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Abstract

This study develops an integrated framework for species specific mangrove habitat suitability and scenario dependent blue carbon planning in Soc Trang Province, Vietnam. The spatial model translates ecological zonation evidence for 11 mangrove taxa into local rules based on coastal distance and elevation, then tests those rules across 19 spatial scenarios. The carbon model carries the resulting group level suitable areas into a simplified Tier 2 assessment with conservative, central, and high parameter settings for sequestration and stock terms. Under the RF1 baseline, cumulative group level suitable area reached 47,997.312 ha. Avicennia spp. represented 15,752.821 ha, or 32.8% of the RF1 opportunity space, while Sonneratia spp., the Rhizophora group, the Interior upper complex, and the Brackish freshwater complex contributed 19.3%, 14.1%, 16.7%, and 17.0%, respectively. The RF1 central carbon outcome was 22.09 million t CO_2e over 30 years, with a conservative to high range of 18.83 to 25.27 million t CO_2e. Across all scenarios, central carbon outcomes ranged from 16.40 million t CO_2e in RF4 to 23.38 million t CO_2e in SC1. These results show that the framework is not merely a mapping exercise. It identifies where ecological opportunity is robust, where carbon outcomes are sensitive to spatial assumptions, and why credit style interpretation must remain separate from planning level blue carbon screening.

Keywords: mangrove, habitat suitability, blue carbon, uncertainty analysis, Soc Trang, Mekong Delta, species richness, restoration planning

1. Introduction

Intertidal mangroves are often introduced as blue carbon assets. That is true, but it is incomplete. Their scientific importance comes from the way carbon storage, shoreline protection, sediment trapping, fisheries support, and habitat provision are produced by the same living coastal structure, a framing increasingly emphasized in recent global mangrove assessments [49]. In low-lying tropical deltas, this structure is exposed to several pressures at once: erosion, subsidence, aquaculture expansion, altered hydrology, salinity change, and sea-level rise [1,3-5,42]. In the Mekong Delta, these pressures are no longer peripheral background conditions. They are already reshaping the geomorphic and hydrological setting in which mangroves establish, persist, or fail. Global mangrove extent products have made the distribution and recent change of mangroves increasingly visible at large spatial scales [2]. Yet such products do not by themselves answer the restoration question addressed here: which taxa are ecologically plausible within a particular deltaic landscape?

Vietnam's restoration experience makes the problem more concrete. Planting has often been treated as the visible sign of recovery, yet survival can remain poor when species are placed at the wrong elevation or in the wrong hydro-sedimentary setting [50,53]. Recent modeling work reaches the same conclusion from a different direction: restoration success depends on where mangroves are placed within the intertidal profile, not only on how many seedlings are planted [51]. For Soc Trang, the question is therefore not simply how much mangrove area might be restored. The more difficult question is which species are plausible in which part of the coastal landscape.

The first gap addressed here is ecological. Much applied mangrove mapping still treats mangroves as a single cover class, or at most as a condition class, even though restoration success depends on species fit. A binary map can say that a pixel is mangrove or non mangrove. It cannot tell a practitioner whether a seaward pioneer, a muddy estuarine Rhizophora stand, or a freshwater associated Nypa system is the more defensible restoration choice. This matters because species mismatch is not a cosmetic error. Recent eco-morphodynamic work shows that planting at the wrong elevation can alter sediment behaviour and reduce carbon storage potential, with poorly placed mangroves near mean sea level causing erosion that undermines the intended benefit [51].

The second gap is carbon interpretation. Blue carbon assessments often begin after the spatial ecology has already been simplified. In that sequence, carbon calculations may appear precise while resting on weak assumptions about where mangroves can actually establish or persist. The finance literature now warns against this separation. In Southeast Asia, Kwan et al. found that 85% of mangroves likely face some form of permanence risk from socioeconomic or climate pressures, which means that carbon value cannot be separated from location, exposure, and long term viability [52]. A planning framework for Soc Trang must therefore treat suitability as a condition of carbon interpretation, not as a decorative map placed before it.

Zonation provides the ecological language for doing this. Mangrove species sort along gradients of inundation, salinity, freshwater influence, substrate condition, and geomorphic position, although the local sequence is never perfectly fixed [21,27-30]. The point is not to impose a universal numeric niche template on Soc Trang. The point is to translate the best available ecological knowledge into local, inspectable rules that can be tested. In this study, coastal distance and elevation are used as practical proxies for that translation. They do not replace field salinity or hydroperiod measurements, but they make the assumptions visible enough to be challenged.

The study develops a species specific habitat suitability framework for 11 mangrove taxa in Soc Trang Province and links the resulting group level suitable areas to a simplified Tier 2 blue carbon model. The workflow uses five ecological groups, 19 spatial scenarios, and three carbon parameter settings. Its purpose is deliberately modest but useful: to show where ecological opportunity is strongest, how stable that opportunity remains when spatial assumptions change, and how carbon outcomes respond when suitable area is passed through group specific carbon parameters. The framework is not designed to issue credits. It is designed to make early stage restoration and conservation planning more honest about ecological fit, spatial uncertainty, and carbon value.

2. Materials and Methods

2.1 Study area

Soc Trang Province lies on the coastal margin of the Mekong Delta in southern Vietnam, where estuarine channels, tidal flats, aquaculture dominated backshore areas, and remnant or restored mangrove belts occur in close proximity. Recent satellite-based assessment of Soc Trang between 2000 and 2020 further shows that the province is not a static mangrove setting, but a landscape where forest extent and coastal land use have changed substantially [7]. This geography is suitable for a scenario based study because the province contains both seaward and landward habitat transitions, not a single uniform mangrove belt. Figure 1 defines the analytical boundary and proxy coastline used to calculate coastal distance. That boundary choice is methodologically important: every hectare later reported in Tables 4, 6, and 7 depends on this spatial frame.

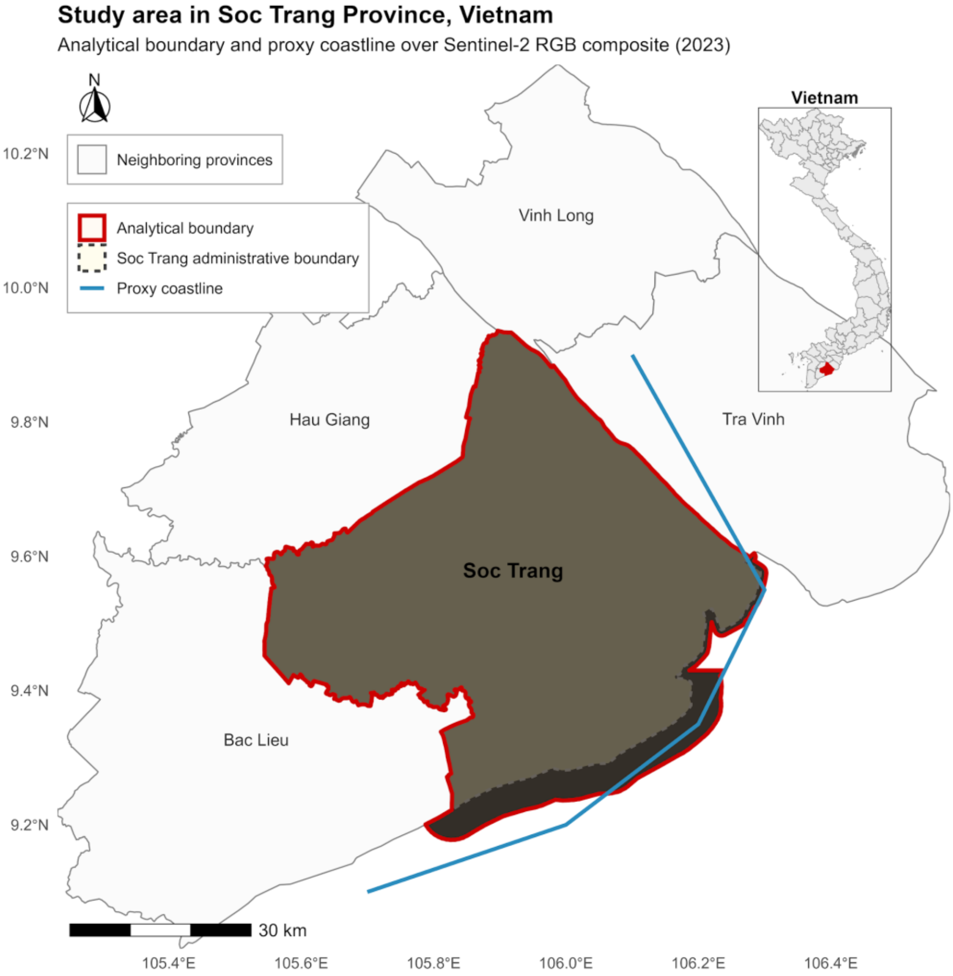

Figure 1. Study area and analytical boundary in Soc Trang Province, Vietnam

Source: Authors' own work

The spatial domain is not a neutral backdrop. It is the boundary within which the later RF1 estimate of 47,997.312 ha is allowed to exist. For that reason, the proxy coastline deserves as much attention as any parameter in the carbon model. It defines the coastDist surface used in the species rules, and a small displacement of that line can move a pixel from a seaward pioneer setting into an intermediate or landward class. This is why the coastline scenarios were retained as a separate uncertainty family rather than hidden inside preprocessing.

2.2 Overall analytical framework

The workflow begins with species ecology and ends with planning level carbon interpretation. Eleven taxa were first assigned to ecological positions using regional mangrove literature. Those positions were then converted into local rules based on coastal distance and DEM thresholds. The resulting suitability areas were summarized for 19 scenarios and passed into a simplified Tier 2 blue carbon model. This structure matters because it keeps the chain of inference visible. A reader can trace a carbon number back to a group area, a group area back to species rules, and a species rule back to ecological reasoning.

Table 1. Ecological grouping of mangrove taxa used in the habitat suitability framework

Species	Habitat group	Expected ecological position	Grouping rationale	Key supporting references

Species	**Habitat group**	**Expected ecological position**	**Grouping rationale**	**Key supporting references**
Avicennia alba	Avicennia spp.	Seaward pioneer on sheltered shores, in relatively saline coastal settings, along tidal river banks and seashore.	Assigned to the Avicennia group as a pioneer coastal taxon with strong saline tolerance and clear seaward affinity.	[21,22]
Avicennia marina	Avicennia spp.	Pioneer on sheltered shores and lower intertidal settings, able to colonize a wide range of tidal habitats including very saline environments.	Grouped with A. alba because both represent pioneer establishment, broad tidal tolerance, and strong affinity with seaward to lower intertidal saline habitats.	[21,22]
Sonneratia alba	Sonneratia spp.	Pioneer in low intertidal, sheltered coastal and estuarine settings, with limited tolerance of prolonged freshwater exposure.	Retained as a distinct Sonneratia group because it represents a seaward to estuarine pioneer niche that is ecologically coherent but not fully equivalent to the Avicennia niche.	[21,25]
Rhizophora apiculata	Rhizophora group	Core mangrove species of muddy intertidal estuarine environments, commonly associated with tidal waterways and regular tidal flooding.	Retained as a distinct Rhizophora group because it represents a central muddy estuarine mangrove niche with characteristic flooding and substrate conditions.	[21,25]
Bruguiera parviflora	Interior upper complex	Interior to intermediate mangrove species on consolidated mud and relatively less frequently inundated sites.	Assigned to the interior upper complex because it reflects less frequently inundated, more interior mangrove conditions than pioneer or core estuarine taxa.	[21,24]
Ceriops tagal	Interior upper complex	Dense shrub forming species on the landward edge of tidal forests, usually on better drained soils and inundated mainly by spring tides.	Grouped with interior upper taxa because it is typical of landward, better drained, higher intertidal positions.	[21,25]
Lumnitzera racemosa	Interior upper complex	Landward fringe to high intertidal species, often occurring on consolidated mud or sandy portions, commonly with freshwater influence.	Assigned to the interior upper complex because of its strong landward fringe affinity and association with high intertidal settings.	[21,22,27]
Aegiceras corniculatum	Brackish freshwater complex	Low intertidal estuarine species, often in creek edge or low tidal positions, but not a core seaward pioneer.	Grouped in the brackish freshwater complex because it occupies transitional estuarine settings and does not fit the core pioneer or interior upper categories.	[21,27]
Sonneratia caseolaris	Brackish freshwater complex	Landward mangrove associated with less saline tidal creeks and upstream reaches.	Assigned to the brackish freshwater complex because it is characteristic of landward tidal creek environments influenced by reduced salinity and stronger freshwater input.	[21,25]
Nypa fruticans	Brackish freshwater complex	Upper tidal waterway species on soft fine grained substrates, strongly dependent on perennial freshwater input and often forming pure stands.	Grouped in the brackish freshwater complex because it represents the strongest freshwater associated tidal habitat within the study system.	[21,26]
Excoecaria agallocha	Brackish freshwater complex	Landward margin species commonly occurring near beach swales or above the high tide mark.	Assigned to the brackish freshwater complex because of its landward margin ecology and marked dependence on freshwater influence.	[21,23]

Source: Authors' own work

The first reduction in ecological complexity is deliberate rather than mechanical. Eleven taxa are condensed into five carbon-relevant ecological groups, but the grouping preserves the gradient that matters for restoration: seaward pioneers, a muddy estuarine Rhizophora niche, interior upper taxa, and the brackish-freshwater assemblage. The largest group, the brackish-freshwater complex, contains four species because upstream tidal creeks and landward margins are ecologically heterogeneous in Soc Trang. By contrast, Sonneratia alba remains alone because its low-intertidal pioneer role would be blurred if it were merged with the less saline Sonneratia caseolaris niche.

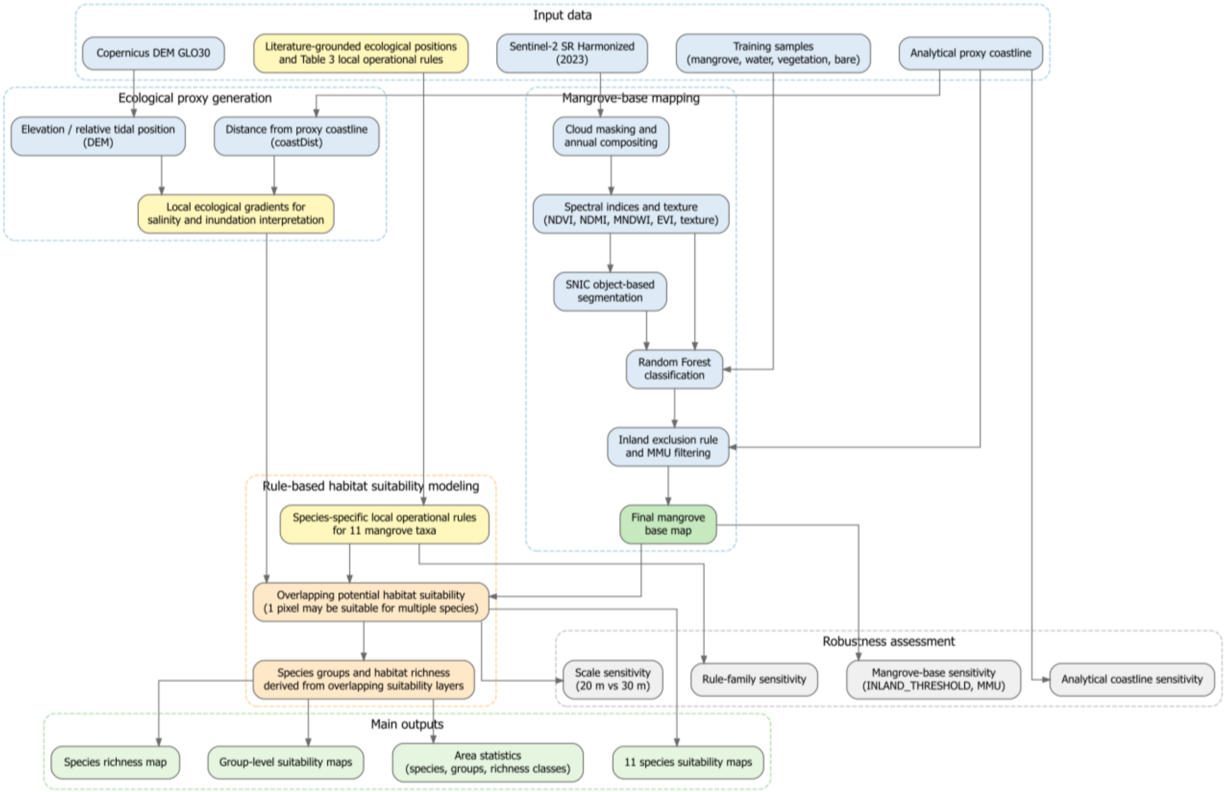

Figure 2. Habitat suitability workflow and scenario design

Source: Authors' own work

The analytical workflow makes the spatial logic of the study explicit before any carbon interpretation is introduced. The process begins with three kinds of evidence: ecological knowledge on species positioning, remote sensing inputs for mangrove-base mapping, and geomorphic proxies derived from DEM and the analytical coastline. These inputs are not treated as independent products. They are brought together to answer a narrower planning question: where can each mangrove taxon be considered ecologically plausible under the local conditions of Soc Trang?

The workflow therefore separates two tasks that are often mixed in mangrove mapping. The first task is to delineate a defensible mangrove base map using Sentinel-2 SR Harmonized imagery, spectral indices, SNIC segmentation, Random Forest classification, and inland exclusion filtering. The second task is to impose species-level ecological rules on that base, using coastal distance and relative elevation as local proxies for salinity exposure and inundation setting. This separation matters because a pixel classified as mangrove-like in spectral space is not automatically suitable for every species. Conversely, an ecologically plausible species zone should still be constrained by the mapped coastal landscape.

The final products are therefore not simple land-cover classes. They are overlapping suitability layers, species-group summaries, richness maps, and area statistics that preserve the possibility that one pixel may be suitable for more than one taxon. The robustness checks then ask whether these spatial conclusions survive changes in rule family, coastline representation, mangrove-base definition, and analytical scale. In this sense, the workflow is designed less as a one-off mapping exercise than as a transparent test of how strongly the inferred habitat opportunities depend on the assumptions used to construct them.

2.3 Species specific ecological rules

The analysis considered 11 taxa: Avicennia alba, Avicennia marina, Sonneratia alba, Rhizophora apiculata, Bruguiera parviflora, Ceriops tagal, Lumnitzera racemosa, Aegiceras corniculatum, Sonneratia caseolaris, Nypa fruticans, and Excoecaria agallocha. For carbon interpretation, these taxa were aggregated into five ecological groups. This aggregation is a modeling step, not a taxonomic claim. It reduces the number of carbon parameter sets while retaining the main ecological separation between seaward, intermediate, interior, and freshwater-influenced mangrove niches [21-28].

The rules were built from expected ecological position rather than imported as universal thresholds. This distinction is central to the study. Zonation is a relational pattern, shaped by tidal elevation, propagule movement, freshwater influence, and local geomorphology; it is not a fixed numeric staircase that can be copied from one delta to another [27-30]. Accordingly, coastDist and DEM thresholds are treated as Soc Trang operational rules. They state the assumptions clearly enough that future field salinity, inundation, or survival data can sharpen them.

Table 2. Species specific operational habitat suitability rules used in the study.

Species	Expected ecological position from literature	Final local operational rule	Confidence	Key supporting references
Avicennia alba	Seaward pioneer on sheltered shores, in more saline settings, along tidal river banks and seashore	coastDist < 7.5 km and DEM < 4.0 m	High	[21,22]
Sonneratia alba	Pioneer in low intertidal, sheltered coastal and estuarine settings; intolerant of long freshwater exposure	coastDist < 7.0 km and DEM < 4.0 m	Moderate to High	[21,25]
Avicennia marina	Pioneer on sheltered shores, able to colonize many tidal habitats, including very saline ones; common in intertidal flora	coastDist < 8.0 km and 2.0 m ≤ DEM < 12.0 m	High	[21,22]
Rhizophora apiculata	Core mangrove species on deep soft muddy soils flooded by normal high tides; often associated with tidal waterways with permanent freshwater input	3.0 km ≤ coastDist < 15.0 km and 4.0 m ≤ DEM < 12.0 m	High	[21,25]
Lumnitzera racemosa	Landward fringe to high intertidal species; prefers consolidated mud or sandy portions and often occurs with distinct freshwater influence	coastDist ≥ 8.0 km and DEM ≥ 8.0 m	High	[21,22,27]
Ceriops tagal	Dense shrublands on the landward edge of tidal forests, usually inundated by spring tides, on better drained soils	3.5 km ≤ coastDist < 14.0 km and 3.0 m ≤ DEM < 8.0 m	Moderate to High	[21,25]
Bruguiera parviflora	Interior to intermediate mangrove species; often associated with Rhizophora on consolidated mud and relatively less frequently inundated sites	4.0 km ≤ coastDist < 14.0 km and 6 m ≤ DEM < 12.0 m	Moderate to High	[21,24]
Aegiceras corniculatum	Low intertidal estuarine species; typically lower or middle low tidal rather than strictly inland; often associated with seaward or creek edge low positions	3.0 km ≤ coastDist < 12.0 km and DEM < 3.5 m	Moderate	[21,27,31,34]
Sonneratia caseolaris	Landward, less saline mangrove areas; muddy soils along tidal creeks and upstream tidal reaches	coastDist ≥ 9.0 km and 3 m ≤ DEM < 10.0 m	High	[21,25]
Nypa fruticans	Upper limits of tidal waterways on soft fine grained substrates; requires high perennial freshwater input and often forms pure stands	coastDist ≥ 12.0 km and DEM < 4.0 m	High	[21,26]
Excoecaria agallocha	Landward margin species; commonly on beach swales or above the high tide mark; requires freshwater input for a large part of the year	coastDist ≥ 10.0 km and DEM ≥ 9.0 m	High	[21,23]

Note: coastDist was calculated in meters in the spatial model but is reported in kilometers in this table for readability. DEM is reported in meters.

Source: Authors' own work

The operational rules define the ecological logic of the spatial analysis. They translate species-level ecological positioning into explicit local thresholds of coastal distance and elevation. Seaward pioneer taxa were intentionally constrained to low-elevation coastal settings. *Avicennia alba* was limited to areas within 7.5 km of the coast and below 4 m elevation, while *Sonneratia alba* was limited to areas within 7.0 km of the coast and below 4 m elevation. Intermediate and channel-associated taxa were assigned broader spatial windows. *Rhizophora apiculata*, for example, was allowed to occupy the 3–15 km coastal-distance band and the 4–12 m elevation range, reflecting its association with muddy estuarine settings and regular tidal influence. Landward and freshwater-associated taxa shifted the model toward more inland or higher positions. *Nypa fruticans* was restricted to areas at least 12 km from the coast but still below 4 m elevation, consistent with its affinity for upper tidal waterways with freshwater influence, whereas *Excoecaria agallocha* was assigned to more elevated landward margins, with coastDist ≥ 10 km and DEM ≥ 9 m. These rules therefore encode zonation as a local, testable spatial hypothesis for Soc Trang, not as a universal formula for mangrove distribution.

2.4 Sensitivity scenarios and mapping workflow

Coastal distance was used as a proxy for marine influence and relative estuarine position. Elevation was used as a proxy for inundation regime. Neither variable directly measures porewater salinity, hydroperiod, or sediment texture. That limitation is accepted rather than hidden. The strength of the design is that each proxy can be interpreted ecologically and stress tested spatially, which is more useful for planning than a black box surface whose errors are difficult to diagnose.

The scenario design was built to disturb the assumptions that matter. Eight rule family scenarios test ecological threshold sensitivity. Three coastline scenarios test the reference line used by coastDist. Six mangrove base scenarios test the support mask. Two scale scenarios test resolution effects. Together, the 19 scenarios ask a practical question: does the ecological interpretation survive when the mapping choices change? This question is central to uncertainty-aware suitability modelling because threshold choice, input representation, and model structure can all shift inferred habitat area [32,33].

Remote sensing is used here as support for ecological reasoning, not as a substitute for it. Random forest, object-based image analysis, and deep learning can map mangrove cover with strong performance in complex coastal landscapes [6,36-40]. Restoration planning, however, needs assumptions that practitioners can inspect. A rule-based structure makes those assumptions visible. It can later be coupled with machine learning, but it should not be hidden inside a black-box classifier when the main decision concerns species placement.

Table 3. Sensitivity scenario design across the 19 alternative settings

Scenario code	Scenario family	Configuration focus	Short description
RF1	Rule family	Baseline rule configuration	Main reference scenario used in spatial interpretation and blue carbon translation.
RF2	Rule family	Alternative rule configuration	Tests sensitivity to a stricter or shifted species rule combination.
RF3	Rule family	Alternative rule configuration	Tests the robustness of group level outcomes to modified species rule interactions.
RF4	Rule family	Alternative rule configuration	Assesses sensitivity to a more restrictive configuration affecting one

Scenario code	Scenario family	Configuration focus	Short description
			or more habitat groups.
RF5	Rule family	Alternative rule configuration	Assesses sensitivity to a more permissive configuration affecting overlap and richness.
RF6	Rule family	Alternative rule configuration	Targets sensitivity of inland coastal discrimination under modified rule assumptions.
RF7	Rule family	Alternative rule configuration	Tests rule sensitivity in a setting with broader overlap among ecological groups.
RF8	Rule family	Alternative rule configuration	Final within family rule perturbation case for comparison against RF1.
PC1	Coastline setting	Baseline proxy coastline	Uses the main analytical coastline representation for coastal distance calculation.
PC2	Coastline setting	Alternative proxy coastline	Tests sensitivity to an alternative coastline representation for coastDist.
PC3	Coastline setting	Modified proxy coastline	Examines how suitability shifts when the coastline proxy is broadened or repositioned.
MB1	Mangrove base setting	Baseline mangrove base	Uses the main mangrove base map before applying ecological suitability rules.
MB2	Mangrove base setting	Alternative mangrove base	Tests sensitivity to a narrower or modified mangrove support mask.
MB3	Mangrove base setting	Alternative mangrove base	Assesses robustness to a second modified mangrove base configuration.
MB4	Mangrove base setting	Alternative mangrove base	Evaluates how a broader mangrove base changes group level suitable area.
MB5	Mangrove base setting	Alternative mangrove base	Provides another within family check against the baseline mangrove base.
MB6	Mangrove base setting	Alternative mangrove base	Final mangrove base perturbation used in the robustness assessment.
SC1	Scale	20 m analytical scale	Tests spatial sensitivity when the analysis is conducted at finer resolution.
SC2	Scale	30 m analytical scale	Reference scale setting used for comparison and reproducibility.

Source: Authors' own work

The uncertainty design has a clear hierarchy. The eight RF scenarios test the ecological rules themselves. The three PC scenarios isolate coastline dependence. The six MB scenarios ask how much the support mask changes the opportunity space, and the two SC scenarios expose scale sensitivity. This structure matters because the final results show different magnitudes of sensitivity: the coastline family varies by about 4,816 ha, whereas the scale contrast reaches almost 9,723 ha between SC1 and SC2. The scenario design therefore does more than list alternatives; it identifies where methodological choices can materially change restoration priorities.

2.5 Species, groups, and richness products

Species maps were aggregated into group level area and an overlapping richness product. This separation is deliberate. A group map asks where a particular ecological option is plausible. A richness map asks where several options remain open at the same pixel. Those two products support different decisions. The first helps select taxa. The second helps identify places where restoration design can remain flexible if field conditions or nursery availability change.

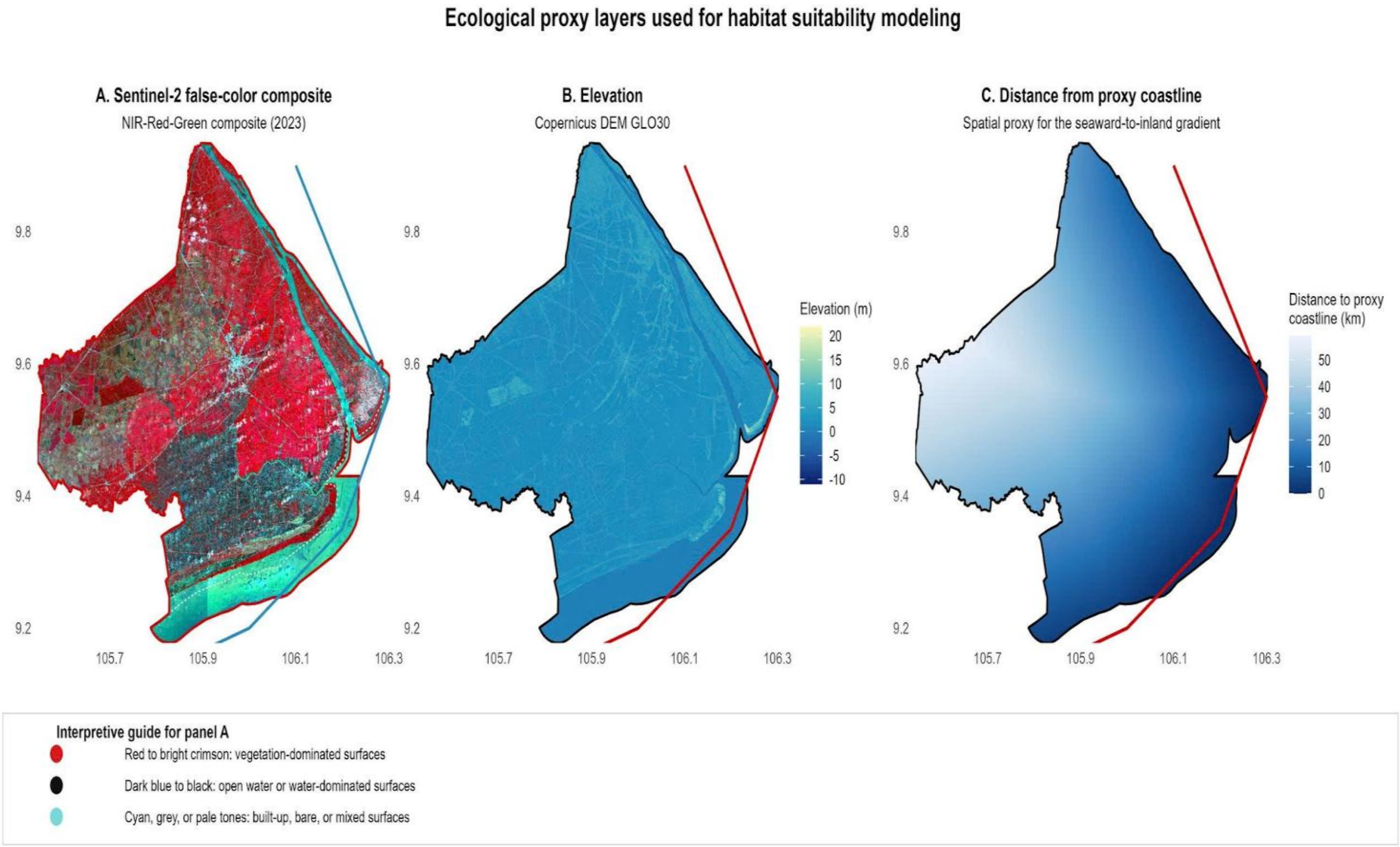

Figure 3. Ecological proxy layers used for habitat suitability modeling

Source: Authors' own work

The physical proxy layers supply the grammar for the species rules. The coastal-distance thresholds recur around three meaningful bands: below about 7-8 km for pioneer settings, up to roughly 15 km for intermediate/channel taxa, and beyond 8-12 km for more landward or freshwater-associated taxa. The DEM thresholds separate low intertidal positions below 4 m from intermediate surfaces around 4-12 m and higher landward positions above about 8-9 m. These numbers should not be read as physiological limits. They are operational translations of zonation evidence into a reproducible Soc Trang rule set.

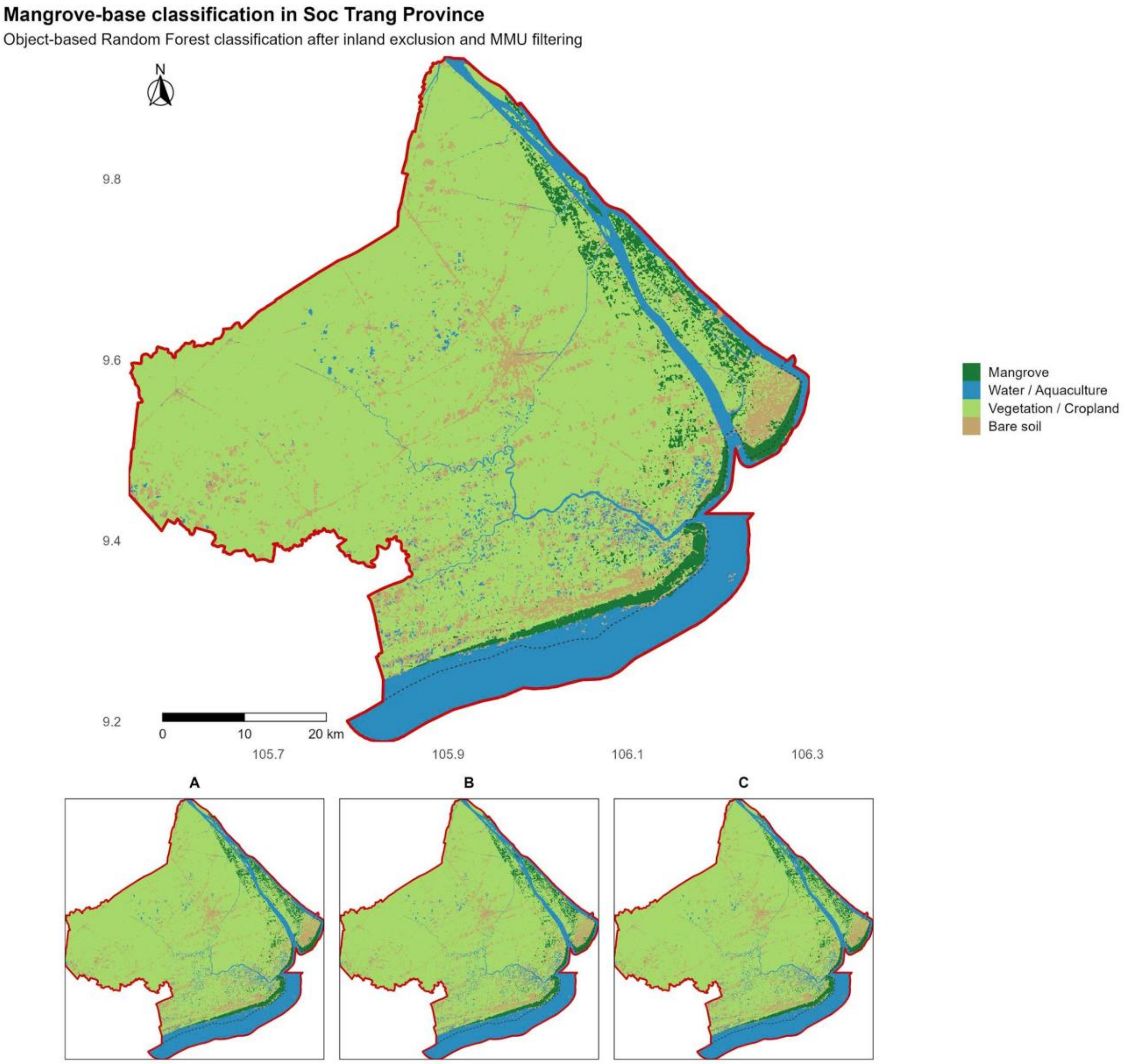

Figure 4. Mangrove base classification in Soc Trang Province

Source: Authors' own work

The support mask is one of the strongest spatial controls in the study. Within the mangrove base family, total cumulative suitable area ranges from 38,085.345 ha in MB3 to 45,033.373 ha in MB4, a difference of 6,948.028 ha. That spread is too large to be treated as a cartographic detail. It means that the base layer can alter the apparent restoration opportunity by an area comparable to the entire RF1 Rhizophora group. For a planning application, the map boundary and mask are therefore not housekeeping steps; they are part of the uncertainty analysis.

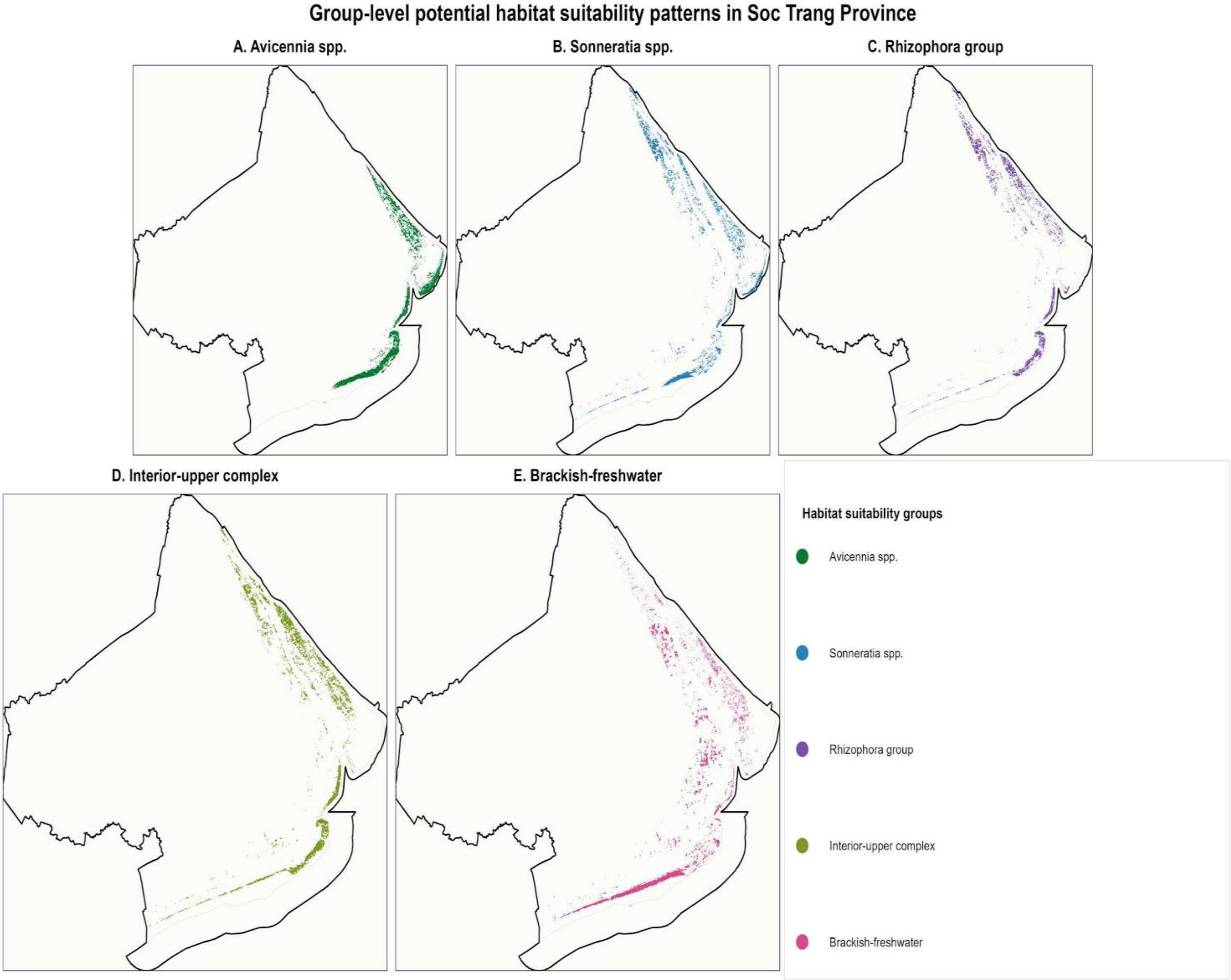

Figure 5. Group level potential habitat suitability patterns in Soc Trang Province

Source: Authors' own work

The mapped group pattern gives the first substantive ecological result. Under RF1, Avicennia spp. has the largest opportunity space, 15,752.821 ha, equal to 32.8% of cumulative RF1 suitability. Sonneratia spp. follows with 9,276.192 ha, while the Brackish freshwater complex and Interior upper complex contribute 8,172.456 and 8,005.188 ha, respectively. Rhizophora is smaller in area, 6,790.655 ha, but later becomes much more important in the carbon results because its group parameters are not proportional to area alone. The ecological lesson is therefore already visible at the mapping stage: the largest habitat opportunity is not automatically the largest carbon opportunity.

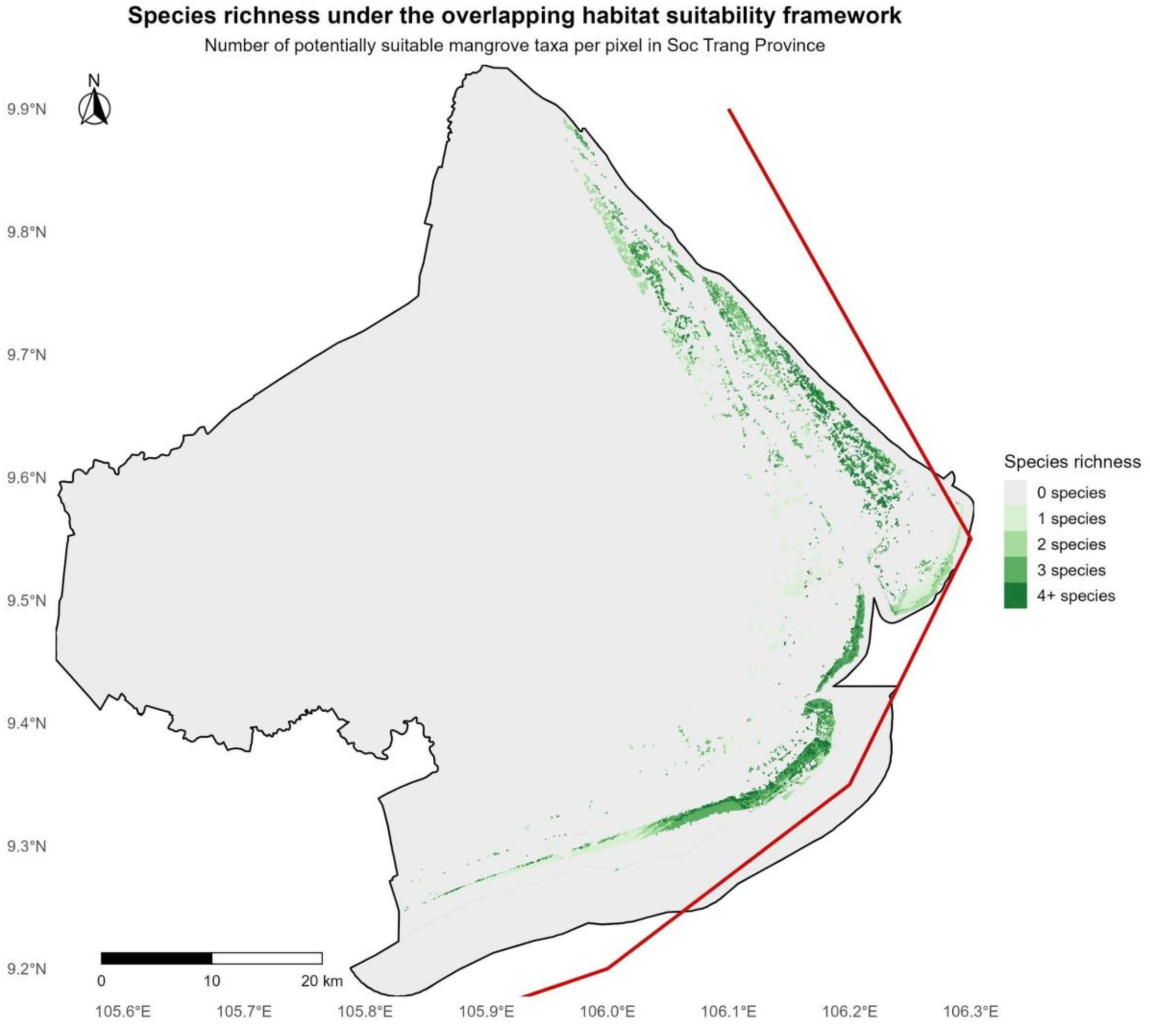

Figure 6. Species richness under the overlapping habitat suitability framework

Source: Authors' own work

The richness layer changes the question from area to choice. A pixel with several plausible taxa is not more forest in a land-cover sense; it is a place where restoration design has more ecological flexibility after field validation. A pixel with only one plausible taxon is more constrained. Such areas may still be valuable, but they leave less room for correcting species choice if salinity, substrate, or seedling survival differ from the proxy-based expectation. This makes richness a practical risk layer, not simply a biodiversity-looking visualization.

2.6 Blue carbon parameterization and scenario analysis

The blue carbon module is a Tier 2 style decision-support model rather than a crediting methodology. This framing is consistent with the broader blue carbon literature, which recognizes the climate significance of mangroves while also emphasizing the need for careful accounting of sequestration, stocks, and site conditions [43]. It receives group-level suitable area from the spatial analysis and applies literature-based parameters for sequestration rate, aboveground biomass stock, and soil organic carbon stock. Sequestration is allowed to vary across conservative, central, and high settings because this is where the planning uncertainty is expected to be largest. Stock terms are held at central values in the main run so that the experiment remains interpretable [9-17].

Table 4. Group level suitable area outcomes across the 19 scenarios

Scenario	Family	Avicennia spp. (ha)	Sonneratia spp. (ha)	Rhizophora group (ha)	Interior upper complex (ha)	Brackish freshwater complex (ha)	Total suitable area (ha)

Scenario	Family	Avicennia spp. (ha)	Sonneratia spp. (ha)	Rhizophora group (ha)	Interior upper complex (ha)	Brackish freshwater complex (ha)	Total suitable area (ha)
RF1	Rule family	15,752.821	9,276.192	6,790.655	8,005.188	8,172.456	47,997.312
RF2	Rule family	10,220.414	5,220.532	6,893.633	8,173.594	8,177.116	38,685.288
RF3	Rule family	13,238.150	7,387.991	9,639.873	8,173.594	8,177.116	46,616.724
RF4	Rule family	12,990.217	7,273.875	3,294.901	8,005.188	8,172.456	39,736.637
RF5	Rule family	12,990.217	7,273.875	6,790.655	14,388.659	8,172.456	49,615.862
RF6	Rule family	13,238.150	7,387.991	6,893.633	3,495.476	8,177.116	39,192.366
RF7	Rule family	13,238.150	9,616.867	6,893.633	8,173.594	11,474.631	49,396.876
RF8	Rule family	12,990.217	6,677.466	6,790.655	8,005.188	4,672.004	39,135.530
PC1	Coastline setting	13,238.150	7,387.991	6,893.633	8,173.594	8,177.116	43,870.484
PC2	Coastline setting	11,595.526	6,419.367	6,606.589	8,312.125	7,297.355	40,230.962
PC3	Coastline setting	13,730.984	7,072.573	5,569.288	6,085.513	6,596.010	39,054.368
MB1	Mangrove base setting	13,440.434	6,155.431	5,788.641	7,570.744	6,367.464	39,322.714
MB2	Mangrove base setting	13,238.150	5,938.336	5,640.518	7,324.394	6,160.404	38,301.802
MB3	Mangrove base setting	13,222.246	5,852.441	5,625.350	7,302.517	6,082.791	38,085.345
MB4	Mangrove base setting	13,440.434	7,653.911	7,076.127	8,458.017	8,404.885	45,033.373
MB5	Mangrove base setting	13,238.150	7,387.991	6,893.633	8,173.594	8,177.116	43,870.484
MB6	Mangrove base setting	13,222.246	7,276.544	6,853.441	8,137.353	8,039.571	43,529.156
SC1	Scale	14,482.931	9,357.048	8,811.254	10,370.059	10,572.651	53,593.943
SC2	Scale	13,238.150	7,387.991	6,893.633	8,173.594	8,177.116	43,870.484

Source: Authors' own work

The spatial outputs become quantitatively meaningful when read across all scenarios. Total cumulative suitable area spans 38,085.345 ha in MB3 to 53,593.943 ha in SC1, while RF1 sits high in the distribution at 47,997.312 ha. The mean across scenarios is 43,112.616 ha, so RF1 is not an average case; it is a relatively expansive but still plausible reference. This matters for interpretation because RF1 is used for the main carbon translation. Its value should be read as a baseline for decision exploration, not as a uniquely correct spatial truth.

Table 5. Parameterization of the Tier 2 simplified blue carbon model

Symbol	Parameter	Value	Unit	Interpretation in the model	Key supporting references
$A_{g,s}$	Suitable area of habitat group g under scenario s	Scenario specific	ha	Group specific suitable area imported from the habitat suitability analysis and summarized in Table 4.	[21,31,34]

Symbol	Parameter	Value	Unit	Interpretation in the model	Key supporting references
$A_{s,total}$	Total suitable area across groups under scenario s	Scenario specific	ha	Sum of suitable area across the five habitat groups under scenario s.	[21,31,34]
α	Protected or treated fraction	0.60	proportion	Fraction of suitable area entering blue carbon accounting.	[9,13,19]
λ	Loss risk fraction	0.01	proportion	Fraction of treated area interpreted as exposed to avoidable loss.	[9,12,19]
r_g	Group specific sequestration rate	Conservative, central, high	$tCO_2e\ ha^{-1}\ yr^{-1}$	Literature grounded sequestration rate assigned at habitat group level.	[11,13,15,16,17,18]
B_g	Group specific aboveground stock	Central	$tCO_2e\ ha^{-1}$	Group specific aboveground biomass stock used in the avoided loss term.	[10,11,16,17]
S_g	Group specific soil organic carbon stock	Central	$tCO_2e\ ha^{-1}$	Group specific soil carbon stock used in the avoided loss term.	[10,11,17]
τ	SOC release horizon	9	Yr	Characteristic time scale governing decay of the soil carbon release term.	[9,12,13]
β	Buffer fraction	0.20	proportion	Conservative deduction applied to raw carbon outcomes.	[13,19]
k	Sequestration growth coefficient	0.32	yr^{-1}	Controls the steepness of the logistic sequestration pathway.	[11,13,15]
t_0	Midpoint year of sequestration transition	4.0	yr	Inflection point of the logistic sequestration pathway.	[11,13,15]
T	Simulation horizon	30	yr	Total time horizon used to aggregate scenario outcomes.	[9,13,19]

Source: Authors’ own work

The parameter set makes the carbon calculation traceable. In RF1, $\alpha = 0.60$ converts 47,997.312 ha of cumulative opportunity into 28,798.387 ha of treated area. The loss-risk fraction $\lambda = 0.01$ then places about 287.984 ha into the avoided-loss component. The buffer $\beta = 0.20$ and the 30-year horizon keep the output conservative, while $k = 0.32\ \text{yr}^{-1}$ and $t_0 = 4$ yr impose a delayed sequestration response. Those choices explain why the modeled outcome is driven mainly by accumulated sequestration rather than by immediate avoided loss.

Within the model, suitable area is first converted to treated area and then to an at risk area. Annual carbon outcome is then divided into avoided aboveground biomass loss, avoided soil organic carbon loss, and ongoing sequestration. The logistic term is included because restoration benefits rarely appear as a straight line from year 1. Early establishment is slower, carbon accumulation increases as stands develop, and later years carry a larger share of the cumulative benefit. This temporal structure is essential for interpreting the economic results in Section 3.3.2.

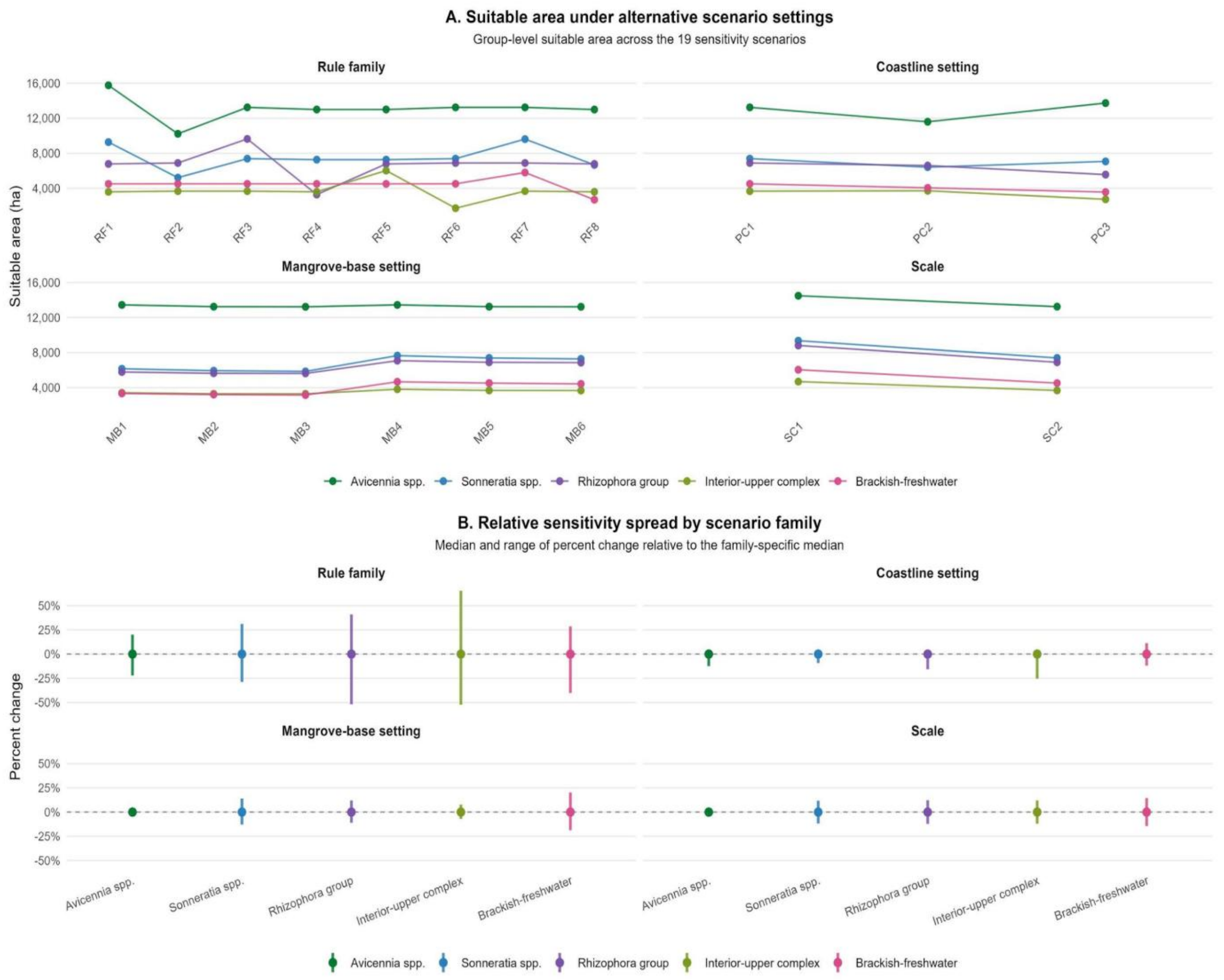

Figure 7. Blue carbon modeling workflow and uncertainty analysis

Source: Authors' own work

The carbon workflow is best read as an accounting discipline rather than as a claim of credit issuance. Beginning with RF1, 47,997.312 ha of cumulative group-level opportunity becomes 28,798.387 ha of treated area before risk, buffer, and time-dependent accumulation are applied. Each step reduces or reshapes the raw spatial signal. That is the reason the model can be useful for planning: the final number is not a direct multiplication of hectares by a generic carbon coefficient.

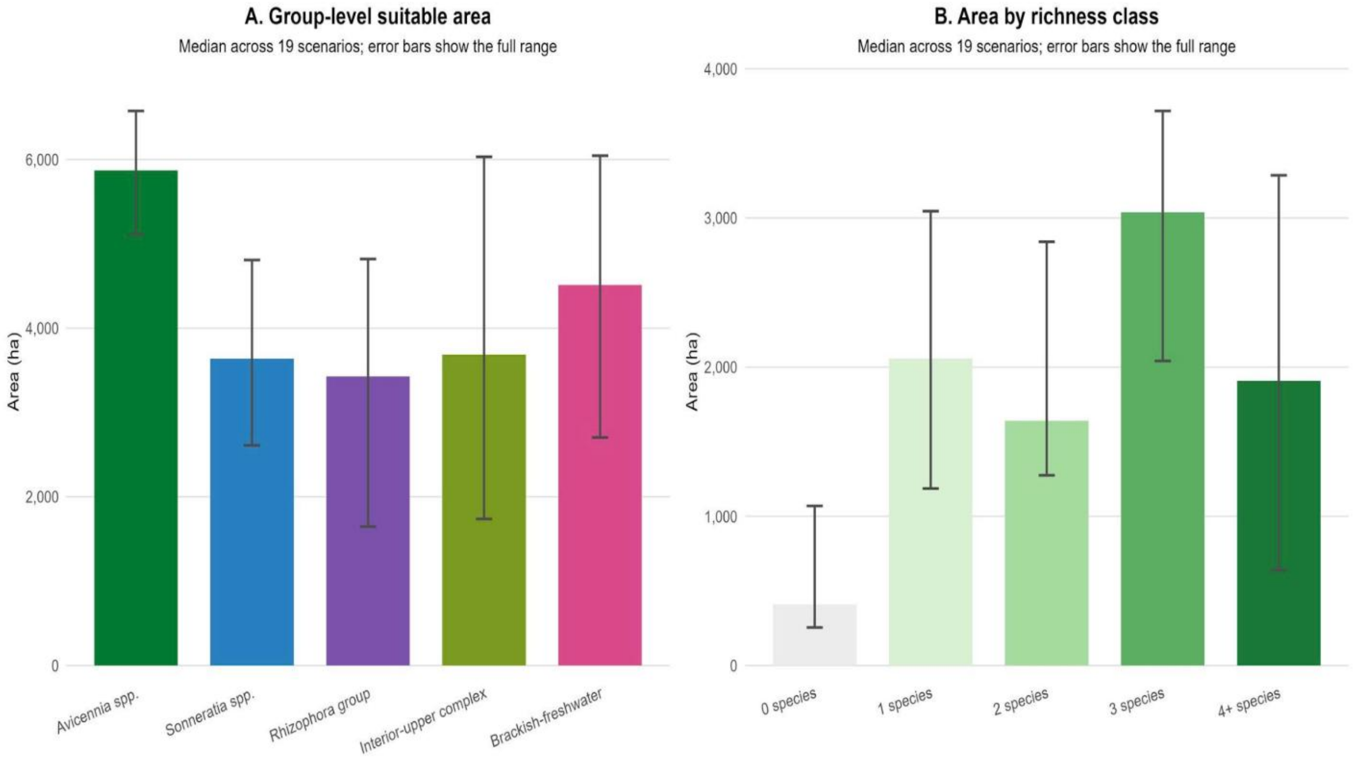

Figure 8. Summary area statistics across scenarios.

Source: Authors' own work

The area envelope is not symmetrical. SC1 produces 53,593.943 ha, about 11.7% above RF1, whereas MB3 produces 38,085.345 ha, about 20.7% below RF1. The lower tail is therefore steeper than the upper tail. In practical terms, conservative choices about the mangrove base can remove more opportunity area than finer scale analysis can add. This is an important warning for restoration planning because overly narrow masks may prematurely exclude places that are ecologically plausible but not yet expressed as current mangrove cover.

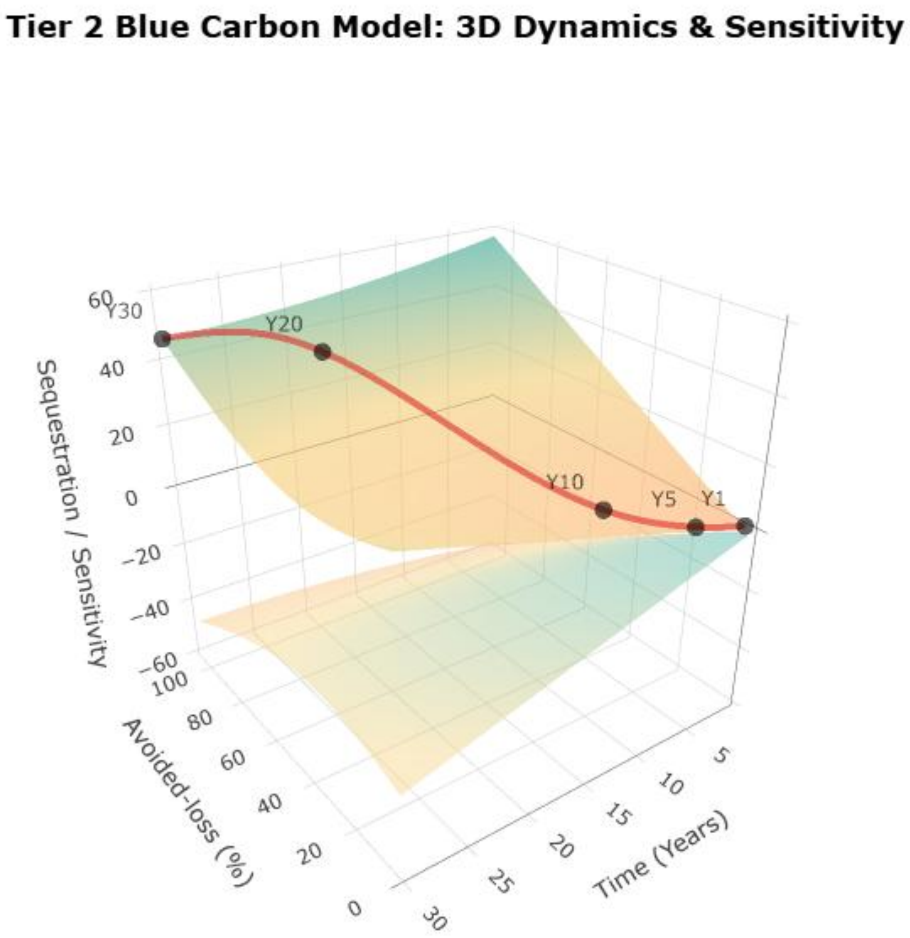

Figure 9. Tier 2 blue carbon model: integrated 3D dynamics and sensitivity manifold

Source: Authors' own work

The carbon response surface has structure rather than noise. RF1 spans 18.83 to 25.27 million t CO_2e from the conservative to high parameter set, a spread of 6.44 million t CO_2e around the 22.09 million t CO_2e central value. That spread is not random. It follows the group composition of the scenarios and the sequestration assumptions assigned to each group. The figure therefore supports a central claim of the manuscript: uncertainty is not something to hide after the model; it is part of the result.

3. Results

3.1 Baseline spatial patterns

Under the RF1 baseline, cumulative group level suitable area reached 47,997.312 ha. Avicennia spp. was the largest contributor, with 15,752.821 ha, equivalent to 32.8% of the RF1 total. Sonneratia spp. contributed 9,276.192 ha, or 19.3%. The Brackish freshwater complex, Interior upper complex, and Rhizophora group followed with 8,172.456 ha, 8,005.188 ha, and 6,790.655 ha, respectively. These values should not be read as mutually exclusive land cover classes. They are cumulative ecological opportunity areas produced by overlapping suitability rules.

The RF1 pattern has a clear ecological order. Avicennia and Sonneratia dominate the seaward opportunity space, while Rhizophora occupies a narrower but carbon important intermediate band. The landward and freshwater associated groups cover smaller or more fragmented zones, but their planning value is not minor. They mark the places where a generic coastal planting strategy would be least defensible. In other words, the map does not simply show where mangroves might be planted. It shows where the choice of taxon becomes ecologically consequential.

The richness product adds a second layer of interpretation. Where several taxa overlap, the planner has room to adjust species choice after field checking. Where only one taxon remains plausible, the margin for error is much smaller. This is particularly important in Soc Trang because hydrology and salinity can change over short distances. A richness map is therefore not a biodiversity census; it is a map of decision flexibility.

3.2 Scenario robustness

The 19 scenarios show a framework that is sensitive but not unstable. Total cumulative suitable area ranges from 38,085.345 ha to 53,593.943 ha. The lowest value occurs in MB3, which confirms the strong influence of the mangrove base mask. The highest value occurs in SC1, showing that scale can open additional opportunity space. Despite this range, the broad ecological ordering remains legible: seaward pioneers remain seaward, Rhizophora remains intermediate, and the freshwater associated complex remains landward or creek oriented.

The robustness result is clearest when the scenario families are compared numerically. Within the RF family, total suitable area ranges from 38,685.288 ha in RF2 to 49,615.862 ha in RF5. The coastline family is narrower, from 39,054.368 to 43,870.484 ha. The mangrove-base family spans 38,085.345 to 45,033.373 ha, and the scale family spans 43,870.484 to 53,593.943 ha. These contrasts show that threshold logic, support-mask choice, and scale do not perturb the model in the same way. A credible planning framework has to expose those differences rather than collapse them into a single uncertainty statement.

3.3 Blue carbon outcomes

3.3.1 Scenario based blue carbon outcomes

The blue carbon model translates those spatial differences into scenario dependent carbon outcomes. Under RF1, the 30 year cumulative outcome is 18.83 million t CO_2e in the conservative run, 22.09 million t CO_2e in the central run, and 25.27 million t CO_2e in the high run. The high and conservative estimates differ by 6.44 million t CO_2e, which is nearly 29% of the central estimate. This spread is not an error band around a field inventory. It is a planning envelope showing how strongly the interpretation depends on carbon parameter choice.

Group composition explains why area alone is insufficient. In the RF1 central run, Avicennia spp. contributes 45.1% of the cumulative modeled carbon outcome, even though it represents 32.8% of the RF1 suitable area. The Rhizophora group contributes 24.8% of the carbon outcome from only 14.1% of the area. Sonneratia spp. contributes 23.7% from 19.3% of the area. The Interior upper complex and Brackish freshwater complex cover 33.7% of the RF1 opportunity area together, yet their carbon contribution is much smaller under the present parameterization. This is the clearest evidence that ecological composition matters, not just hectares.

The RF1 decomposition also shows that the modeled outcome is dominated by sequestration rather than avoided loss. That result is a property of the selected λ, stock assumptions, and 30 year horizon. It should not be generalized to every mangrove landscape. In a site facing imminent clearing or severe degradation, avoided loss could dominate. In Soc Trang, under the assumptions used here, the planning value lies mainly in identifying where restoration and protection could sustain future accumulation.

The uncertainty pattern is orderly. Conservative, central, and high runs separate through time without producing erratic reversals. This is a useful diagnostic because it suggests that the model is responding to ecological group area and sequestration parameters in a coherent way. The dominant sensitivity to Avicennia and Rhizophora sequestration assumptions also points to a practical research priority: future fieldwork should measure growth and sediment carbon accumulation by ecological position, not only produce a single province wide carbon density.

The strongest numerical signal in the carbon results is the gap between spatial area and carbon response. Central outcomes range from 16.40 million t CO_2e in RF4 to 23.38 million t CO_2e in SC1, a 42.6% difference between the lowest and highest central scenarios. RF2 and RF4 are both about 25.8% below RF1, whereas SC1 is 5.8%

above it. The important point is not only that the totals change. They change according to group composition, which is why the carbon model cannot be replaced by a single hectare-based multiplier.

Table 6. Blue carbon outputs across the 19 scenarios under conservative, central, and high parameter settings

Scenario	Family	30 year cumulative t CO_2e, conservative	30 year cumulative t CO_2e, central	30 year cumulative t CO_2e, high	Relative change vs RF1 central (%)
RF1	Rule family	18,833,803.010	22,093,071.551	25,272,278.616	0.000
RF2	Rule family	13,496,460.336	16,406,488.359	19,241,281.502	-25.739
RF3	Rule family	17,896,289.491	21,749,454.174	25,521,341.144	-1.555
RF4	Rule family	14,268,429.223	16,400,932.933	18,460,136.656	-25.764
RF5	Rule family	16,653,598.099	19,841,050.819	22,950,904.498	-10.193
RF6	Rule family	16,049,246.031	19,079,861.047	22,032,575.657	-13.639
RF7	Rule family	17,821,064.475	21,053,175.703	24,180,313.903	-4.707
RF8	Rule family	15,639,740.080	18,604,558.768	21,518,340.866	-15.790
PC1	Coastline setting	16,420,622.279	19,537,534.052	22,576,545.419	-11.567
PC2	Coastline setting	14,758,099.465	17,665,193.386	20,502,422.657	-20.042
PC3	Coastline setting	15,570,326.767	18,276,148.837	20,917,753.669	-17.277
MB1	Mangrove base setting	15,120,860.294	17,879,534.227	20,576,535.445	-19.072
MB2	Mangrove base setting	14,775,484.580	17,469,472.131	20,103,763.955	-20.928
MB3	Mangrove base setting	14,705,040.907	17,390,540.134	20,017,040.607	-21.285
MB4	Mangrove base setting	16,815,561.972	20,008,143.262	23,120,592.037	-9.437
MB5	Mangrove base setting	16,420,622.279	19,537,534.052	22,576,545.419	-11.567
MB6	Mangrove base setting	16,318,418.306	19,417,921.396	22,440,725.476	-12.109
SC1	Scale	19,547,739.803	23,382,945.626	27,117,795.159	5.838
SC2	Scale	16,420,622.279	19,537,534.052	22,576,545.419	-11.567

Source: Authors' own work

Area and carbon ranking do not perfectly coincide. SC1 has the largest cumulative suitable area and the largest central carbon outcome, but its carbon per hectare is lower than RF1. RF4 retains 39,736.637 ha of cumulative suitable area, yet falls to 16.40 million t CO_2e because its group composition is less carbon-intensive. The model is therefore sensitive not only to how much area is classified as suitable, but also to which ecological group carries that area. This is precisely the kind of distinction that a species-informed framework is meant to reveal.

3.3.2 Economic interpretation of RF1 blue carbon outcomes

The RF1 modeled outcome is large enough to ask a financing question. Under the central parameterization, the model produces 22.09 million t CO_2e over 30 years. The purpose of the following calculation is to test whether that magnitude could matter for restoration finance. It is not a claim of issued credits. VM0033 and the VCS Program would require project specific validation, monitoring, additionality proof, leakage assessment, verification, and non permanence risk analysis [45,46]. The number reported here should therefore be read as VCU equivalent potential at pre-feasibility stage.

The modeled value already includes the protected fraction $\alpha = 0.60$ and the internal buffer $\beta = 0.20$. The screening conversion applies three further haircuts: additionality and policy-overlap risk, possible leakage, and the gap between a Tier 2 literature-based estimate and field-verified MRV. This is intentionally cautious. It keeps the calculation on the side of pre-feasibility finance screening rather than crediting-grade accounting.

$$VCU_{eq} = C_{modeled} \times (1 - \delta_{add}) \times (1 - \delta_{leak}) \times (1 - \delta_{unc}) \quad (1)$$

In this formulation, VCUeq is the screening-level VCU-equivalent volume and Cmodeled is the RF1 30-year cumulative mitigation estimate from Table 6. The three deduction terms represent additionality and policy-overlap risk, leakage risk, and model-to-field uncertainty/MRV risk. They are not presented as VM0033 equations. They are transparent screening assumptions used to avoid overstating the finance potential.

For the central case, the additionality and policy overlap haircut is 0.30, the leakage haircut is 0.05, and the uncertainty/MRV haircut is 0.20. The first value reflects a real policy problem: some restoration benefits may overlap with existing coastal protection forest mandates or public restoration programs in Vietnam [8,41,50]. The leakage deduction is deliberately modest because it is only a screening assumption, not a VM0033 leakage calculation. In VM0033, leakage depends on project specific activity shifting, market effects, and ecological conditions, and may be zero if the relevant requirements are met [45]. The uncertainty deduction is the most methodological of the three. It acknowledges that a Tier 2 estimate built from literature parameters will not behave like a field verified project inventory.

Under the central setting, the retention factor is 0.70 × 0.95 × 0.80 = 0.53. The resulting VCU equivalent volume is 11.75 million t CO_2e over 30 years. The internal $\beta = 0.20$ buffer is treated only as a screening level reserve. It should not be described as replacing the formal VCS non permanence buffer, which requires project specific assessment using the applicable AFOLU risk tool [46].

The economic translation uses three price scenarios: 15, 25, and 35 USD t CO_2e^{-1}. These values are not forecasts. They define a plausible financing envelope for a market where price depends on project quality, co-benefits, buyer demand, permanence risk, and verification status [47,48,54]. The useful question is therefore not whether one price is correct. The useful question is whether the RF1 outcome remains financially meaningful across a reasonable price range.

Table 7. Screening level VCU equivalent volume and gross revenue for the RF1 central parameterization

Scenario	δ_{add}	δ_{leak}	δ_{unc}	**Combined retention**	**30 year VCU equivalent volume (million t CO_2e)**	**30 year gross revenue at 15 / 25 / 35 USD t CO_2e^{-1} (million USD)**
Conservative	0.40	0.08	0.30	0.39	8.54	128 / 213 / 299
Central	**0.30**	**0.05**	**0.20**	**0.53**	**11.75**	**176 / 294 / 411**
Optimistic	0.20	0.03	0.10	0.70	15.43	231 / 386 / 540

Notes: All values use $C_{modeled}$ = 22.09 million t CO_2e, corresponding to the RF1 central estimate reported in Table 6. The central row is used as the reference case. The full envelope across the nine price and deduction combinations ranges from 128 to 540 million USD, equivalent to a factor of approximately 4.2 between the lowest and highest outcomes.

Source: Authors' calculations based on Table 6 and the screening assumptions described in the text.

The reference case combines the central deductions with the central price of 25 USD t CO_2e^{-1}. It yields 294 million USD over 30 years, or 9.8 million USD yr^{-1}. Spread across the full RF1 suitable area of 47,997 ha, this equals about 204 USD ha^{-1} yr^{-1}. If the denominator is restricted to the α treated area of 28,798 ha, the value rises to about 340 USD ha^{-1} yr^{-1}. The second denominator is more relevant for implementation because only the treated area is assumed to enter the accounting boundary.

The margin calculation is deliberately simple. With a central price of 25 USD t CO_2e^{-1}, illustrative MRV costs of 8-15 USD t CO_2e^{-1} leave 10-17 USD t CO_2e^{-1} before establishment, maintenance, transaction, financing, tenure, and governance costs [47,50]. Applied to the reference case, that range corresponds to about 3.9-6.7 million USD yr^{-1}. The result is not a profitability claim, but it does show that the carbon signal is large enough to justify a more detailed project-finance assessment.

The timing of revenue is also uneven. The logistic sequestration term uses $k = 0.32\ yr^{-1}$ and $t_0 = 4$ yr. As a result, years 1-3 contribute only about 4% of the 30 year cumulative total, years 4-20 contribute about 57%, and years 21-30 contribute about 39%. This pattern is financially important. A project would need upfront capital during establishment, while a substantial part of the carbon benefit arrives only after the stand has survived and developed.

Actual VCU issuance would require a different level of evidence. A project would need a defensible boundary, baseline scenario, proof of additionality, field allometry, survival monitoring, relevant soil measurements, leakage accounting, uncertainty quantification, non-permanence risk assessment, independent validation, and periodic verification under VCS and VM0033 requirements [45,46]. The screening result should therefore be read as a financing hypothesis, not as a claim of immediately issuable credits.

4. Discussion

The main contribution of this study is the way it joins ecological suitability with blue carbon interpretation. The framework does not begin with carbon. It begins with the question of whether each taxon is plausible in a particular part of Soc Trang. Only after that ecological question is answered does the model translate area into carbon outcomes. This order matters because carbon estimates are only meaningful if the underlying restoration or conservation area is ecologically defensible.

The rule based strategy is also a methodological choice. It sits between generic global thresholds and purely statistical niche models. Generic thresholds risk ignoring the geomorphic and hydrological character of the Mekong Delta. Purely statistical models can be powerful, but their ecological meaning may be difficult to inspect when field occurrence data are sparse or biased by past land use. The rules used here are simpler than a mechanistic hydrodynamic model, but they are visible. That visibility is valuable in restoration planning because successful mangrove rehabilitation depends on matching species, hydrology, sediment setting, and site conditions rather than treating planting as a generic operation [35].

Machine learning still has an important role. Recent remote sensing studies show strong performance for mangrove mapping, and systematic reviews emphasize the growing value of remote sensing for blue carbon estimation [6,16]. The argument is more specific: a restoration framework should not outsource ecological judgment to the classifier. In this study, classification supports the boundary and masks, while the species rules keep the restoration logic visible.

The Soc Trang results also show why scenario testing should be standard practice. The difference between MB3 and SC1 is 15,508.598 ha of cumulative suitable area. The difference between RF4 and SC1 in central carbon outcome is 6.98 million t CO_2e. These are not minor formatting changes in a GIS workflow. They are large enough to affect restoration priorities, expected carbon benefit, and the credibility of any financing narrative built on the map.

The blue carbon results should be interpreted with the same caution. A central RF1 outcome of 22.09 million t CO_2e over 30 years is substantial, but it is not a credit issuance number. The economic translation gives a reference gross revenue of 294 million USD under the central price and deduction assumptions. That number is useful precisely because the manuscript also shows how quickly it can change: the Table 7 envelope spans 128 to 540 million USD. A serious planning framework must present both the opportunity and the uncertainty.

The strongest scientific signal is the role of sequestration assumptions. Under the present formulation, stock terms matter less than the group specific accumulation trajectories. This finding gives a clear direction for field research. A future Soc Trang campaign should measure stand development, sediment accretion, and soil carbon accumulation along the same ecological gradient used in the model. More local carbon density values would help, especially given the broad environmental variation in mangrove ecosystem carbon stocks [44], but they would not resolve the main uncertainty if sequestration trajectories remain generic.

The limitations are substantial. Coastal distance and elevation do not measure salinity, inundation frequency, porewater chemistry, sediment texture, or seedling survival. The overlapping suitability design does not model competition, facilitation, or planting sequence. The carbon module remains literature based and should not be mistaken for field inventory. These limitations, however, are not hidden. They are part of the framework's honesty. The model is meant to support early decisions and guide field priorities, not to replace the field.

The results point to a practical research direction for deltaic mangrove planning. Species suitability, spatial robustness, and carbon value are not separate problems. This integrated view is consistent with recent calls for mangrove research that crosses ecological, geomorphic, monitoring, and governance boundaries [20]. When these dimensions are joined carefully, the output becomes more than a map and more than a carbon total. It becomes a defensible way to ask where restoration is ecologically plausible, how fragile that judgment is, and whether the resulting opportunity is large enough to matter for climate finance.

5. Conclusions

This study developed an integrated framework linking species specific mangrove habitat suitability with scenario based blue carbon modeling in Soc Trang Province. The central contribution is not simply a set of maps or a carbon total. It is the transparent chain that begins with species ecology, passes through local rules based on coastal distance and elevation, tests 19 spatial assumptions, and only then converts ecological opportunity into carbon interpretation.

Under RF1, cumulative group level suitable area reached 47,997.312 ha. The central 30 year carbon outcome was 22.09 million t CO_2e, bracketed by 18.83 and 25.27 million t CO_2e under conservative and high settings. Across all scenarios, central outcomes ranged from 16.40 to 23.38 million t CO_2e. The economic screening showed a central VCU equivalent volume of 11.75 million t CO_2e and a reference gross revenue of 294 million USD over 30 years, but only under explicitly stated screening assumptions.

The broader implication is straightforward. In dynamic coastal settings such as Soc Trang, blue carbon planning should not begin with a generic hectare count. It should begin with species plausibility, then test how stable that judgment remains under alternative spatial assumptions, and only then translate the result into carbon-oriented decision support. That sequence does not remove uncertainty. It makes uncertainty visible enough to be used.

CRediT authorship contribution statement

Manh-Dung Vu: Conceptualization, Methodology, Software, Formal analysis, Investigation, Data curation, Visualization, Writing – original draft, Writing – review & editing, Project administration.

Ming-Che Hu: Supervision, Conceptualization, Methodology, Writing – review & editing, Project administration.

Hoang-Minh Cong To: Writing – review & editing, Validation.

Harvey Hoang-Anh Ly: Writing – review & editing, Validation.

Thanh-Cong To: Writing – review & editing, Validation.

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Declarations

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References

1. Friess DA, Rogers K, Lovelock CE, Krauss KW, Hamilton SE, Lee SY, et al. The State of the World's Mangrove Forests: Past, Present, and Future. Annu Rev Environ Resour. 2019;44:89-115. Available from: https://doi.org/10.1146/annurev-environ-101718-033302
2. Bunting P, Rosenqvist A, Hilarides L, Lucas RM, Thomas N, Tadono T, et al. Global Mangrove Extent Change 1996 to 2020: Global Mangrove Watch Version 3.0. Remote Sens. 2022;14(15):3657. Available from: https://doi.org/10.3390/rs14153657
3. Hagger V, Friess DA, Krauss KW, Goldberg L, Kennedy H, Whelan KRT, et al. Drivers of global mangrove loss and gain in social ecological systems. Nat Commun. 2022;13:7562. Available from: https://doi.org/10.1038/s41467-022-33962-x
4. Anthony EJ, Brunier G, Besset M, Goichot M, Dussouillez P, Nguyen VL. Linking rapid erosion of the Mekong River delta to human activities. Sci Rep. 2015;5:14745. Available from: https://doi.org/10.1038/srep14745
5. Minderhoud PSJ, Erkens G, Pham VH, Bui VT, Erban LE, Kooi H, et al. Impacts of 25 years of groundwater extraction on subsidence in the Mekong delta, Vietnam. Environ Res Lett. 2017;12:064006. Available from: https://doi.org/10.1088/1748-9326/aa7146
6. Tran TV, Reef R, Zhu X, Gunn A. Characterising the distribution of mangroves along the southern coast of Vietnam using multispectral indices and a deep learning model. Sci Total Environ. 2024;923:171367. Available from: https://doi.org/10.1016/j.scitotenv.2024.171367
7. Duong PT, Van Trung P, Nguyen HQ, et al. The state of mangrove forest in Soc Trang Province, Vietnam based on satellite imagery between 2000 and 2020. J Trop For Sci. 2024;36(2):224-236. Available from: https://jtfs.frim.gov.my/jtfs/article/view/2725
8. Tinh PH, Hai PM, Huyen BT. Mangrove restoration in Vietnamese Mekong Delta during 2015 to 2020: achievements and challenges. Front Mar Sci. 2022;9:1043943. Available from: https://doi.org/10.3389/fmars.2022.1043943
9. McLeod E, Chmura GL, Bouillon S, Salm R, Björk M, Duarte CM, et al. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. Front Ecol Environ. 2011;9(10):552-560. Available from: https://doi.org/10.1890/110004
10. Donato DC, Kauffman JB, Murdiyarso D, Kurnianto S, Stidham M, Kanninen M. Mangroves among the most carbon rich forests in the tropics. Nat Geosci. 2011;4:293-297. Available from: https://doi.org/10.1038/ngeo1123
11. Alongi DM. Carbon sequestration in mangrove forests. Carbon Manag. 2012;3(3):313-322. Available from: https://doi.org/10.4155/cmt.12.20
12. Pendleton L, Donato DC, Murray BC, Crooks S, Jenkins WA, Sifleet S, et al. Estimating global blue carbon emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One. 2012;7(9):e43542. Available from: https://doi.org/10.1371/journal.pone.0043542
13. Macreadie PI, Costa MDP, Atwood TB, Friess DA, Kelleway JJ, Kennedy H, et al. Blue carbon as a natural climate solution. Nat Rev Earth Environ. 2021;2:826-839. Available from: https://doi.org/10.1038/s43017-021-00224-1
14. Gao G, Beardall J, Bao M, et al. A review of existing and potential blue carbon contributions to climate change mitigation in the Anthropocene. J Appl Ecol. 2022;59(7):1693-1709. Available from: https://doi.org/10.1111/1365-2664.14173
15. Song S, Ding Y, Li W, Meng Y, Zhou J, Gou R, et al. Mangrove reforestation provides greater blue carbon benefit than afforestation for mitigating global climate change. Nat Commun. 2023;14:756. Available from: https://doi.org/10.1038/s41467-023-36477-1
16. Roy AD, Pitumpe Arachchige PS, Watt MS, Kale A, Davies M, Heng JE, et al. Remote sensing based mangrove blue carbon assessment in the Asia Pacific: a systematic review. Sci Total Environ. 2024;938:173270. Available from: https://doi.org/10.1016/j.scitotenv.2024.173270

17. Sanderman J, Hengl T, Fiske GJ, Solvik K, Adame MF, Benson L, et al. A global map of mangrove forest soil carbon at 30 m spatial resolution. Environ Res Lett. 2018;13:055002. Available from: https://doi.org/10.1088/1748-9326/aabe1c
18. Dabalà A, Dahdouh Guebas F, Dunn DC, Friess DA, Lovelock CE, Goldberg L, et al. Priority areas to protect mangroves and maximise ecosystem services. Nat Commun. 2023;14:5747. Available from: https://doi.org/10.1038/s41467-023-41333-3
19. Rodríguez Rodríguez JA, Costa MDP, Wartman M, Rasheed AR, Palacios M, Macreadie PI. Global trends in applying decision science in mangrove restoration: are we missing some dimensions? Ocean Coast Manag. 2024;254:107172. Available from: https://doi.org/10.1016/j.ocecoaman.2024.107172
20. Dahdouh Guebas F, Friess DA, Cannicci S, Diele K, Lovelock CE, Ragavan P, et al. Cross cutting research themes for future mangrove forest research. Nat Plants. 2022;8:1131-1135. Available from: https://doi.org/10.1038/s41477-022-01245-4
21. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Bangkok: FAO and Wetlands International; 2007. Available from: https://www.fao.org/4/ag132e/ag132e00.pdf
22. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Trees and shrubs Part 1. Bangkok: FAO and Wetlands International; 2007. Available from: https://openknowledge.fao.org/server/api/core/bitstreams/91c0ba88-e9e0-4120-87c3-d91a6f08a43a/content
23. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Trees and shrubs Part 2. Bangkok: FAO and Wetlands International; 2007. Available from: https://www.fao.org/4/ag132e/ag132e08.pdf
24. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Trees and shrubs Part 3. Bangkok: FAO and Wetlands International; 2007. Available from: https://www.fao.org/4/ag132e/ag132e09.pdf
25. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Trees and shrubs Part 4. Bangkok: FAO and Wetlands International; 2007. Available from: https://www.fao.org/4/ag132e/ag132e10.pdf
26. Giesen W, Wulffraat S, Zieren M, Scholten L. Mangrove guidebook for Southeast Asia. Palms, cycads and pandans. Bangkok: FAO and Wetlands International; 2007. Available from: https://www.fao.org/4/ag132e/ag132e07.pdf
27. Ma W, Krolik J, Gao Y, Guo H, Xiao P, Li B, et al. Zonation of mangrove flora and fauna in a subtropical estuarine wetland based on surface elevation. Ecol Evol. 2020;10:9164-9178. Available from: https://doi.org/10.1002/ece3.6467
28. Wang W, Wang M, Huang M, Tseng YH, Lin KH, Yang IH, et al. Propagule dispersal determines mangrove zonation at intertidal and estuarine scales. Forests. 2019;10(3):245. Available from: https://doi.org/10.3390/f10030245
29. Clarke PJ, Myerscough PJ. Testing patterns of zonation in mangroves: scale dependence and environmental correlates in the forests of southeastern Australia. J Ecol. 2001;89(5):813-824. Available from: https://doi.org/10.1046/j.1365-2745.2001.00500.x
30. Bunt JS, Williams WT, Clay HJ. Overlap in mangrove species zonal patterns: some methods of analysis. Hydrobiologia. 1999;413:237-248. Available from: https://doi.org/10.1023/A:1009995426021
31. Elith J, Leathwick JR. Species distribution models: ecological explanation and prediction across space and time. Annu Rev Ecol Evol Syst. 2009;40:677-697. Available from: https://doi.org/10.1146/annurev.ecolsys.110308.120159
32. Raiter KG, Hawlena D. Managing multiple uncertainties in species distribution modelling. Divers Distrib. 2024;30:e13857. Available from: https://doi.org/10.1111/ddi.13857
33. Harris J, Pirtle JL, Laman EA, Siple MC, Thorson JT. An ensemble approach to species distribution modelling reconciles systematic differences in estimates of habitat utilization and range area. J Appl Ecol. 2024;61:351-364. Available from: https://doi.org/10.1111/1365-2664.14559
34. Shih SS, Yang CH, Lee HY. Spatial habitat suitability models of mangroves with Kandelia obovata. Forests. 2020;11(4):477. Available from: https://doi.org/10.3390/f11040477
35. Lewis RR III. Ecological engineering for successful management and restoration of mangrove forests. Ecol Eng. 2005;24(4):403-418. Available from: https://doi.org/10.1016/j.ecoleng.2004.10.003
36. Breiman L. Random forests. Mach Learn. 2001;45:5-32. Available from: https://doi.org/10.1023/A:1010933404324
37. Belgiu M, Drăguţ L. Random forest in remote sensing: a review of applications and future directions. ISPRS J Photogramm Remote Sens. 2016;114:24-31. Available from: https://doi.org/10.1016/j.isprsjprs.2016.01.011
38. Blaschke T. Object based image analysis for remote sensing. ISPRS J Photogramm Remote Sens. 2010;65(1):2-16. Available from: https://doi.org/10.1016/j.isprsjprs.2009.06.004
39. Wang L, Sousa WP, Gong P. Integration of object based and pixel based classification for mapping mangroves with IKONOS imagery. Int J Remote Sens. 2004;25(24):5655-5668. Available from: https://doi.org/10.1080/014311602331291215

40. Ghorbanian A, Zaghian S, Mohammadzadeh A, Jamali S, Feizizadeh B, Hashemi H. Mangrove ecosystem mapping using Sentinel 1 and Sentinel 2 satellite images and random forest algorithm in Google Earth Engine. Remote Sens. 2021;13(13):2565. Available from: https://doi.org/10.3390/rs13132565

41. Phan MH, Stive MJF. Managing mangroves and coastal land cover in the Mekong Delta. Ocean Coast Manag. 2022;219:106013. Available from: https://doi.org/10.1016/j.ocecoaman.2021.106013

42. Tran TV, Reef R, Zhu X. Long term changes of mangrove distribution and its response to anthropogenic impacts in the Vietnamese Southern Coastal Region. J Environ Manage. 2024;370:122658. Available from: https://doi.org/10.1016/j.jenvman.2024.122658

43. Alongi DM. Global significance of mangrove blue carbon in climate change mitigation. Sci. 2020;2(3):67. Available from: https://doi.org/10.3390/sci2030067

44. Kauffman JB, et al. Total ecosystem carbon stocks of mangroves across broad global environmental and physical gradients. Ecol Monogr. 2020;90(2):e01405. Available from: https://doi.org/10.1002/ecm.1405

45. Verra. VM0033 Methodology for Tidal Wetland and Seagrass Restoration, v2.1. Verified Carbon Standard. 2023. Available from: https://verra.org/methodologies/vm0033-methodology-for-tidal-wetland-and-seagrass-restoration-v2-1/

46. Verra. AFOLU Non-Permanence Risk Tool, v4.0. Verified Carbon Standard. 2019. Available from: https://verra.org/wp-content/uploads/2019/09/AFOLU_Non-Permanence_Risk-Tool_v4.0.pdf

47. Perera N, et al. Trends in market-based blue carbon projects. Sustain Dev. 2025;33:3377-3402. Available from: https://doi.org/10.1002/sd.3293

48. Farahmand S, Hilmi N, Duarte CM. The rise and flows of blue carbon credits advance global climate and biodiversity goals. npj Ocean Sustain. 2025;4:39. Available from: https://doi.org/10.1038/s44183-025-00141-6

49. Global Mangrove Alliance. The State of the World's Mangroves 2024. 2024. Available from: https://www.mangrovealliance.org/mangrove-forests

50. Hai NT, Dell B, Phuong VT, Harper RJ. Towards a more robust approach for the restoration of mangroves in Vietnam. Ann For Sci. 2020;77:18. Available from: https://doi.org/10.1007/s13595-020-0921-0

51. Bessely SM, van der Wegen M, Reyns J, Grueters U, Dijkstra JT, Roelvink D. Strategic mangrove restoration increases carbon stock capacity. Commun Earth Environ. 2025;6:359. Available from: https://doi.org/10.1038/s43247-025-02401-2

52. Kwan V, Friess DA, Sarira TV, Zeng Y. Permanence risks limit blue carbon financing strategies to safeguard Southeast Asian mangroves. Commun Earth Environ. 2025;6:57. Available from: https://doi.org/10.1038/s43247-025-02035-4

53. Lewis RR, Brown B. Ecological Mangrove Rehabilitation: a field manual for practitioners. Mangrove Action Project, Canadian International Development Agency, and OXFAM; 2014. Available from: https://blue-forests.org/wp-content/uploads/2020/04/Whole-EMR-Manual-English.pdf

54. Ecosystem Marketplace. State of the Voluntary Carbon Market 2025. Forest Trends; 2025. Available from: https://www.ecosystemmarketplace.com/publications/2025-state-of-the-voluntary-carbon-market-sovcm/