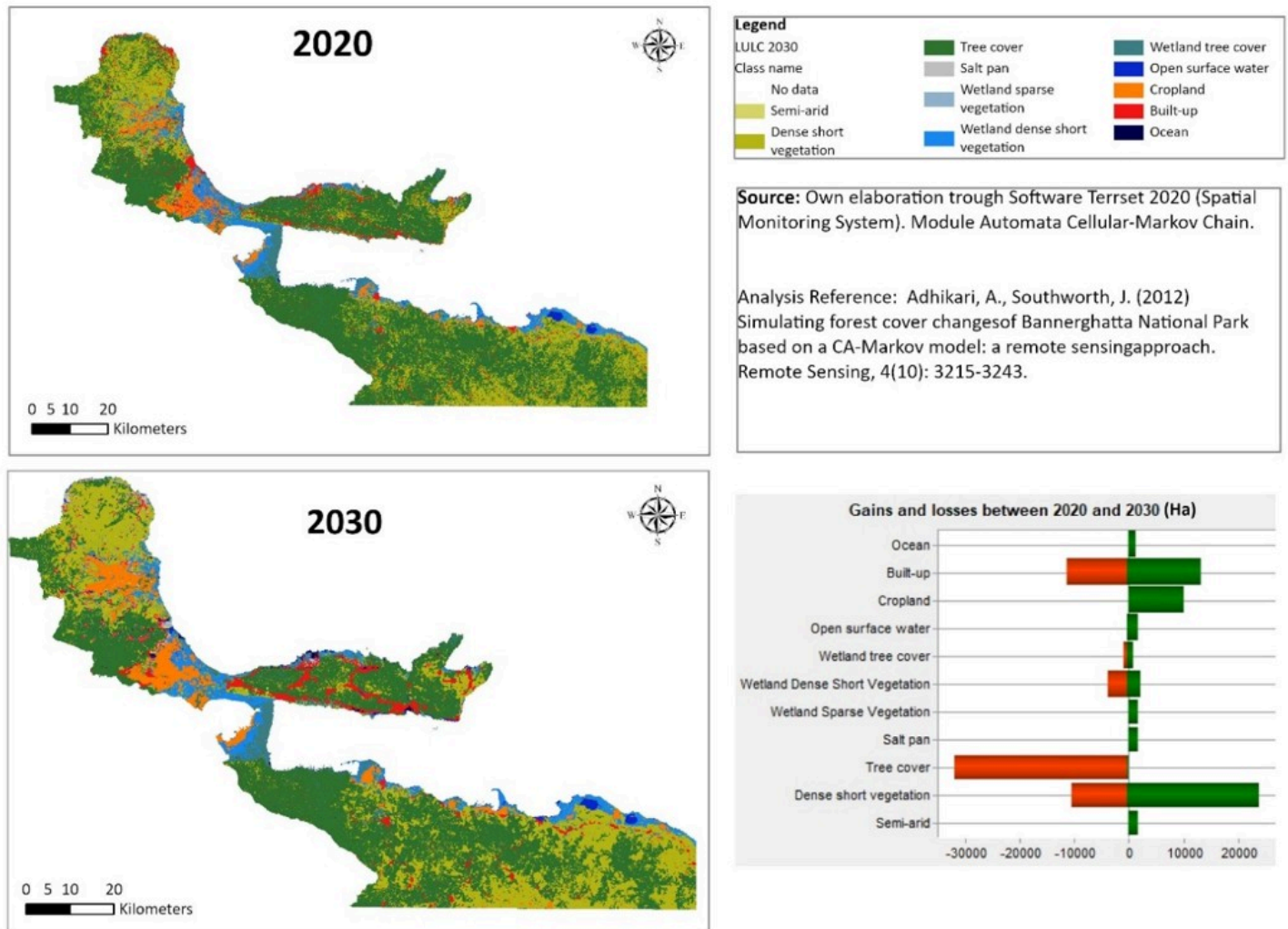
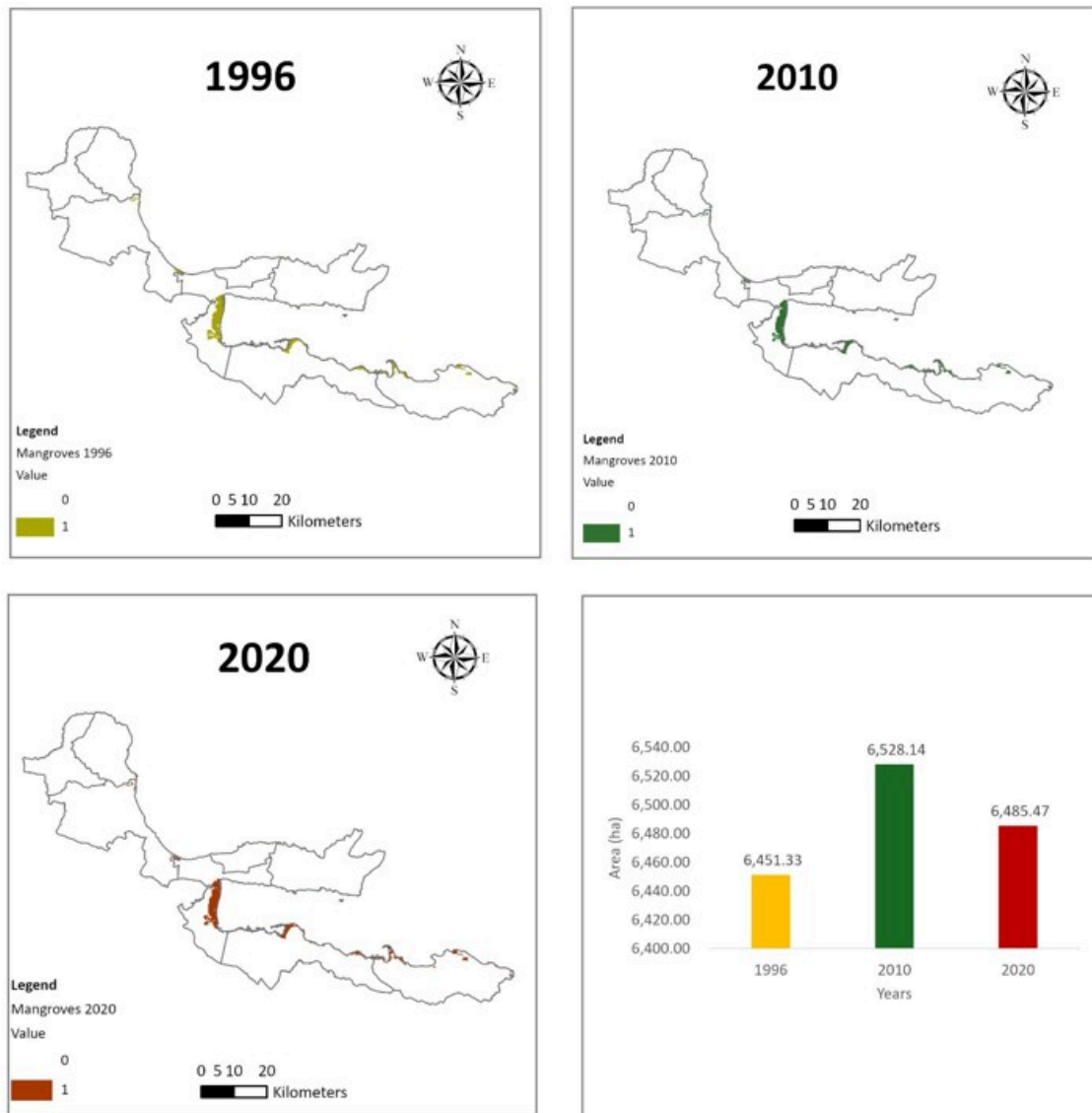


Figure 5

Observed LULC in 2020 and projected LULC in 2030 under a business-as-usual CA–Markov scenario. Panel (c) shows projected net gains and losses by class between 2020 and 2030. The projection is based on transition probabilities estimated from 2000–2020 LULC data. Source: Own elaboration.

Mangroves Ecosystem in Samana Bay and its Surroundings (BSE) 1996-2020



Source: Own elaboration based on Spalding M, Kainuma M, Collins L (2010). *World Atlas of Mangroves (version 3.1)*. A collaborative project of ITTO, ISME, FAO, UNEP-WCMC, UNESCO-MAB, UNU-INWEH and TNC. London (UK): Earthscan, London. 319 pp. Data DOI: <https://doi.org/10.34892/w2ew-m835>

Note: 1 indicates presence of mangroves

Figure 6

Mangrove extent in the BSE in (a) 1996, (b) 2010, and (c) 2020, with (d) total mangrove area by year. Value 1 indicates mangrove presence. Mangrove extent data are based on the global mangrove dataset used in this study. Source: Own elaboration.

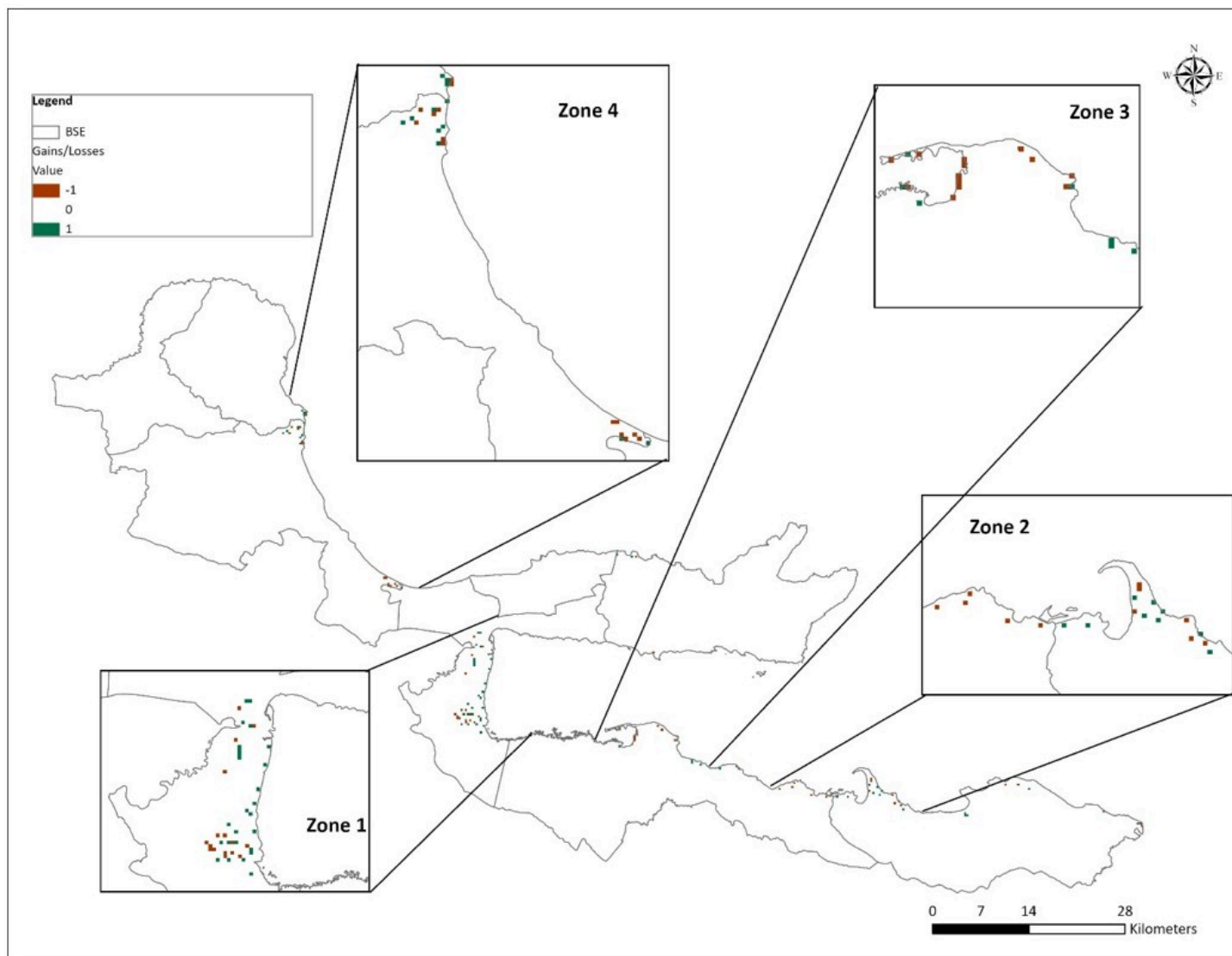


Figure 7

Spatial distribution of mangrove gains and losses in the BSE between 1996 and 2020. Values indicate loss (-1), no change (0), and gain (1). Insets show areas with higher concentration of mangrove change across the study area. Source: Own elaboration from ArcGIS Pro Raster Calculator

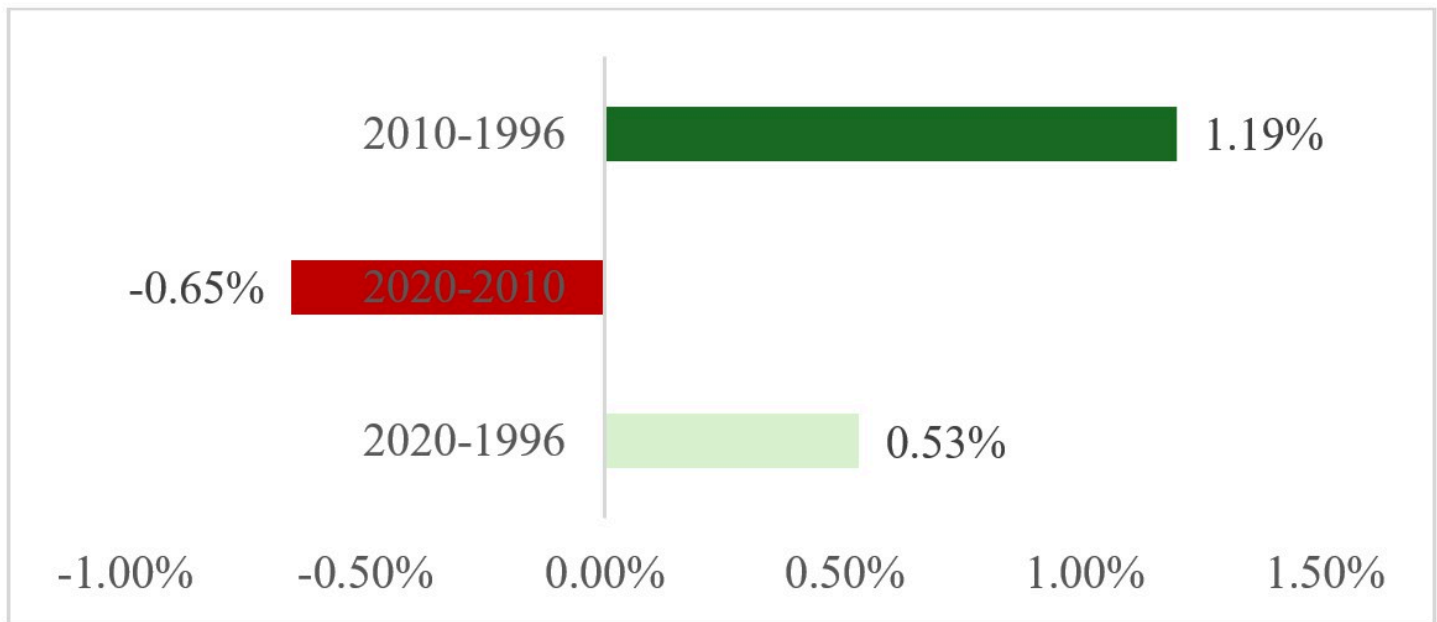


Figure 8

Variation and annual rate of mangrove change by period. Source: Own elaboration

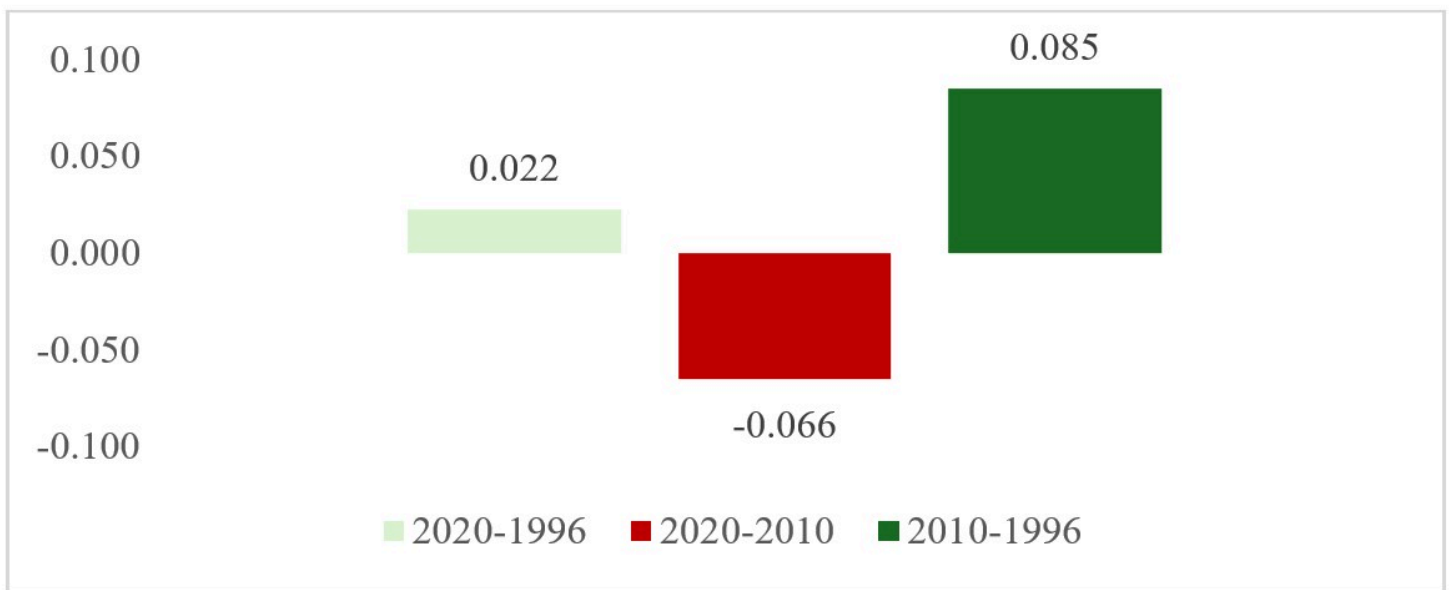


Figure 9

Variation and annual rate of mangrove change by period. Source: Own elaboration

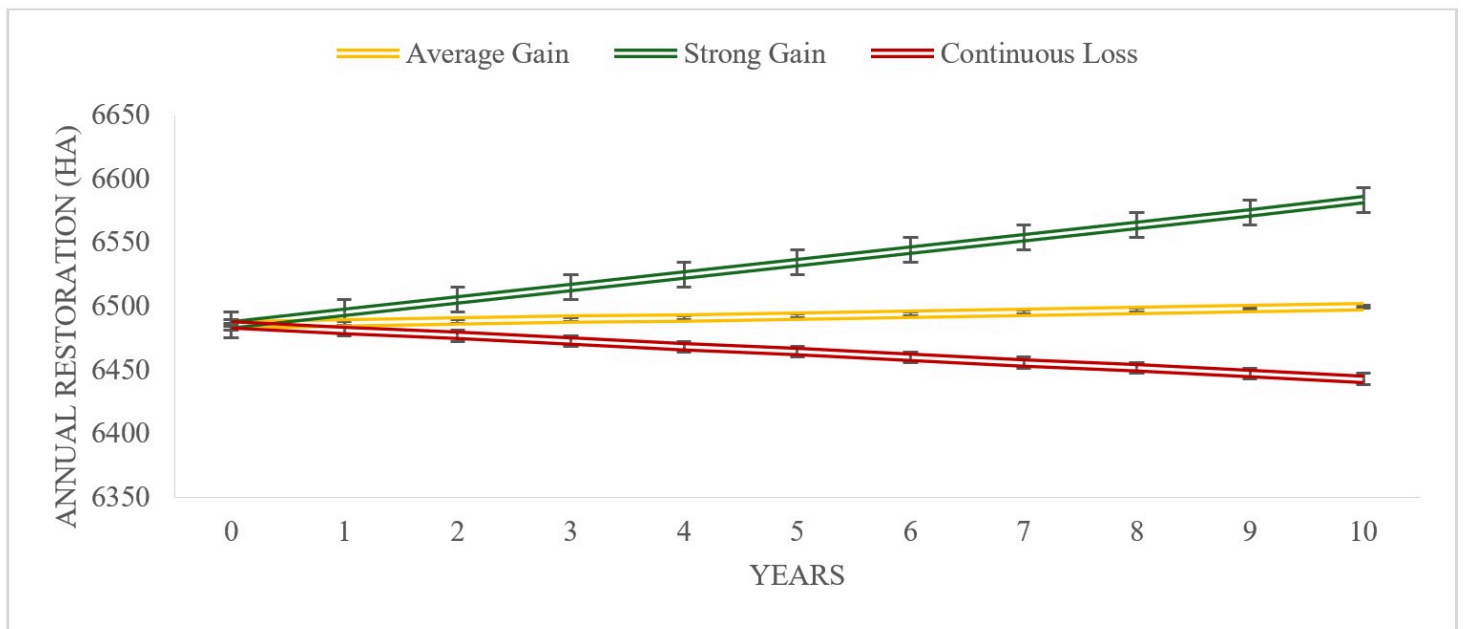


Figure 10

Mangrove restoration/conservation scenarios to 2030. Source: Own elaboration

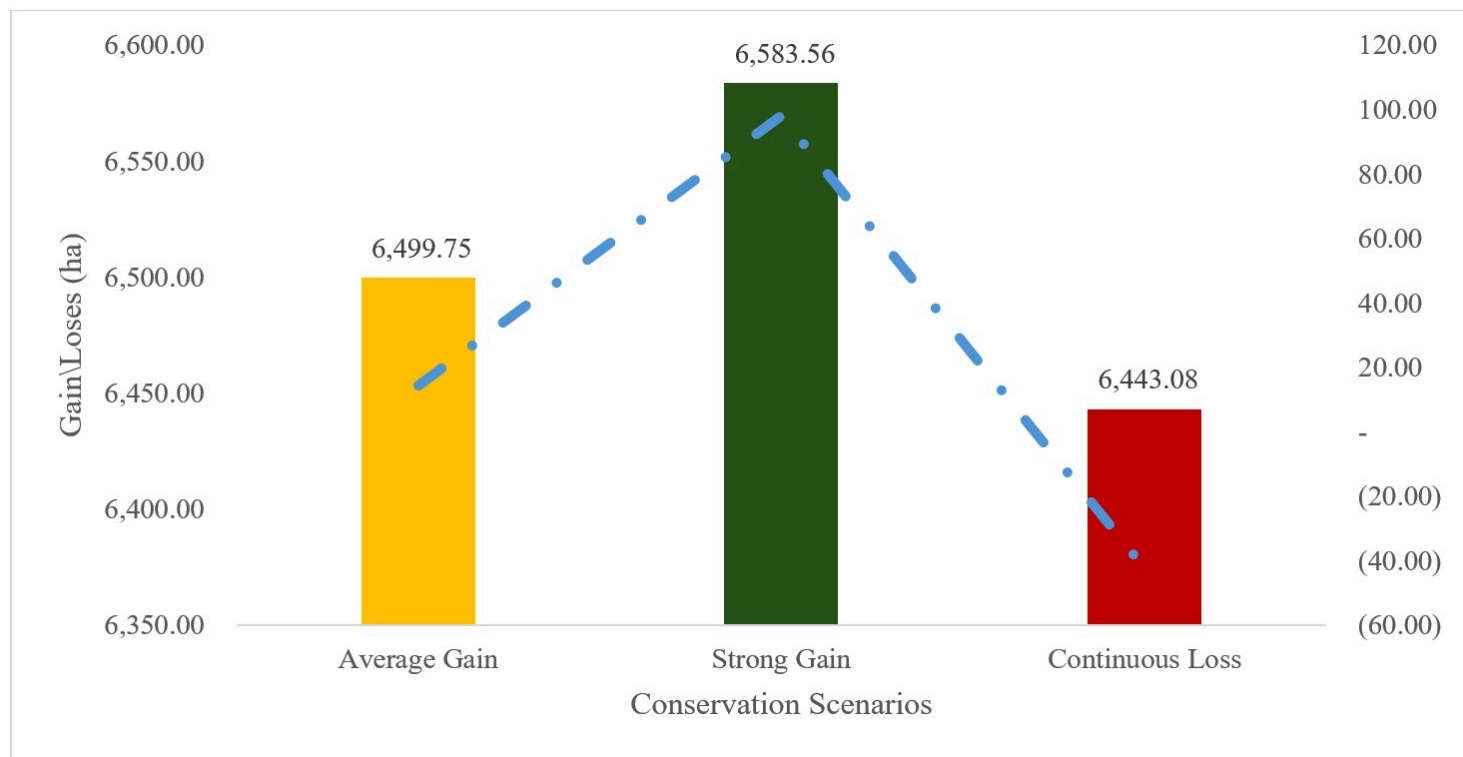


Figure 11

Annual trajectories by scenario. Source: Own elaboration

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- [AppendixA.DataAnalysis.xlsx](#)
- [AppendixBMarkovModelvalidation.txt](#)